Constructed Farm Wetlands (CFWs) designed for remediation of farmyard runoff: an evaluation of their water treatment efficiency, ecological value, costs and benefits

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Declaration

I hereby declare that this Thesis is my own work, except where otherwise stated, and has not been submitted in any form for another degree, diploma or professional qualification at any university or other institution.

Information derived from the published or unpublished work of others has been thoroughly acknowledged and referenced.

Edinburgh, September 2009

Gouriveau.

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Abstract

Farmyard runoff, i.e. the effluent generated by the rain falling over farmyards, tracks and roofs, is a significant and overlooked source of nutrients and pathogens which degrades aquatic ecosystems through eutrophication, siltation and wildlife poisoning, raises public health concerns, and incurs considerable costs for society. Among other Best Management Practices implemented to address agricultural water pollution and help achieve compliance with the Water Framework Directive, Constructed Farm Wetlands (CFWs), i.e. shallow surface flow wetlands comprising several vegetated cells in series, are being recommended for remediation of farmyard runoff, due to their capacity to remove or store pollutants. Investigation is therefore needed of their long-term water treatment efficiency and ecological value to optimize their design and cost-effectiveness and minimize their negative externalities.

The main aims of this study were to: 1) evaluate the treatment performance of CFWs and the link between design, hydrology and efficiency; 2) assess their ecological value and the influence of water quality and design on wetland ecology; 3) identify their costs, benefits and the way they are perceived by farmers; and 4) inform guidelines for the design, construction and aftercare of sustainable CFWs.

Research focused on two CFWs in south-east Scotland, one at a dairy farm and one at a mixed beef-arable farm, which receive runoff from yards and roofs, field drainage and septic tank overflow. From February 2006 to June 2008, rainfall, evaporation, water levels and flow at the CFWs were monitored, and their treatment efficiency was assessed from water samples collected manually regularly or with automatic samplers during storm events, and analysed using standard methods. In addition, their ecological value was assessed twice a year from vegetation and aquatic macroinvertebrate surveys. Finally, semi-structured interviews with eight farmers and a farm advisor and discussions with three CFW designers in Scotland and Ireland allowed collection of technical and economic data on farm practices, CFW construction and maintenance, and helped assess CFW cost-effectiveness and acceptance by farmers.

Both CFWs reduced pollutant concentrations between inlet and outlet, with efficiencies at CFW1 and CFW2 respectively of 87% and < 0% for five-day biochemical oxygen demand, 86% and 83% for suspended solids, 68% and 26% for nitrate/nitrite, 42% and 34% for ammonium, and 12% and 31% for reactive phosphorus. Nevertheless, the concentration of all pollutants at the outlet of CFW1, and concentration of nitrate/nitrite at the outlet of CFW2 frequently exceeded river water quality standards. Water treatment efficiency varied seasonally, being significantly lower in winter, mainly due to lower temperatures, increased volume of inputs and reduced residence time.

The ecological value of the two CFWs differed greatly. At CFW1 and CFW2 respectively, 14 and 22 wetland plant species and 24 and 46 aquatic macroinvertebrate species (belonging to 13 and 27 BMWP scoring families respectively) were recorded, illustrating the greater biodiversity conservation value of CFW2, which was one year older, larger, cleaner, comprised several ponds with a combination of open water and densely vegetated areas, and was subsequently more structurally diverse.

The socio-economic study revealed that, despite significant costs associated with their construction (£20 000-£50 000 ha^{-1}) and maintenance (£900-£1500 ha^{-1} yr^{-1}), CFWs may still represent a more cost-effective alternative than conventional methods. However, their adoption, implementation and sustainable use by farmers were conditioned by land availability and suitability, existing farm infrastructure, detailed information on limitations and maintenance requirements, and adequate financial support for both construction and aftercare.

To ensure a long-term, consistent and efficient water treatment, and to enhance biodiversity and landscape, well-maintained, large, vegetated, multi-cell CFWs with shallow overflows are recommended. Their size should be adapted to local precipitation patterns and catchment characteristics.

Keywords: agriculture, best management practice (BMP), biodiversity, constructed farm wetland (CFW), costs, farmyard runoff, water pollution, water treatment.

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Si l'Homme déployait plus d'efforts pour préserver la Nature que pour la réinventer, alors peut-être la Terre aurait-elle une chance…

Table of Contents

List of Tables

List of Figures

Chapter 1: Introduction - Agricultural Water Pollution

The chapter introduces the issue of agricultural diffuse water pollution, clarifying its origin, impacts and cost. It gives an overview of the main policy and legislation addressing water pollution, focuses on the legislation for disposal of farmyard dirty water in the UK, presents the most common BMPs for control and remediation of diffuse pollution and underlines the potential role of Constructed Farm Wetlands (CFWs) for farmyard dirty water treatment. It finally details the specific objectives of the research, main hypotheses, and summarizes the structure of the Thesis.

1.1 Agricultural diffuse water pollution: origin, impacts and cost

1.1.1 The origin and extent of agricultural water pollution

In the UK, the impacts of most agricultural "point sources" of water pollution have been mitigated during recent decades. Indeed, the collection, storage and land spreading of farm effluents such as parlour washings, slurry from slatted buildings and silage runoff are dealt with under strict regulations, e.g. Control of Pollution (Silage, Slurry and Agricultural Fuel Oil) Regulations 1991 (SSAFO), and guidance has been developed for farmers to attenuate the risks associated with the management of those effluents, e.g. the Four Point Plan (SEERAD *et al.*, 2004), Prevention of Environmental Pollution From Agricultural Activity Code (PEPFAA) (Scottish Executive, 2005a), the Code of Good Agricultural Practice for the Prevention of Pollution of Water (DARD, 2008), or more recently, Protecting our water, soil and air: Code of Good Agricultural Practices for farmers, growers and land managers (DEFRA, 2009).

Current legislation and European directives such as the Bathing Water Directive (BWD, 76/160/EEC), Nitrates Directive (ND, 91/676/EEC) and Water Framework Directive (WFD, 2000/60/EEC) require significant efforts to address "diffuse pollution", which occurs when rainfall, snowmelt or irrigation water runs over land and impervious surfaces (e.g. farmyards, roofs) or percolates through the ground, picking up nutrients, faecal pathogens, suspended solids, organic matter and pesticides, transporting them and eventually depositing them into rivers, lakes, coastal waters or groundwater (Brewer *et al.*, 1999; Cumby *et al.*, 1999; Neumann *et al.*, 2000; SEPA, 2009a).

Diffuse pollution arises from urban and rural land-use activities that are dispersed across a catchment or sub-catchment, and does not arise as a process effluent, municipal sewage effluent, deep mine or farm effluent discharge. It also includes pollution caused by a multiplicity of dispersed, often individually minor, point sources, whose collective environmental and economic impacts are significant at the catchment scale (D'Arcy *et al.*, 2000; DEFRA, 2002; Scottish Executive, 2005b; SEPA, 2009a). In contrast to point source pollution, diffuse pollution is chronic and insidious, difficult to identify, and therefore, difficult to monitor, contain and cure. It is nowadays one of the main causes of water pollution and its impacts seem to be increasing, both in terms of magnitude and frequency (D'Arcy *et al.*, 2000).

Studies in the UK and Ireland have shown that agricultural water pollution is strongly linked to farm type and management and to local environmental conditions. It is exacerbated by the excessive use of soluble inorganic fertilisers, inadequate animal diets, inappropriate storage and spreading of steading runoff, manure and slurry, leaking of silage pits, accidental spillages, mismanagement of soils, filling and washing of pesticide sprayers over hard surfaces and intensive use and misuse of pesticides (Schofield *et al.*, 1990; Harris *et al.*, 1991; Brewer *et al.*, 1999; Cumby *et al.*, 1999; DAFRD, 2000; Withers *et al.*, 2000, Dunne *et al.*, 2005a, b).

The composition of farmyard runoff, including for example runoff from farm buildings, livestock tracks or collecting areas, roofs and septic overflows, is very variable and can reach potentially high and harmful concentrations (Table 1.1). It depends mainly on the cleanliness of the yard, influenced by the farm type (higher contamination is expected in dairy farms than in arable ones), number of cows and frequency of scraping, and on the dilution with other inputs (e.g. roof or field drainage).

Table 1.1 Quality of the farmyard runoff entering Integrated Constructed Wetlands (ICWs) in Ireland (Scholz *et al.*, 2007a) and Constructed Farm Wetlands (CFWs) in Scotland (Edwards *et al.*, 2008).

Water quality parameter	Farmyard runoff mean concentration (± 1 SD) (13 ICWs)	Farmyard runoff mean concentration (± 1 SD) (4 CFWs)
Chemical oxygen demand	1790 ± 2250 mg l^{-1}	NA
Biochemical oxygen demand	791 ± 1590 mg l^{-1}	NA
Suspended solids	358 ± 318 mg l^{-1}	NA
Total dissolved nitrogen	NA	101 ± 318 mg l^{-1}
Ammonia-nitrogen	68.7 ± 48.9 mg l^{-1}	70.4 ± 29.3 mg l^{-1}
Nitrate-nitrogen	NA	1.24 ± 0.66 mg l^{-1}
Total dissolved phosphorus	NA	4.73 ± 1.07 mg l^{-1}
Dissolved reactive phosphorus	19.8 ± 20.9 mg l^{-1}	3.90 ± 1.10 mg l^{-1}
Dissolved organic phosphorus	NA	1.40 ± 0.54 mg l^{-1}
E. coli	833 400 ± 2 022 000 cfu^{1} 100 ml^{-1}	

SD: standard deviation; NA: not available; cfu: colony forming unit.

Additionally, the large scale drainage of agricultural land across Europe and over-abstraction by water companies has exacerbated the issue of diffuse pollution, leading to a dramatic reduction in the area of natural wetlands and subsequently to the loss of many of the ecosystem services they provide, such as flood control, water quality improvement or habitat provision (Schuyt and Brander, 2004; Wetland Vision, 2008). Moreover, drains are often direct pathways for contaminated runoff (e.g. after slurry application) between cultivated land or farmyards and freshwater bodies, reducing the potential for water treatment.

In the Scotland River Basin District, which covers 113 920 km^2 of land and water from Shetland in the North to Glasgow, Ayr and Edinburgh, agriculture impedes the quality of 69% of the rivers, 56% of the lochs, 62% of transitional waters, 47% of the coastal waters and 100% of the groundwater (SEPA, 2007).

1.1.2 The impacts of agricultural water pollution

The impacts of agricultural pollution vary spatially and temporally, mainly depending on the "assimilative capacity" of receiving water bodies (e.g. linked to their size, flow, sensitivity to nutrient enrichment), climate (influences mobilisation, dilution and degradation of pollutants and biological processes), soil type (affects pollutant retention) and land use. Pollution results in the degradation of water quality and aquatic ecosystems, through eutrophication, contamination of groundwater, siltation and direct toxicity to organisms (Harper, 1992; Mason, 2002; Angelier, 2003). It consequently affects fisheries, angling, tourism, biodiversity and causes public health concerns (Horne and Dunson, 1995; D'Arcy *et al.*, 2000; Dodds, 2002; Nicolet *et al.*, 2004). Examples of impacts of agricultural pollution in the UK include algal blooms in freshwaters in England and Wales in summer 1989 and blooms of cyanobacteria in Loch Leven (Scotland) in June 1992 (Haygarth *et al.*, 2000). The major pollutants of concern and their main impacts are presented below.

Organic wastes such as livestock manure and slurries, milk or silage effluents contribute to microbial contamination, oxygen depletion (expressed as Biochemical or Chemical Oxygen Demand), water acidification and cause reduced fitness and asphyxiation of fish and invertebrates (Horne and Dunson, 1995; Klein, 1996; Richardson, 1976; Bloxham, 1999).

Phosphorus is the main factor limiting the growth of plants and algae in freshwater. Its excess triggers eutrophication, which alters ecosystem productivity (Richardson and Qian, 1999; Clarke and Baldwin, 2002), reduces biodiversity (Sloey *et al.*, 1978), affects water transparency, light penetration and photosynthesis, and triggers blooms of algae and cyanobacteria which produce toxins affecting organisms and causing health problems (EC, 1991a; EA, 1998; Haygarth *et al.*, 2000; Withers *et al.*, 2000; Dodds, 2002; Mason, 2002).

Nitrogen**,** along with phosphorus, contributes to eutrophication (Winkler, 1981) and nitrogenous compounds undergo aerobic reactions that exert oxygen demand: organic compounds are converted to ammonium which is oxidised to nitrites and finally to nitrates. Nitrates may affect human health (Mason, 2002), and together with nitrites, they have direct noxious effects on freshwater organisms such as amphibians, affecting their reproduction and survival (Johansson *et al.*, 2001; Camargo *et al.*, 2005). Ammonia is toxic to aquatic organisms at concentrations as low as 0.2 mg l^{-1} and contributes to water acidification which affects in turn decomposition rates, invertebrate communities, phytoplankton, vertebrates and availability of toxic metals (Dodds, 2002; Mason, 2002). When ammonium is biologically oxidised to nitrate, it exerts a nitrogenous oxygen demand of 4.3 g of oxygen per g of ammonium (Henze *et al.*, 1995).

Faecal Indicator Organisms (FIOs) such as faecal coliforms (e.g. *Escherichia coli*) and faecal *streptococci* are one of the main causes for many of the 80 designated bathing waters in Scotland failing the EC Bathing Water standards (Aitken, 2000; Kay *et al.*, 2001; Scottish Executive, 2002a; Vinten *et al.*, 2002 and 2003). Other pathogens such as *Cryptosporidium*, *Campylobacter* and *Giardia intestinalis* also represent a threat to drinking water supplies or to water used for irrigation purposes (Groves *et al.*, 2002). Microbial contamination is spatially and temporally linked to farming activities such as calving or field spreading of manure, but is also influenced by wild animals (Bodley-Tickell *et al.*, 2002).

Chemical pesticides (e.g. herbicides, insecticides, sheep dip) negatively affect invertebrates (Thiere and Schulz, 2004), impair the immune system of amphibians (Christin *et al.*, 2004) and represent a human health hazard (Virtue and Clayton, 1997; Williams and Croxford, 2000).

Sediment (e.g. clay, silt, organic and inorganic matter) washed from farm tracks or yards, arable land and river banks into water courses can lower primary productivity by shading algae and macrophytes, affects aquatic animals by impeding respiration, reproduction, fish spawning and thereby decreases the ecological and amenity value of streams and lakes (Dodds, 2002). It adversely affects economic uses of waterways, rivers, reservoirs through sedimentation, siltation of water intakes, loss of storage capacity, and incurs high costs. Sediment also binds and transports potentially harmful compounds (SOAEFD, 1998; Ferrier and Ellis, 2000).

1.1.3 The cost of agricultural water pollution

The costs incurred by water pollution involve direct or indirect market as well as non-market costs and are therefore difficult to quantity. Market costs include higher food prices due to lower productivity, loss of profit from fisheries or tourism, increased costs related to water treatment, cleaning of waterways and loss of reservoir storage capacity. Non-market costs include the loss of biodiversity and amenity, and emissions of greenhouse gases (Pretty *et al*., 2003). Estimates are very variable but indicate that millions of pounds are spent each year in the UK to treat water, clean waterways and reservoirs and to monitor water quality (Ferrier and Ellis, 2000; Williams and Croxford, 2000). Skinner *et al.* (1997) estimated the cost for the UK to comply with nitrate drinking water standards at £199 M over 20 years (from 1997 onwards), pesticides removal cost UK water companies £500 M between 1992 and 1994 (D'Arcy *et al.*, 2000), and overall, Pretty *et al*. (2003) valued the damage due to eutrophication in England and Wales between £75 and £114 million per year.

1.2 Key European policy and legislation addressing water pollution

During the last 20 to 30 years, European Water legislation has evolved quickly and became more stringent, exerting greater pressure on the authorities and farm businesses. Several Directives have been adopted to address water pollution in a more holistic and integrated manner.

1.2.1 Bathing Water Directive

The Bathing Water Directive (BWD, 76/160/EEC) aims to protect public health and the environment from faecal pollution at bathing waters. It requires European Member States to identify popular bathing areas and to monitor water quality throughout the bathing season. It sets microbiological and physico-chemical standards that bathing waters must either comply with (mandatory) or endeavour to meet (guideline). In March 2006, it was revised to introduce tighter microbiological standards to be met by 2015, action plans to improve water quality in failing beaches, and to improve public awareness (EC, 2006).

1.2.2 Nitrates Directive and Nitrate Vulnerable Zones

In accordance with the requirements of the Nitrates Directive (91/676/EEC) (EC, 1991b), Nitrate Vulnerable Zones (NVZs) were created in 2002 in Scotland in areas where the concentration of nitrates in surface or groundwater is expected to be higher than 50 mg l^{-1} or to trigger eutrophication. NVZs cover c. 14% of Scottish agricultural land (Scottish Executive, 2002b). Binding rules, known as Action Programmes, are implemented in NVZs to reduce and prevent nitrate from agricultural sources polluting the water environment. The application of fertilisers and manures is restricted in space and time, monitored and controlled and the adoption of Good Practices by farmers is strongly encouraged (SEERAD, 2001).

1.2.3 Water Framework Directive

The Water Framework Directive (WFD, 2000/60/EC) requires European Member States to establish a programme of measures (operational by 2012), within defined River Basin Districts (RBDs) to protect, enhance and restore water bodies and achieve good ecological and chemical status by 2015 in all inland (rivers, lochs), transitional (estuaries) and coastal waters. Measures include controls over point and diffuse sources of pollution, groundwater recharge, abstractions, impoundments and engineering works on water bodies (EC, 2000; DEFRA, 2006a). Scotland is covered by three RBDs: the Scotland RBD covering most of the country, the Solway-Tweed RBD and the Northumbria RBD (Scottish Executive, 2003 and 2004).

The Water Environment and Water Services (Scotland) Act 2003 (WEWS Act) transposed the WFD into Scots law and introduced regulatory controls over activities to improve Scotland's water environment whilst supporting the social and economic interests of people who depend on it (SEPA, 2006). Since April 2006, the Water Environment (Controlled Activities) Regulations (CAR) control activities liable to cause water pollution, abstraction of water, construction, alteration or operation of impounding works in or in the vicinity of surface waters or wetlands (Scottish Executive, 2005c; SEPA, 2006). CAR provide for three levels of authorization including General Binding Rules (GBRs) covering "low risk" activities, Registrations controlling small scale activities, and site-specific Water Use Licences.

1.3 Agri-environmental policy and environmental protection

1.3.1 Common Agricultural Policy

In June 2003, the Common Agricultural Policy (CAP) was modified to better integrate environmental aspects. Single Farm Payments (SFP) were introduced to reduce the link between support and production ("decoupling"), to help farmers be less intensive, more competitive and obtain a more stable income (EC, 2004). SFP are linked to the respect of environmental, food safety and animal welfare standards, and imply keeping all farmland in good agricultural and environmental condition (GAEC) ("cross-compliance").

1.3.2 Land Management Contracts

The CAP reform took effect in Scotland in January 2005 and was accompanied by the creation of the Land Management Contracts (LMCs) (Scottish Executive, 2005b). LMCs are intended to bring social, economic and environmental benefits and comprise three Tiers: Tier 1 is the SFP and Cross Compliance, Tier 2 is the Menu Scheme and Tier 3 replaces the Rural Stewardship Scheme and the Organic Aid Scheme. LMCs may support the adoption of measures aiming at preventing soil erosion and compaction and water pollution, e.g. buffer zones, nutrient planning or constructed wetlands.

1.3.3 Scotland Rural Development Programme 2007-2013

The Scotland Rural Development Programme (SRDP) 2007-2013 was adopted in 2007 (Scottish Government, 2008a). Axis 2 is the principal means for supporting the outcomes on "enhanced biodiversity and landscape, improved water quality and tackling climate change", and sets out six priorities: biodiversity conservation, preservation and development of high nature value farming and forestry systems, protection of traditional agricultural landscapes, improvement of water and soil quality, mitigation of climate change and animal health and welfare. Farm wetlands are briefly mentioned (p. 65) as a tool to treat low-level contaminated water, and grant aid has been made available to support their construction since 2009.

1.4 Legislation and farming practices addressing water pollution in Scotland

1.4.1 Silage Slurry and Agricultural Fuel Oil (Scotland) Regulations 2003

The Silage Slurry and Agricultural Fuel Oil (Scotland) Regulations 2003 (SSAFO) urge farmers to collect, store and dispose of contaminated farmyard and roof (for poultry and pig farms) runoff. Runoff may be stored in tanks or slurry stores and spread on land by tanker or sprinkler system when weather conditions are suitable. However, often for technical and financial reasons (e.g. limited storage capacity, cost of roofing, guttering or modifying the drainage system), runoff is allowed to drain freely to surface water, polluting nearby watercourses (Alan Frost, *pers. comm.*). The SSAFO was recently amended and a General Binding Rule now allows "dirty water" runoff from defined areas of the farmyard to drain to a constructed wetland for treatment, reducing storage need and water pollution (Scottish Government, 2008b).

1.4.2 Good practices and best management practices

Multiple practices to attenuate pollution are listed in the "Prevention of Environmental Pollution From Agricultural Activity" (PEPFAA) (Scottish Executive, 2005a) and in the "4 Point Plan" (SEERAD *et al.*, 2004). Some are enforced by law, some are a requirement for receipt of the Single Farm Payment ("Good Farming Practices"), but many are voluntary measures and actions. Table 1.2 summarizes the most common measures which can be implemented at the farm scale to reduce diffuse pollution from the field and from the farmyard (adapted from Hilton, 2003; Vinten *et al.*, 2004; CRAPL, 2007; Cuttle *et al.*, 2007; SEPA, 2009b).

Table 1.2 On-farm measures for remediation of diffuse pollution from fields and farmyards with their place in the source-mobilisation-delivery continuum.

Field pollution remediation measures	Farmyard pollution remediation measures
Control of the source and mobilisation of pollutants	
Nutrient budgeting to avoid build-up of excess nutrients in the soil.	Animal diet improvement to reduce nutrient losses.
Reasonable pesticide use (small quantities, split up applications).	Diversion of the clean roof rainwater to drains to reduce the volume of dirty water generated.
Manure/slurry application plan: suitable application timing and amount to limit surface runoff and infiltration.	Maintenance or upgrading of buildings, manure/slurry, fuel or pesticide storages to avoid leaks and spillages.
Management of stocking density and grazing.	Use of woodchip corrals for housed over-wintering of livestock, if effluent is collected and stored appropriately before spreading.
Fencing water margins and bridging to prevent livestock access to watercourses.	Biobeds for pesticide washwater treatment.
Relocation of livestock feeders, water troughs and access tracks away from water bodies.	Collection tanks for pesticides and sediment from machinery washings.
Use of conservation tillage, grass cover on runoff-carrying depressions and on tramlines and crop residue mulches.	Roofing of silage pits and areas of farmyard where excrements are expected to accumulate.
Field drainage maintenance.	Reduction of the impermeable surface area in the farm.
Irrigation scheduling.	Contingency plans for spillages.
Control of the delivery of pollutants	
Swales and ditches to convey, store and treat dirty runoff.	Swales and ditches to convey, store and treat dirty farmyard runoff.
River restoration to improve flow and habitat.	Surface or subsurface flow constructed wetlands to treat farmyard runoff.
Buffer strips or retention ponds and wetlands to stop contaminants before they reach watercourses.	Farmyard runoff spreading or irrigation over grassland, crops, forested areas, orchards, etc.

1.5 Constructed wetlands for farmyard runoff treatment

Constructed wetlands (CWs) and ponds are man-made systems designed to create poorly drained soils and to allow for wetland vegetation, fauna, soils and microorganisms to develop and interact for the primary purpose of pollutant removal from wastewater, runoff (e.g. field, road, farmyard) or sewage (Hammer, 1992; USDA *et al.*, 1995a, b; US EPA, 2000). They have been used worldwide for many years for the treatment of municipal and agricultural effluents, from point or diffuse sources, with rather positive but variable results. They can play an important role at farm and catchment scale in the collection and treatment of contaminated farmyard runoff at the end of a treatment train, i.e. a set of measures whose aims range from pollution source control to dirty water collection and treatment.

Indeed, CWs may improve water quality by promoting uptake, transformation and inactivation of nutrients, metals and pathogens by microorganisms and plants, filtration, adsorption and chemical precipitation by contact with plants, substrate and litter, settling of suspended solids, chemical transformation (e.g. nitrification, denitrification), predation and natural die-off of pathogens. Treatment efficiency differs for different pollutants and varies considerably spatially and temporally, depending predominantly on design and age of the system, loadings, climate and maintenance (Reddy and DeBusk, 1987; Daukas *et al.*, 1989; Tanner *et al.*, 1995; USDA *et al.*, 1995b; Kadlec and Knight, 1996; Simeral, 1998; US EPA, 2000; IWA, 2000, Hunt and Poach, 2001; Senzia *et al*. 2003, Shilton, 2006).

Although the processes involved in water treatment (e.g. nitrification, denitrification, P sorption) within CWs are under scrutiny, the extent to which they improve water quality, their optimal design, long-term efficiency, and maintenance requirements are not fully understood (Brix, 1993 and 1994; Kadlec, 1994 and 1999), nor is their contribution to ammonia (from the transformation of NH_4 at high water pH) and greenhouse gas emissions (Johansson *et al.*, 2004; Fey *et al*. 1999; Poach *et al.*, 2004a; Strom and Lamppa, 2006), and their real cost-effectiveness and practicality (Ashford and Horsefield, 2005; Turpin *et al*. 2005).

More recently, concern has been growing with regard to the risk of infiltration and contamination of groundwater within unlined systems, whose number is increasing, due to their affordability and ease of management, which makes investigation in this field necessary. Additionally, until now, research in the UK has mainly focused on the ecology of Sustainable Urban Drainage Systems (SUDS) (Lancaster *et al.*, 2004; Culhane, 2007) or rural ponds impacted by lightly contaminated agricultural runoff (Williams *et al.*, 2003; Gee *et al.*, 1994), and only a few studies have assessed the ecological value (e.g. diversity of aquatic macroinvertebrates, amphibians and plants) of CFWs receiving yard runoff, e.g. Harrington *et al.* (2005) in Ireland and Coletto (2008) in Scotland.

However, long-term exposure of wildlife to high concentrations of pollutants might be a critical issue, causing bioaccumulation (accumulation of a toxic element within an organism) and biomagnification (accumulation along the food chain) (Camargo and Ward, 1995; Thiere and Schulz, 2004; Camargo *et al*., 2005), and justifies therefore the need for the long-term monitoring of CFWs.

The lack of understanding and the scope for improvement of the design of CFWs make it necessary to explain further the processes involved in water treatment, to quantify treatment efficiency and identify the factors influencing efficiency. More information is needed on changes of performance over time in response to loadings (Cronk, 1996; Cole, 1998), climate and ecosystem changes, and on the possible prediction and mitigation of these changes. Moreover, assessing the real ecological value and potential toxicity of CFWs is necessary to avoid endangering wildlife.

Finally, assessing carefully the cost-effectiveness of CFWs, the way they are perceived by farmers and the obstacles which hinder their implementation is essential if those systems or others are to be further promoted. Research should help orient practical decisions with regard to wetland design, in such a way that compromises are found between farm constraints, water treatment efficiency, biodiversity conservation, technology, aesthetics and costs. This should allow further development of existing design guidance (e.g. Carty et *al*., 2008a, b).

1.6 Research objectives and hypotheses

Detailed monitoring of water quality and pond ecology was carried out at two constructed farm wetlands designed to collect and treat farmyard (or "steading") runoff and field drainage, and whose construction in 2004 and 2005 was supported by the Scottish Environment Protection Agency's (SEPA) Diffuse Pollution Initiative and the Scottish Executive Environment and Rural Affairs Department (SEERAD).

1.6.1 Research objectives

The specific objectives of this study were to:

1) Study the relationship between climate, farm characteristics and practices, and the quality and quantity of wastewater entering the constructed wetlands.

2) Assess the water treatment performance of the CFWs and identify the factors influencing efficiency, which may be related to wetland characteristics and management (e.g. size, age, location in landscape, residence time), to the physical environment (e.g. rainfall, temperature), and to farm characteristics (e.g. pollutant and hydraulic loadings).

3) Assess the ecological value (e.g. habitat creation, biodiversity) of CFWs and the factors influencing this value (e.g. pollutant load, system size, proximity of existing wetlands).

4) Evaluate the costs (capital and running costs) and benefits (e.g. water quality improvement, amenity) of CFWs and their acceptance by famers, and compare them with other methods for farmyard water disposal (e.g. storage in slurry stores and spreading) or treatment (e.g. subsurface flow constructed wetlands).

5) Make sustainable recommendations for constructed wetland design, construction, maintenance and monitoring in order to optimize water treatment performance, ecological value and cost-effectiveness, and reduce negative side-effects, under local constraints.

1.6.2 Research hypotheses

This research aimed to test the following hypotheses:

1) Performance varies strongly from one CFW to the other and fluctuates over time for a given CFW because it is driven by a multiplicity of controllable as well as uncontrollable factors:

   1.1 Water treatment performance varies between CFWs due to the influence of hydraulic and pollutant loadings (volume and concentration of the influent) and design (influences hydraulic residence time and treatment processes).

   1.2 The efficiency of a given system decreases in winter when effluent volumes and pollutant loadings are expected to be higher and temperatures are lower: the CFW may become a source rather than a sink of pollutants.

   1.3 The long-term performance of a CFW (especially for phosphorus removal) decreases with increasing age of the system: flow patterns change and pollutants accumulate or are flushed out. Sediment dredging is a maintenance operation required to increase pond performance.

2) CFWs may enhance biodiversity by offering habitat, but may have detrimental impacts due to toxicity of water and sediment to invertebrates, amphibians and fish or invasion of alien species. These impacts are correlated with the water quality gradient, and the highest aquatic animal and plant diversity is expected to be found close to the outlet. Biodiversity is therefore enhanced in large multi-cell systems with improved water quality.

3) CFWs, if properly designed, are effective, but involve a significant investment for farmers. Their adoption is strongly conditioned by land availability and suitability and availability of financial and technical support for construction and maintenance.

1.6.3 Structure of the Thesis

The Thesis is organised in eight chapters. Chapter 1 introduces the issue of agricultural water diffuse pollution, the potential role of constructed wetlands for water treatment and presents the aims of the research. Chapter 2 gives a general background on the typology, characteristics and efficiency of ponds and wetlands used for remediation of agricultural runoff. Chapter 3 presents the two farms and constructed farm wetlands investigated.

Chapters 4, 5 and 6 are written in a paper format (although more detailed), and include the methods, results and discussions associated with each one of the aspects (i.e. water quality, ecology, costs-benefits) investigated at the two CFWs. Chapter 4 presents the assessment of the water balance and treatment efficiency of the CFWs, Chapter 5 focuses on the assessment of their ecological value, and Chapter 6 details the evaluation of their cost-effectiveness and perception by farmers.

Chapter 7 acts as a short overall discussion combining the findings of the different aspects investigated, underlining practical implications and making suggestions for sustainable design and maintenance of CFWs.

Finally, Chapter 8 concludes briefly the Thesis by summarizing the major findings and proposing a few key recommendations regarding future research in the field of constructed wetlands.

Chapter 2: Ponds and Constructed Wetlands for Wastewater Treatment

The chapter underlines the importance of the ecosystem services that natural wetlands provide and emphasizes the potential for constructed wetlands (CWs) to recreate partly those services. It describes the typology and hydrology of CWs, their advantages and limitations, the processes involved in water treatment, and focuses on the design and performance of surface flow CWs used in rural contexts with specific reference to Constructed Farm Wetlands and Integrated Constructed Wetlands.

2.1 Introduction: wetlands and ecosystem services

The dramatic loss of over 50% of the world's wetlands during the 20th century triggered a late response from the International Community but their preservation is nowadays a priority (UNESCO, 1994; Moser *et al.*, 1996; OECD and IUCN, 1996; Ramsar Convention on Wetlands, 2009). Wetlands cover c. 6% of the world's land surface, provide numerous goods and ecosystem services and have a high economic value. They contribute to flood attenuation, erosion control, water purification, groundwater recharge and carbon sequestration, they enhance habitat, biodiversity, genetic diversity and ecosystems stability, are used for educational and scientific purposes, and provide humans with food, fuel, fibre, medicines and water (Costanza *et al.*, 1997; Mitsch and Gosselink, 2000; Otte, 2003; Schuyt and Brander, 2004).

In Europe, rising concern about the extent and impacts of extensive drainage of traditional agricultural landscapes has led to changes in agri-environmental policy (e.g. the introduction of single farm payments and cross-compliance). The WFD (2000) also includes guidance on wetlands and several restoration projects are under implementation in the UK (Wetland Vision, 2008).

Natural ponds and wetlands have been used for centuries to dispose of wastewater, but it is only in the 20th century that they started to be created for the primary purpose of wastewater collection and treatment.

There are nowadays thousands of these systems all over the world used as an alternative to conventional systems, to collect and treat effluents from households, communities, holiday camps, mining activities, farms, meat or vegetable industries as well as road or field runoff. They aim at removing nitrate, ammonium, organic matter, sediment, pesticides, pathogens and metals from those effluents. They are often also used to enhance habitat for wildlife, help with land reclamation (e.g. after mining) or improve the amenity or landscape value of a site (Hammer, 1989 and 1992; Kadlec and Knight, 1996; IWA, 2000; Shilton and Walmsley, 2006). Constructed wetlands can, to a certain extent, play a role in the restoration of lost ecosystem services.

2.2 Typology of constructed ponds and wetlands

2.2.1 Typology of ponds

The term "pond" or "lagoon" refers to a man-made body of open water of variable size and depth, holding water for whole or part of the year, with a variable vegetation cover and variable infiltration patterns. Ponds used for treatment are often referred to as "waste stabilization ponds" (Mara *et al.*, 1992; Shilton, 2006) and classified into different categories (e.g. infiltration, maturation, facultative, anaerobic ponds), according to their properties (e.g. depth, infiltration, treatment processes). They are often used in series to ensure better hydraulic efficiency and improve treatment (IWA, 2000; Shilton and Walmsley, 2006).

Anaerobic ponds are rather deep (a few meters), typically have short residence times and allow settling and a significant reduction of the organic load, and are usually designed on a volume basis. Facultative ponds are relatively shallow (< 1.5 m), comprise aerobic and anaerobic zones, promote the development of algae and interaction with bacteria, and are designed on an area basis, since they are mainly driven by solar energy. Maturation ponds are shallow, have lower organic loadings and are subsequently better oxygenated. They are used to ensure effluent polishing and pathogens removal (Mara *et al*., 1992; Davies-Colley, 2006).

2.2.2 Typology of constructed wetlands

A constructed wetland is a more or less engineered system, designed to enhance the interaction between vegetation, fauna, soils and microorganisms for the primary purpose of pollutant removal from agricultural wastewaters (e.g. parlour washings), runoff (e.g. field, road, farmyard) or sewage (Hammer, 1992; USDA *et al.*, 1995a, b; US EPA, 2000; CRAPL, 2007; Carty *et al.*, 2008a). The term CW may refer to subsurface flow (SSF) wetlands or surface flow (SF) wetlands. In subsurface flow wetlands, water flows vertically or/and horizontally (Figure 2.1) through a porous substrate (e.g. gravel, sand) planted with macrophytes.

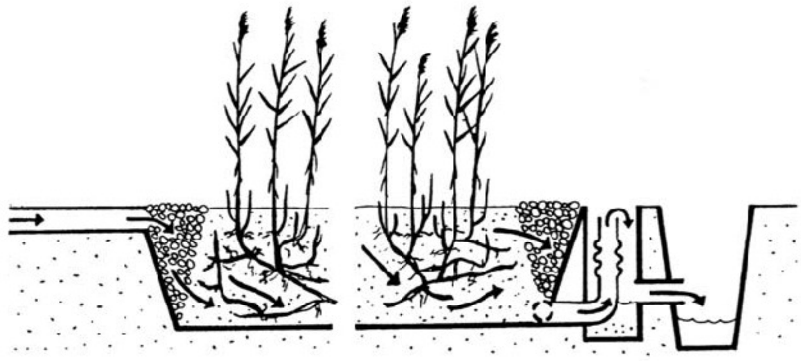


Figure 2.1 Longitudinal section of a subsurface flow wetland (Brix, 1993).

In surface flow wetlands (Figure 2.2), which may comprise several cells of variable depth and characteristics, water flows from the inlet to the outlet over the substrate (depth c. 30-40 cm) through the vegetation, made of submerged, emergent or floating-leaved plants.

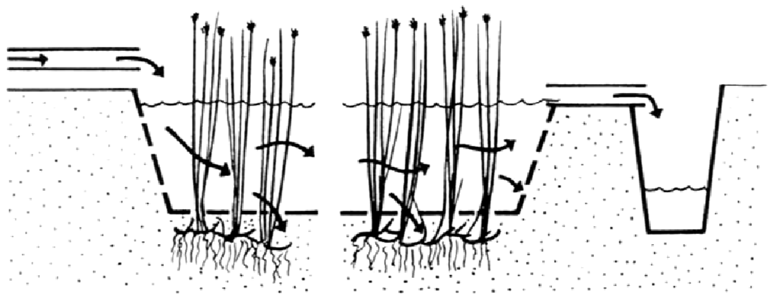


Figure 2.2 Longitudinal section of a surface flow wetland (Brix, 1993).

The present research focuses on surface flow wetlands, which are sometimes referred to as reedbeds, Integrated Constructed Wetlands (ICWs, Ireland) or Constructed Farm Wetlands (CFWs, Scotland). The term CFW has been recently defined as "one or more shallow, free surface flow constructed cells containing emergent vegetation, which is designed to receive and treat lightly contaminated surface water runoff from farm steadings, in such a manner that any discharge from the wetland will not pollute the water environment" (Carty *et al.*, 2008b; Scottish Government, 2008b). CFWs can be used for treatment of runoff from livestock handling areas (if livestock is held occasionally for < 24 h), roof drainage from pig and poultry housing, lightly contaminated concrete areas as a result of vehicle and occasional livestock movements, machinery washings (unless contaminated with pesticides or veterinary medicines), and winter run-off from silage pits and baled silage. They are not designed for the treatment of more nutrient rich effluent types such as slurries, silage effluent and raw milk, or the treatment of run-off containing veterinary medicines such as sheep dip, or pesticides, such as sprayer or dipping equipment washings.

2.3 Advantages and limitations of constructed wetlands

2.3.1 Advantages of constructed wetlands

Constructed wetlands have several advantages if properly designed (Hammer, 1992; USDA *et al.*, 1995a, b; Cooper *et al.*, 1996; Cronk, 1996; Kadlec and Knight, 1996; IWA, 2000; Braskerud, 2002a, b; Mason, 2002; Poe *et al*., 2003; Carty *et al*., 2008a, b): they can provide high and consistent level of treatment for nutrients, pathogens and hydrocarbons, contribute to runoff and flood management if built large enough, act as long-term carbon stores, are easy to manage, require little maintenance and energy use and are cheaper than alternative methods for farm runoff disposal. They minimize odours produced by agricultural wastes, due to their dense plant cover and shallow surface flow, are aesthetically pleasing if designed in a sensible manner, bring additional value to farmland and enhance habitat and biodiversity. They can be used as contingency measures against accidental spillages, for irrigation if large enough and they reduce the need for dirty water storage, decrease land area needed for application and allow better timing of land spreading.

2.3.2 Limitations of constructed wetlands

Constructed wetlands have some limitations: their construction requires relatively large areas in comparison with conventional treatment systems and they can be costly, their performance is not consistent during the year (lower in winter) and in the long-term and may be reduced when pollutants enter rapidly and in large amounts, and they require a minimum of water to maintain ecosystem function (USDA *et al.*, 1995b; Kadlec, 1999; US EPA, 2000).

Moreover, the creation and mismanagement of wetlands may alter existing wetlands or local hydrology, e.g. creating a pathway between the farm and waterbody where it was previously inexistent, can introduce invasive species, disrupt and intoxicate plant and animal communities. CWs may also contribute to "pollution swapping" causing emissions of greenhouse gases or/and groundwater degradation by infiltration, may cause health and safety hazards due to the presence of water and pathogens (Verhoeven *et al.*, 1990; SEPA and Pond Action, 2000; US EPA, 2000; Bruyère and Questel, 2001; Johansson *et al.*, 2004; Van de Weg *et al.*, 2008). Additionally, since CFWs are only an "end of pipe" measure, promoting them as an ideal tool to deal with diffuse pollution may increase the risk that farmers will put less efforts in controlling the sources of pollutants using other on-farm BMPs.

Groundwater contamination by infiltration through CFW substrate is a critical issue but recent detailed studies on this topic are limited. Experiments are being carried out in Ireland to assess infiltration volume and quality under ICWs (Rory Harrington, *pers. comm.*). In theory, CFWs built on clay soils and above the water table, show low infiltration and therefore present a low risk for groundwater. While some studies in the USA have shown that the permeability of lagoons receiving animal waste may decrease naturally over time due to sealing by accumulation of organic matter and sediment (Ritter, 1983; Miller *et al.*, 1985; Rowsell *et al.*, 1985; Bouwer *et al.*, 2001), others found that significant seepage and groundwater contamination can occur in more permeable soils (Korom and Jeppson, 1994; Westerman *et al.*, 1995).

2.4 Pond and constructed wetland hydrology

Quantifying the hydraulics of ponds and wetlands is crucial to understand their function and assess their efficiency in removing pollutants. Indeed, hydrology influences plant species composition (Bunce *et al*., 1999), soil characteristics and nutrient cycles (Kadlec and Knight, 1996; DeBusk, 1999a, b). The flow and storage volume determine the time water spends in the system as well as the degree of mixing, influencing the interactions between pollutants and wetland ecosystem and consequently pollutant removal and gas emissions.

2.4.1 Main terminology

The hydraulic loading rate (HLR, m d^{-1}) is expressed as:

(Eq. 2.1) $$HLR = \frac{Q_i}{A}$$

Where: Q_i: wastewater inflow (m^3 d^{-1}), A: wetland top surface area (m^2).

The pollutant loading rate at inlet (LR_i, kg m^{-2} d^{-1}) is defined as:

(Eq. 2.2) $$LR_i = HLR \times C_i$$

Where: C_i: inlet concentration (kg m^{-3}).

The real hydraulic residence time (RT, d), i.e. the time water spends in the wetland, can be assessed from tracer studies (Persson and Wittgren, 2003), but RT is usually expressed in a simpler form assuming complete mixing within the wetland, as:

(Eq. 2.3) $$RT = \frac{V}{Q_o}$$

Where: V: mean volume of water in the wetland (m^3), Q_o: mean outflow (m^3 d^{-1}).

2.4.2 Water budget

The simplified water budget for a wetland may be written as (IWA, 2000):

(Eq. 2.4) $$Q_i - Q_o + Q_c - Q_b + Q_{sm} + (P - ET - I) \times A = \frac{dV}{dt}$$

Where: Q_i: inflow (m^3 d^{-1}), Q_o: outflow (m^3 d^{-1}), Q_c: catchment runoff (m^3 d^{-1}), Q_b: bank loss (m^3 d^{-1}), Q_{sm}: snowmelt (m^3 d^{-1}), P: precipitation (m d^{-1}), ET: evapotranspiration (m d^{-1}), I: infiltration to groundwater (m d^{-1}), A: area of the wetland (m^2), t: time (day), V: water storage in the wetland (m^3).

2.4.3 Wetland background concentrations

Natural wetlands are characterised by a net surplus of fixed carbonaceous material as they are more autotrophic than heterotrophic and they release dissolved or particulate elements which stay in the water column, get buried in sediment or are flushed out (Mitsch and Gosselink, 2000). The concentrations released under these conditions vary between systems and are called "background concentrations". The IWA (2000) gives the following ranges: BOD_5: 1-10 mg l^{-1}, suspended solids: 1-6 mg l^{-1}, organic and total nitrogen: 1-3 mg l^{-1}, ammonium-nitrogen: $\leq$ 0.5 mg l^{-1}, nitrate-nitrogen: $\leq$ 0.1 mg l^{-1}, total phosphorus: $\leq$ 0.1 mg l^{-1}, faecal coliforms: 50-500 cfu 100 ml^{-1}.

2.5 Processes involved in water treatment and fate of pollutants

Figure 2.3 illustrates the main elements and processes contributing to water quality improvement in surface flow wetlands (Wallace and Knight, 2006).

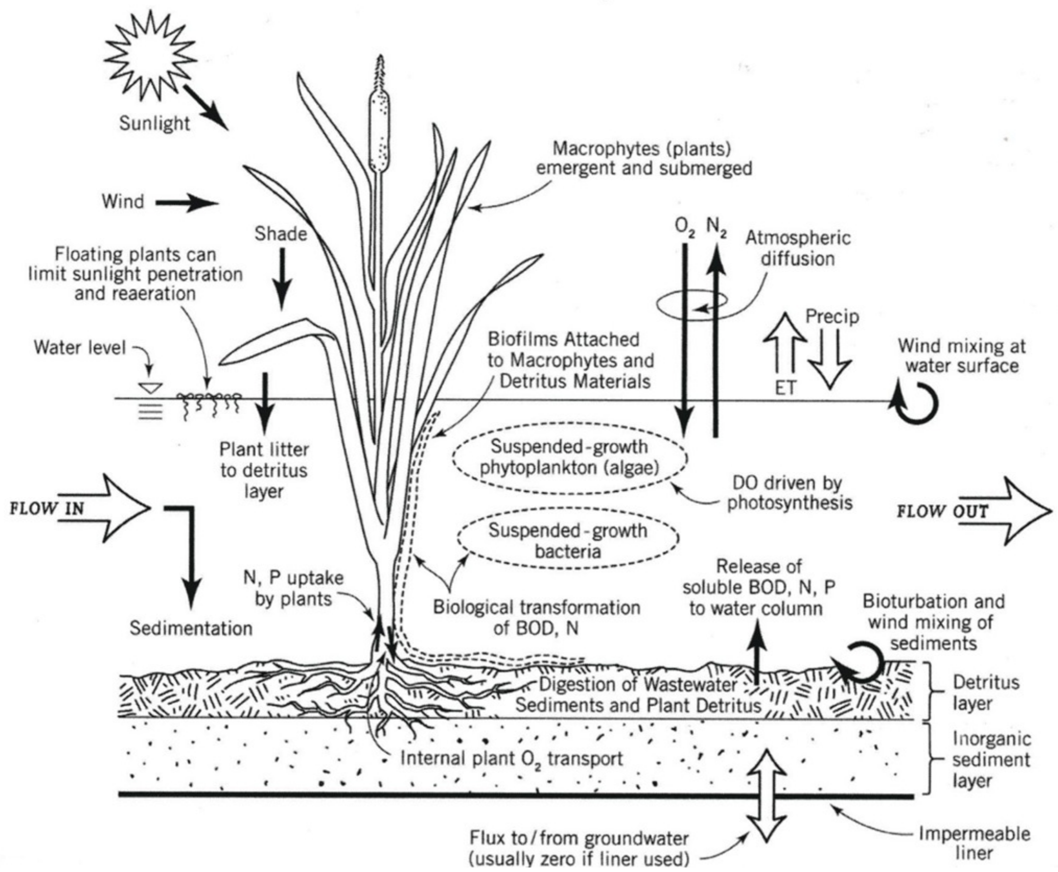


Figure 2.3 Major water treatment processes in surface flow wetlands (Wallace and Knight, 2006).

Organic Matter and Biochemical Oxygen Demand (BOD) are removed in wetlands by microbial degradation and breakdown. As removal efficiency is directly linked to microbial activity, it is mainly influenced by temperature and pH (Tanner *et al.*, 1995; Vymazal, 1999; Karathanasis *et al.*, 2003).

Several studies about the fate of N in wetlands were reviewed (Reddy and DeBusk, 1987; Brix, 1993; Kadlec and Knight, 1996; Soto *et al.*, 1999; Koskiaho *et al.*, 2003; Senzia *et al.*, 2003; Scholz and Lee, 2005). They indicate that nitrogen (N) enters wetlands in organic (dissolved or particulate) or inorganic forms and may also originate from N deposition or N fixation by anaerobic or aerobic bacteria or cyanobacteria containing the nitrogenase enzyme. Figure 2.4, adapted from DeBusk (1999), illustrates the nitrogen cycle in wetlands.

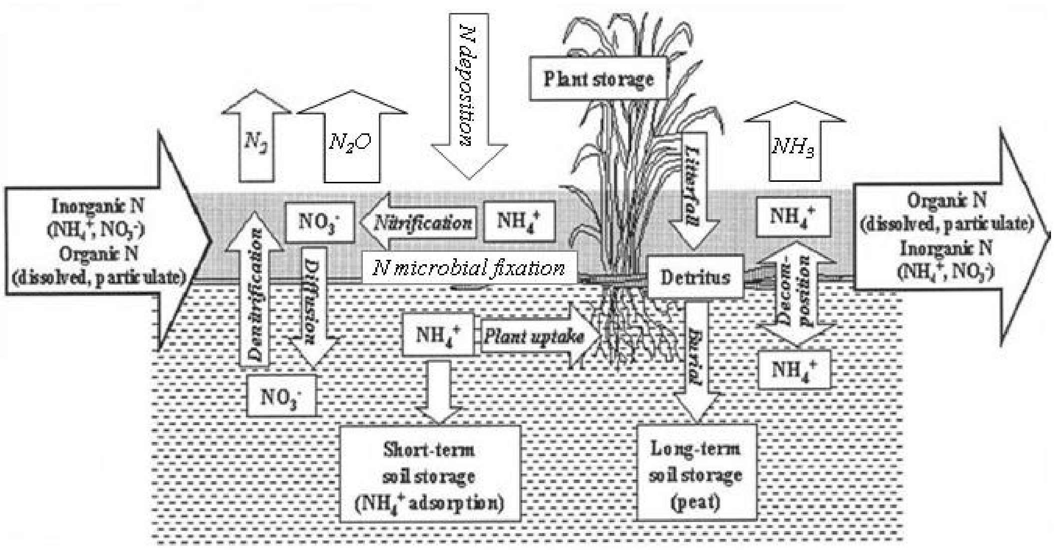


Figure 2.4 Fate of nitrogen in wetlands (adapted from DeBusk, 1999a).

Organically bound N is transformed through ammonification into ammonium (NH_4^+) under aerobic or anaerobic conditions. NH_4^+ may be taken up by plants or micro-organisms, stored in anaerobic reduced layers of sediment, oxidized to NH_3 and lost by volatilization, oxidized to nitrate (NO_3^-) through nitrification, or oxidised into N_2 under anaerobic conditions using hydroxylamine derived from nitrite (NO_2^-) as oxidant, following the "anaerobic ammonia oxidation" (ANAMMOX) process, described by the equation $NH_4^+ + NO_2^- \rightarrow N_2 + 2\ H_2O$ (Mulder *et al.*, 1995; Hunt and Poach, 2001).

Nitrification occurs under aerobic conditions, within surface water or in the aerobic sediment layer or oxidized rhizosphere of wetland plants, with the following reaction, described in two steps involving chemoautotroph bacteria using N oxidation as energy source and CO_2 as carbon source: 1) $2\ NH_4^+ + 3\ O_2 \rightarrow 2\ NO_2^- + 2\ H_2O + 4\ H^+$ + energy, involving *Nitrosomonas;* 2) $2\ NO_2^- + O_2 \rightarrow 2\ NO_3^-$ + energy, involving *Nitrobacter.* Nitrification is influenced by pH, dissolved oxygen, temperature and alkalinity. Nitrate may be taken up by plants, leached out of the system or be reduced to nitrous oxide (N_2O) or nitrogen gas (N_2) through denitrification.

Denitrification is carried out in anaerobic zones by bacteria such as *Pseudomonas* spp., following: $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$. It is a major pathway for nitrogen removal in wetlands and is mainly influenced by the available organic matter content, redox potential, dissolved oxygen (high DO impedes denitrification), temperature and pH. Nitrate may also be transformed into N_2O and N_2 through aerobic abiotic process, e.g. chemodenitrification.

Figure 2.5 illustrates the phosphorus cycle and its fate in wetlands (Kadlec and Knight, 1996; Kadlec, 1999; Reddy *et al.*, 1999; Richardson and Qian, 1999; Koskiaho *et al.*, 2003; Dierberg *et al.*, 2005; Scholz and Lee, 2005).

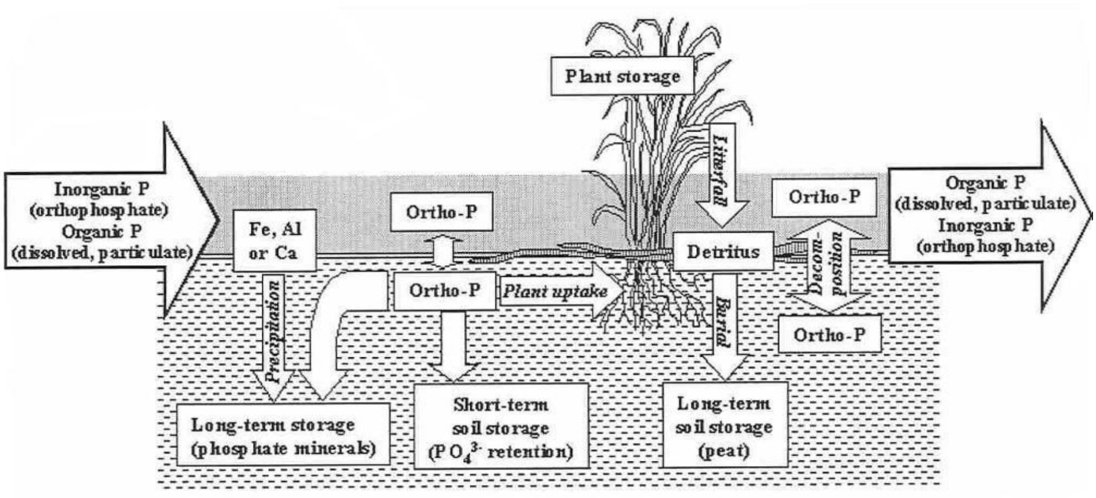


Figure 2.5 Fate of phosphorus in wetlands (DeBusk, 1999b).

Phosphorus (P) enters CWs as soluble or insoluble, organic or inorganic complexes. It originates mainly from the use of mineral fertilisers and from manures or slurries washed off land/soil during rainy periods. P may be adsorbed onto suspended solids (e.g. clay particles, organic matter), fixed as aluminium or iron phosphates (especially under acidic conditions) or bound by calcium and magnesium (in more alkaline conditions). It is retained in the wetland mainly through physical sedimentation of the particles to which it is bound. P binding or solubilisation is influenced by pH, temperature, redox potential, interstitial soluble P and microbial activity. Soluble P may be taken up by plants, bacteria, and algae and incorporated in organic matter.

Faecal pathogens, including bacteria, enteric viruses and protozoa (e.g. *Cryptosporidium*, *Giardia intestinalis*) are removed mainly by natural or induced die-off (e.g. by UV radiation or toxins from other organisms), predation (e.g. protozoans feeding on bacteria), sedimentation, filtration and adsorption onto sediment and organic litter. Pathogen removal is mainly influenced by temperature, UV exposure, presence of vegetation, and is enhanced by a longer residence time (Gersberg *et al.*, 1989; Tanner *et al.*, 1995; Kadlec and Knight, 1996; Soto *et al.*, 1999; Perkins and Hunter, 2000; Vinten *et al.*, 2002; Karathanasis *et al.*, 2003; Karim *et al.*, 2004).

Suspended solids are removed from the water column by flocculation and physical sedimentation, and to a lesser extent, by filtration, adsorption and degradation by aquatic organisms, plants and bacteria. Sedimentation is relatively fast and enhanced by low flow velocity, low turbulence, presence of obstacles (e.g. vegetation) and long residence time (Hammer *et al.*, 1993; Raisin and Mitchell, 1995; Tanner *et al.*, 1995; Kadlec and Knight, 1996; Karathanasis *et al.*, 2003; Koskiaho *et al.*, 2003).

Pesticides being often bound to suspended solids, they are therefore removed mainly through sedimentation, adsorption to plants, sediment or organic litter. They also undergo bio-chemical transformations into less toxic compounds, influenced by temperature, pH and residence time (Kadlec and Knight, 1996; Rodgers and Dunn, 1992; Moore, 1999).

Heavy metals (e.g. from oil spillages) may be removed or stored by sedimentation, adsorption to plants and sediment, plant uptake, biological assimilation, decomposition, chemical transformation and volatilisation, these processes being mainly influenced by temperature, pH, redox potential and availability of adsorption sites (Sinicrope *et al.,* 1992; Eger, 1994; Crites *et al.*, 1997; Mitsch and Wise, 1998; Walker and Hurl, 2002).

2.6 Design considerations for surface flow constructed wetlands

2.6.1 General approaches and models

The design of CWs depends on the specific objectives of the farmers or small communities, on environmental agencies who set target concentrations, on the assimilative capacity of receiving water bodies, and on the resources (land, capital) available. Design criteria include the surface area, volume, length to width ratio, depth, number of cells, vegetation and hydraulic and pollutant loadings (USDA *et al.*, 1995a; US EPA, 2000). Due to the complexity of wetland ecosystems, no consensus exists on the "ideal" design of CWs. However, multi-cell systems with a combination of open water and fully vegetated areas with variable water depth are usually recommended to create anaerobic and aerobic zones. Well oxygenated open water enhances nitrification, while more anaerobic, densely vegetated areas promote denitrification, sediment settling and phosphorus retention (Kadlec and Knight, 1996; Braskerud, 2001; Kadlec, 2005; Thullen *et al.*, 2005).

The USDA-NRCS-US EPA (1995) wetland design approach for animal wastewater was mainly based on BOD_5 removal ("presumptive method"), and recommended a residence time of at least 12 days (Stone *et al.*, 2000). Two main other approaches have since been developed and are used worldwide to calculate the minimum area required for a treatment wetland to achieve a given outlet concentration, both based on first-order kinetics area-based models.

Reed *et al.* (1995) propose the following model to estimate the surface area of wetland required (A, m^2):

(Eq. 2.5) $$A = \frac{Q \times (\ln C_i - \ln C_o)}{K_T \times y \times n}$$

Where: Q: inflow (m^3 d^{-1}), C_i: inlet concentration, C_o: outlet concentration, K: first order constant (d^{-1}), y: water depth (m), n: porosity coefficient, T: temperature (^{0}C).

With the concentration of the effluent (C_o, mg l^{-1}) expressed as:

(Eq. 2.6) $$C_o = C_i \times e^{-K_T t}$$

Where: t: retention time (d).

Kadlec and Knight (1996) proposed a slightly different model (known as the k-C^* model) which does not account for porosity and depth, but incorporates background concentrations. The required wetland area (A, ha) is calculated as:

(Eq. 2.7) $$A = \frac{0.0365 \times Q}{K} \times \ln \frac{C_i - C^*}{C_o - C^*}$$

Where: Q: inflow (m^3 d^{-1}), K: first-order constant (m yr^{-1}), C_i: concentration at inlet (mg l^{-1}), C_o: concentration at outlet (mg l^{-1}), C^*: background concentration (mg l^{-1}).

The influence of temperature on performance is considered negligible, due to seasonality effect counteracting or hiding temperature effect, except for nitrogen which is strongly depending on bacterial activity and hence on temperature (Kadlec, 2003). CH2M Hill and Payne Engineering (1997) compared both models and found that the method proposed by Kadlec and Knight (1996) usually requires larger wetlands surface areas (Stone *et al*., 2000).

2.6.2 The treatment volume approach: experience from the UK

In the UK, the "treatment volume" approach, which is recommended for SUDS and based on hydraulic loadings, is also used for the design of CFWs. The treatment volume (V_t) is an estimate of the volume of runoff that contains the most polluted part of the runoff from a storm (Woods-Ballard *et al*., 2007). In the UK, the volumes of ponds, wetlands and swales are often expressed as multiples of *Vt* commonly ranging between 1 and 5 *Vt*. A large *Vt* multiplier is associated with a longer residence time and hence a higher level of treatment of the contaminated water.

Two methods are used to assess *Vt*, the Variable Rainfall Depth Equation (VRDE), and the half inch rule (HIR). The VRDE is based on the Flood Studies Report (NERC, 1975) and Wallingford Procedure (Wallingford and Institute of Hydrology, 1981), which contains maps showing the distribution of rainfall depths throughout the UK and soil indices based on the Winter Rain Acceptance Potential (WRAP). The VRDE is meant to ensure the capture of 90% of storm runoff and is as follows:

(Eq. 2.8) $$V_t = 9 \times \mathrm{D} \times \left\{\frac{\mathrm{SOIL}}{2} + (1 - \frac{\mathrm{SOIL}}{2}) \times \mathrm{I}\right\}$$

Where: V_t: Water quality treatment volume (m^3 ha^{-1}), SOIL: Soil Index (from Flood Studies or Wallingford Procedure WRAP map); I: Fraction of the catchment area which is impervious; D: M5-60 minute rainfall depth (i.e. 5-year return period, 60 min duration storm depth determined by the Wallingford Procedure).

However, it does not consider pollution loadings, target concentrations to be reached, nor account for changes in rainfall (frequency and intensity) resulting from climate change since it is based on rainfall data from before the early 1970s in the UK. The HIR is a simple but rather empirical method which suggests an average V_t of 11 to 15 mm of rainfall depth applied to the total impervious area of the catchment.

2.6.3 Semi-empirical approaches: experience from New Zealand and Ireland

The case of New Zealand

In New Zealand, the use of surface flow constructed wetlands is suggested after conventional waste stabilization ponds (one anaerobic followed by one facultative pond) to treat dairy farm waste water, i.e. mainly parlour washings. Tanner and Kloosterman (1997) proposed a design targeting BOD_5 removal and based on the number of cattle and subsequent wastewater volume produced, suggesting a wetland surface area of about 3.6 m^2 per head of cattle. However, wetland effluent concentrations are still high for this level of treatment, i.e. 25-50 mg l^{-1} for BOD_5, 40-70 mg l^{-1} for TN, 30-65 mg l^{-1} for NH_4-N and 16-30 mg l^{-1} for TP.

The case of Ireland: Integrated Constructed Wetlands

Integrated Constructed Wetlands (ICWs) appeared in the Republic of Ireland in the 1990s to improve water quality as part of a wider community based ecological project in the Anne Valley Catchment, in the area of Waterford (Carroll *et al.*, 2005; Harrington *et al.*, 2005). ICWs are large (from a few thousand m^2 to more than 1 ha), multi-cell (three or more), fully vegetated, shallow surface flow wetlands that are used to treat dairy parlour washings, farmyard runoff, seepage from silage silos or slated buildings, woodchip corrals, human sewage and kitchen water. The small depth and dense vegetation of the cells reduce excavation costs, risk of infiltration to groundwater, drowning hazards and odours. Several pipes can be used to spread the inflow, the distance between inlets and outlets is maximised and water level is maintained below 40 cm by the use of elbow pipes (Harrington *et al.*, 2005). ICW sizing is based on peak influent, related to precipitation and retention time during a storm flow situation (e.g. 100 mm of rainfall over 2 days). No straightforward relationship was found between treatment efficiency and ratio wetland surface area: farmyard area, but experience suggests that a ratio of 2:1 allows adequate water discharge and robust treatment (Scholz *et al.*, 2007a, b).

2.7 Treatment performance of ponds and constructed wetlands

2.7.1 Assessing water treatment efficiency

The performance of CWs has been assessed worldwide by comparing amounts of pollutants at the inlet and outlet of these systems. Results are heterogeneous due to the fact that the wetlands investigated are unique, showing wide ranges of wastewater strength, mass and hydraulic loadings, residence time, infiltration patterns, size, depth and vegetation characteristics. Moreover, wetland performance is variable both temporally and spatially, depending on the design and age of the system, quantity and quality of wastewater treated and on the climate (Kadlec and Knight, 1996). Performance may be determined using concentration or mass reduction efficiencies. However, removal percentage is useful only when correlated with influent characteristics, design and operating conditions (IWA, 2000).

2.7.2 Overview of the performance of constructed ponds and wetlands

Kadlec and Knight (1996) reported the average performance of 70 North American surface flow wetlands treating domestic or agricultural effluent (Table 2.1). Concentration and mass reductions are from the same order, higher for BOD_5, solids and NO_x-N or TN, and smaller for ortho-P and TP.

Table 2.1 Average performance by concentration and mass of 70 surface flow CWs treating domestic or agricultural effluent (Kadlec and Knight, 1996).

Water quality parameter	Concentration reduction (%)	Mass reduction (%)
BOD_5	74%	71%
TSS	70%	68%
NH_4-N	54%	38%
$NO_2 + NO_3$-N	61%	51%
TKN	43%	47%
TN	53%	55%
Ortho P	37%	41%
TP	57%	34%

CH2M HILL and Payne Engineering (1997) reported higher efficiencies for beef cattle wastewater treatment, and lower performance for poultry wastewater, mainly due to the higher strength of influent (Table 2.2), illustrating the link between performance and the origin and nature of the influent treated.

Table 2.2 Average pollutant concentration reduction in CWs treating agricultural wastewater (CH2M HILL and Payne Engineering, 1997).

Water quality parameter	Average concentration reduction (%)
BOD_5	25% (poultry) to 83% (beef cattle)
TSS	52% (swine) to 81% (beef cattle)
NH_4-N	20% (poultry) to 57% (beef cattle)
TN	22% (poultry) to 51% (dairy)

Table 2.3 summarizes treatment efficiencies by mass for a variety of surface flow wetlands used to treat agricultural runoff or wastewater. Negative efficiencies correspond to the influent concentration being smaller than outlet concentration, due to inlet concentration being low or to a release of pollutants at outlet.

Table 2.3 Performance of individual CWs treating agricultural wastewater (adapted from Heal *et al*., 2006a).

Reference and study information	% mass removal: mean (min – max values)				
	BOD_5	SS	NH_4-N NO_3-N*	TN	TP RP*
Braskerud (2002a, b): instream wetlands (arable and dairy runoff), Norway	-	-	(-10–3)	(3–15)	(21-44)
Dunne *et al.* (2005b): constructed wetland (dairy farmyard water), Ireland	(86–99)	(93–98)	(38–95)	-	(5–84)*[1]
Fink and Mitsch (2004): restored wetland (arable and forest runoff), USA	-	-	41*	-	28
Geary and Moore (1999): pond-wetland (dairy wastewater), Australia	61	-	26 (3.2–45)	-	28 (-8.6–57)
Raisin and Mitchell (1995): constructed wetland (livestock runoff), Australia	-	-	-	(–25–40)	(-15–45)
Reddy *et al.* (2001): experimental marsh-pond (swine wastewater), USA	-	66–69	43–60[3]	37–51	31–44
Thorén *et al.* (2004): pond-marsh (agricultural/urban runoff), Sweden	-	-	-	17 (6–36)[4]	-

- Data not available.
[1]SRP: 5% removal in winter; 81–84% removal rest of year. [3]Most removed in warmer months. [4]Annual removal, but 40% of annual N removal exported February to March 2001.

Despite high strength influents, Irish ICWs (Scholz *et al.*, 2007b) achieve very high concentration removal efficiencies for all pollutants, between 70% and 99% (Table 2.4), made possible by the large size of the wetlands and increased residence time.

Table 2.4 Concentration reduction efficiency of 13 ICWs treating farmyard runoff for selected water quality parameters (Scholz *et al.*, 2007b).

ICW No.	Farm and ICW characteristics				Concentration reduction efficiency (%)				
	Yard area (m^2)	Cow No.	ICW area (m^2)	ICW: yard area ratio	BOD_5	SS	NH_4^+	RP	EC
1	4500	60	3906	0.9	99	99	99	99	99
2	14 750	60	22 966	1.6	97	84	99	98	99
3	5400	50	10 288	1.9	95	83	98	81	99
4	9200	100	10 327	1.1	96	96	98	93	99
5	4000	35	3940	1	95	93	99	98	99
6	9800	80	12 691	1.3	91	90	99	99	99
8	2300	NA	3940	2	71	79	99	97	95
9	4800	55	7964	1.7	97	94	98	96	100
10	2100	50	4375	2.1	92	95	99	99	NA
11	5000	77	7676	1.5	98	93	99	92	100
12	13 600	85	10 748	0.8	97	91	99	99	NA
13	5000	NA	5610	1.1	56	90	99	93	NA

Overall, treatment efficiency of surface flow wetlands is higher for BOD_5 and suspended solids, while removal of nitrogen and phosphorus appears more limited, requiring significantly larger areas. Pathogen removal is usually very efficient, possibly higher in deep ponds and wetlands (García *et al.*, 2004). For example, 90-99% reduction was found in a constructed wetland in Wisconsin (Spangler *et al.*, 1976), and Hammer *et al.* (1993) found more than 98% removal for faecal coliforms and *streptococci* in a surface flow wetland treating livestock wastewater.

2.7.3 Factors influencing wetland performance

CW performance varies strongly spatially and temporally, and wetlands may act as sinks or sources of contaminants, depending on their age, location, design, wastewater characteristics, loadings, retention time, hydrological conditions, season, biological activity and management (IWA, 2000; Woltemade, 2000; Dunne *et al.*, 2005a; Scholz and Lee, 2005).

Performance is seasonally dependent: efficiency seems to be lower in winter, due to low temperatures inhibiting plant growth and microorganism activity, to vegetation decay, and higher rainfall reducing retention time and increasing the risk of nutrient flushing. Anaerobic conditions at the water-sediment interface, e.g. due to ice formation in winter or increased microbial activity in summer, might cause the release of P bound to sediment and make it susceptible to leaching or available to algae (Kadlec and Knight, 1996; Mason, 2002).

Vegetation in CWs (*Phragmites australis, Typha latifolia or Scirpus* spp.) has an overall positive impact on treatment efficiency: it stabilizes the surface of the wetland, reduces flow velocity and facilitates sedimentation, takes up nutrients from sediment and stores them in green parts or other organs (roots, tubers), adsorbs metals, provides fixation sites for microorganisms, conducts oxygen to sediment, produces aerobic conditions which enhance nitrification, and provides wildlife with habitat and food (Mitchell and Williams, 1982; Brix, 1994; IWA, 2000; Lambers and Colmer, 2005).

Plant nutrient uptake is not the major pathway for N and P removal but can contribute 16-75% removal of total nitrogen and 12-73% removal of total phosphorus (Reddy and DeBusk, 1987). An appropriate plant selection can improve wetland efficiency: plants should be native, perennial, highly productive for rapid nutrient uptake, produce rhizome or storage organs, and be tolerant to high pollutant loads and anaerobic conditions (Langergraber, 2004). However, dying plants and accumulation of debris might increase BOD, decrease dissolved oxygen or release nutrients and affect treatment performance (Langergraber, 2004). Vegetation removal can be a way to export nutrients from the wetland, but it is costly, time-consuming and may disturb wetland function and decrease efficiency (Mason, 2002).

The variability in the design, use and performance of CFWs, and the lack of detailed studies investigating simultaneously the hydrology, ecology and economics of individual systems justifies the necessity to explore the efficiency, limitations and sustainability of the particular design used in Scotland until now.

Chapter 3: Field Sites

In this chapter are presented the location, design, inputs and main ecological characteristics of the two constructed farm wetlands (CFWs) which were investigated between February 2006 and June 2008. The various methods used for the research (water monitoring, ecological monitoring and socio-economic surveys) are presented in the following chapters, together with results and discussions for each of the different aspects studied.

3.1 Study area and constructed wetland selection

To address the research hypotheses, constructed wetlands (CFW1 and CFW2) receiving farmyard runoff and field drainage were studied between February 2006 and June 2008 at two farms (F1 and F2) located in the catchment of the River Tweed (Solway Tweed River Basin District), about 60 km southeast from Edinburgh, in the Scottish Borders (Figure 3.1). Both CFWs were within a Nitrate Vulnerable Zone (NVZ Edinburgh, East Lothian and the Borders).

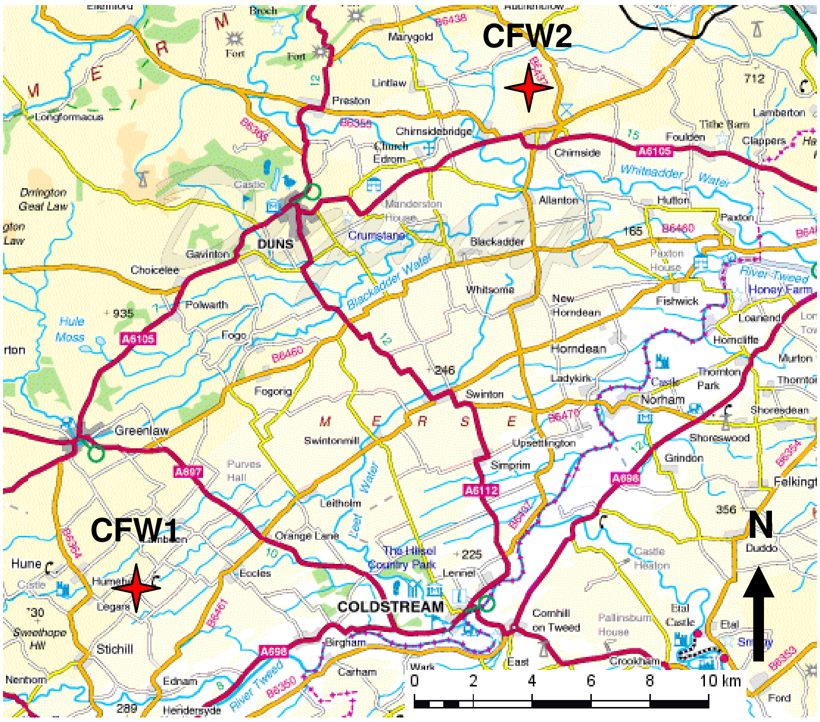


Figure 3.1 Location of the CFWs; South East Scotland (EDINA, Digimap).

The monitoring and protection of the River Tweed is a high priority due to its high ecological value and economic importance, in particular for salmon fishing which represents a significant local source of income and employment, i.e. a Gross Value Added of £7 M yr^{-1} and 487 direct jobs (Tweed Forum, 2007). Overall, water quality across the catchment is good according to SEPA's river classification scheme, but certain areas are locally nutrient enriched, e.g. the rivers Leet, Eden and Till (Tweed Forum, 2003). The Solway Tweed RBD is strongly impacted by agricultural diffuse pollution, which affects 923 km of rivers, 2 km^2 of loch, 322 km^2 of transitional water, 177 km^2 of coastal water and 5182 km^2 of groundwater (SEPA and EA, 2007). In order to address agricultural pollution in the Tweed Catchment, the construction of two CFWs was promoted in 2004 and 2005 by a SEPA-SEERAD initiative. Later on in 2007, the construction of eleven other farm wetlands was promoted by the Tweed Forum and FWAG (Farming and Wildlife Advisory Group) on a range of farms including dairy, beef, arable and mixed units (Tweed Forum, 2007).

The two CFWs were selected for this research for the following reasons: 1) They were part of a demonstration project fostered by SEPA: financial support was given to farmers and they agreed to give access to the public and researchers; 2) Built in 2004 and 2005, they were more than 6 months old at the start of the research; 3) They were located within two very contrasting farms (a large dairy and a mixed beef-arable farm), which allowed a comparison of the impacts of agricultural practices on pollutant loadings; 4) Their design was well documented and contrasting, which allowed the relationship between design and performance to be investigated.

3.2 Study sites description

3.2.1 Farm 1 (F1) and Constructed Farm Wetland 1 (CFW1)

F1 and CFW1 are located at NT 729412; 55°39' N, 2°25' W. The average annual rainfall in the area is 685 mm yr^{-1} (2000-2007 data from Lochton Land Station 5 km south east of CFW1, BADC), ranging from 424 to 843 mm yr^{-1}, with October being usually the wettest month. Minimum and maximum air temperatures ranged between - 8°C in winter and 21°C in summer for the study period (2006-2008).

F1 is a large dairy farm (total area of 320 ha; 184 ha arable, 91 ha rotational grassland, 24 ha permanent grassland) holding 450 dairy cows (Prim' Holstein), the average for Scotland being 106 cows per holding in 2006 (SEERAD, 2007).

Adult cows are housed on cubicles, milked twice a day (5-9 am, 4-8 pm), and graze outside in late summer only. Calves are housed on straw and are allowed to graze outside all year round. Fertilizers are applied between March and July over grassland and wheat. The slurry storage capacity is 5000 m^3 (2 tanks of 2500 m^3 each), and slurry is spread every three weeks, between 15th February and 10th September. The yard is scraped twice a day and faeces are directed to the slatted building, to be stored in the slurry tanks. Runoff from the farmyard is not collected in dirty water tanks, and before pond's construction, it was left to drain freely to the adjacent ditch.

CFW1 (Figure 3.2) was built in August 2005 in an improved pasture, at a cost of c. £4500, without the land cost. It is located c. 300 m downhill from the farm (c. 95 m asl), and 500 m from a 10-year old amenity pond and is fenced, the whole system occupying c. 0.4 ha (Figure 3.3).

Figure 3.2 View of CFW1 (summer 2007). The end of the inlet swale is visible in the foreground; the outlet is located on the opposite bank.

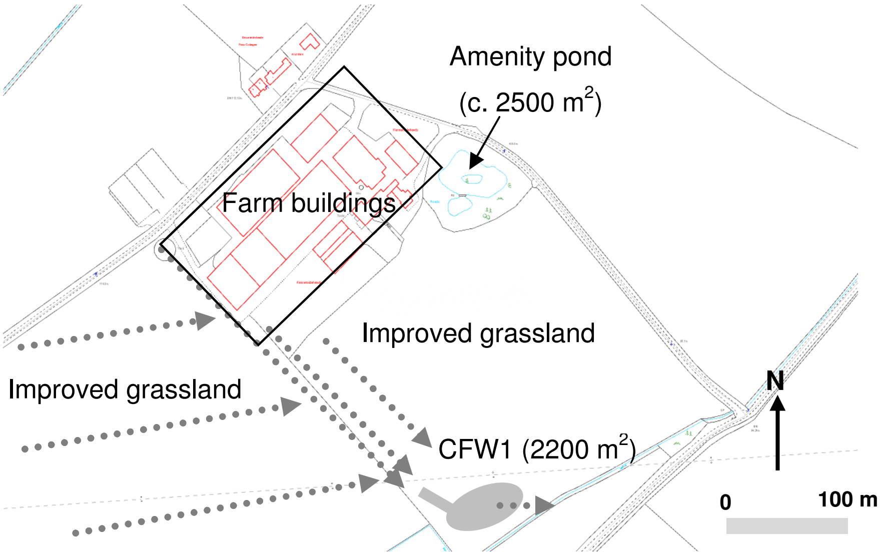


Figure 3.3 Location of CFW1 and farm buildings (shaded). The direction of farmyard runoff, field drainage, overland flow and wetland discharge is indicated by dashed arrows (EDINA, Digimap).

The pond has an area of 2200 m^2, a maximum depth of 1.5 m in the centre and a volume of 1500 m^3 (Figure 3.4). Water holding capacity relies solely on the clay substrate (no polyethylene liner was used). It is subdivided into two sections by a shallow bar of earth which is covered by water most of the time and was intended to be planted with reeds. The first section is supposed to act as a sediment trap.

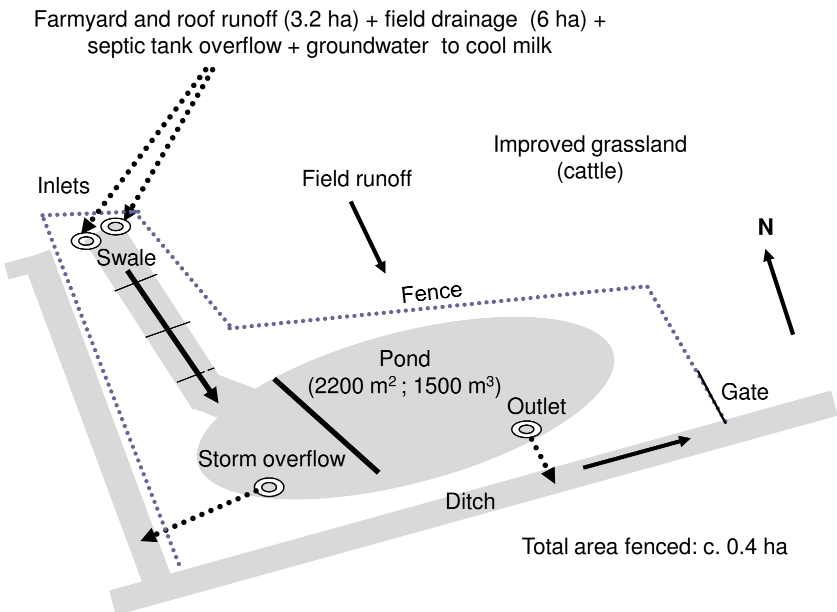


Figure 3.4 Sketch of CFW1 (not to scale). Dashed arrows represent flow in subsurface pipes and full arrows represent surface flow.

CFW1 was designed according to the treatment volume approach (see Section 2.6.2) to receive runoff from 3.22 ha of farmyard and roofs (of which 2.28 ha are impermeable), domestic septic tank overflow and silage pit runoff. The maximum volume to be treated was estimated to be 105 m^3 ha^{-1} (i.e. a V_t of 340 m^3), calculated with a 15 mm maximum rainfall in 60 min with a five-year return period, Whitsome soil series with runoff potential of 4 and Soil Index of 0.45. However, the actual input into CFW1 also includes the drainage from c. 6 ha of the adjacent pasture as well as significant volumes of groundwater pumped from a borehole (16-30 m^3 d^{-1}) used to cool the milk, and discharged in a surface water drain reaching the pond. Lateral runoff into the pond is expected to be limited due to its edges being raised and due to the presence of a field drain above it. CFW1 was built over soils of the Whitsome series, comprising clay loam topsoils and subsoils, with expected Cation Exchange Capacity (CEC) above 20 meq 100 g^{-1} soil. The subsoils are of low permeability and risk of infiltration is thus believed to be low. Nevertheless, in the lower-lying parts of the system close to the ditch, the soil between 500 and 800 mm depth is somewhat lighter (sandy loam) and is underlain by sandy clay loam subsoil.

Wastewater from the farmyard and field discharges into a swale (c. 25 m long) from two pipes (29 and 23 cm in diameter) over paving slabs to minimise erosion. It runs down the swale over a series of small weirs (wooden railway sleepers) and enters the pond. Water finally leaves CFW1 through a pipe (14.5 cm in diameter, 10 m long) located on the south-east corner of the pond and flows into a ditch. Under heavy rainfall conditions, pond water level may rise above the normal outlet and leave through a vertical "stormwater pipe" (30 cm in diameter) located in the first section, c. 20 m from the inlet, and connected to the ditch.

CFW1 was not planted initially and vegetation is still sparse after three years. Bulbs of *Iris pseudacorus* and rhizomes of *Typha latifolia* were transplanted from the existing 10 year-old pond by the farmhouse in 2005 and some established on the east edge of the pond. Since May 2006, *T. latifolia* is spreading and *Phragmites australis*, *Juncus effusus* and *Phalaris arundinacea* are colonizing the edges at a slow pace.

3.2.2 Farm 2 (F2) and Constructed Farm Wetland 2 (CFW2)

F2 and CFW2 (Figure 3.5) are located at NT 856584; 55°49' N, 2°13' W. The average annual rainfall in the area is 825 mm, ranging from 600 to 900 mm, October being often the wettest month (2000-2007 data from Harelaw 2 Land Station (BADC), 2.2 km south east of CFW2). Air temperatures ranged between - 6°C in winter and 21°C in summer during the monitoring period.

The farm is a mixed beef and arable farm (total area of 550 ha; 364 ha arable, 57 ha permanent grassland, 57 ha rotational grassland) holding 130 cows (Angus x Frison, Limousine x Simmental) and produces potatoes, winter wheat, barley and carrots. Cows graze outside in summer and are housed on straw from November to March. Manure is stored in a midden and spread three times a year on potato fields. The outdoor feed passage is roofed and sprayers are filled over hard packed earth not adjacent to any drain, which limits pollution risks. However, there is no separation between roof water and farmyard runoff, which increases the volumes to be treated. Forty four sheep from a neighbouring farm graze in the field adjacent to the wetland.

Figure 3.5 View of CFW2 (summer 2008), which comprises five ponds in series planted with *P. australis* around the edges. The last and largest pond is seen in the foreground. The CFW inlet is located in the background.

CFW2 is located 0.8 km from the farm, at the foot of a hill (c. 65 m asl) and occupies 0.8 ha (Figure 3.6). It was constructed between April and May 2004 in a naturally wet area (rough grazing) for c. £8000 (including fencing but not the land cost), to control pollution from the fields (33.6 ha), farmyard (1.8 ha), and septic tanks (farm house and cottages). It is fenced and bordered by a small woodland and a pasture grazed by sheep and cattle. The wetland was originally planted with several species (e.g. *P. australis* at a density of 1 plant m^{-2}, *T. latifolia, I. pseudacorus*) and natural colonisation has also occurred. Vegetation is well establish and diverse (*Glyceria fluitans, Holcus lanatus, Lemna minor, Ranunculus omiophyllus*), and CFW2 hosts waterfowl (ducks, moorhen, coots and swans). A natural wetland is present c. 300 m from CFW2 and might have influenced regeneration and invertebrate colonization.

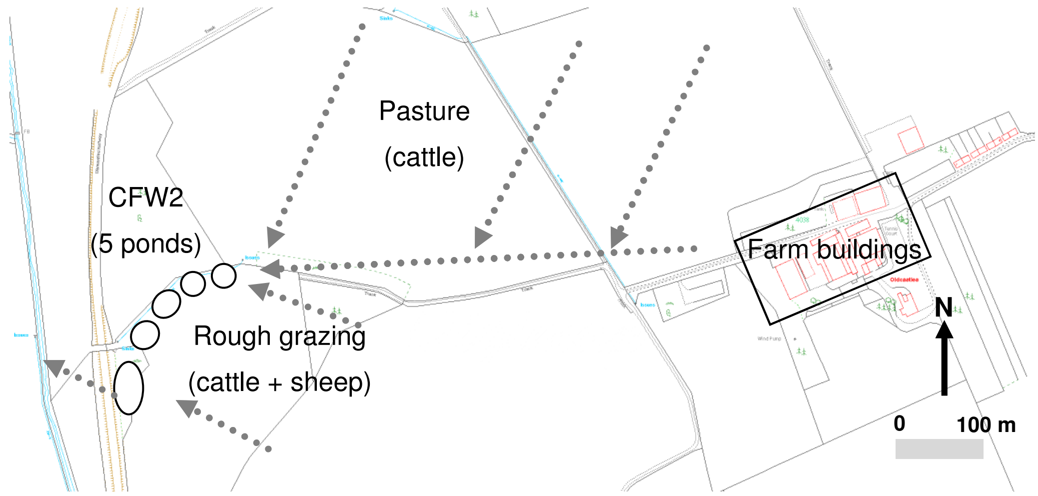


Figure 3.6 Location of CFW2 and farm buildings (shaded). The direction of farmyard runoff, field drainage, overland flow and wetland discharge is indicated by dashed arrows (Edina Digimap).

CFW2 was designed following the treatment volume approach and comprises five ponds (referred to as P1, P2, P3, P4 and P5) lined by compacted clay, separated by shallow vegetated areas submerged in wet conditions (Figure 3.7). The maximum volume to be treated was estimated at 39 m^3 ha^{-1}(V_t of 1377 m^3) calculated from a 18 mm rainfall in 60 min with a five-year return period, Hobkirk and Eckford soil series (clay-loam soil with fluvio-glacial sand as parent material; CEC >20 meq 100 g^{-1} soil) with runoff potential of 3 and Soil Index of 0.4. The actual volume of the overall system is c. 2400 m^3, i.e. 1.7 times *Vt*.

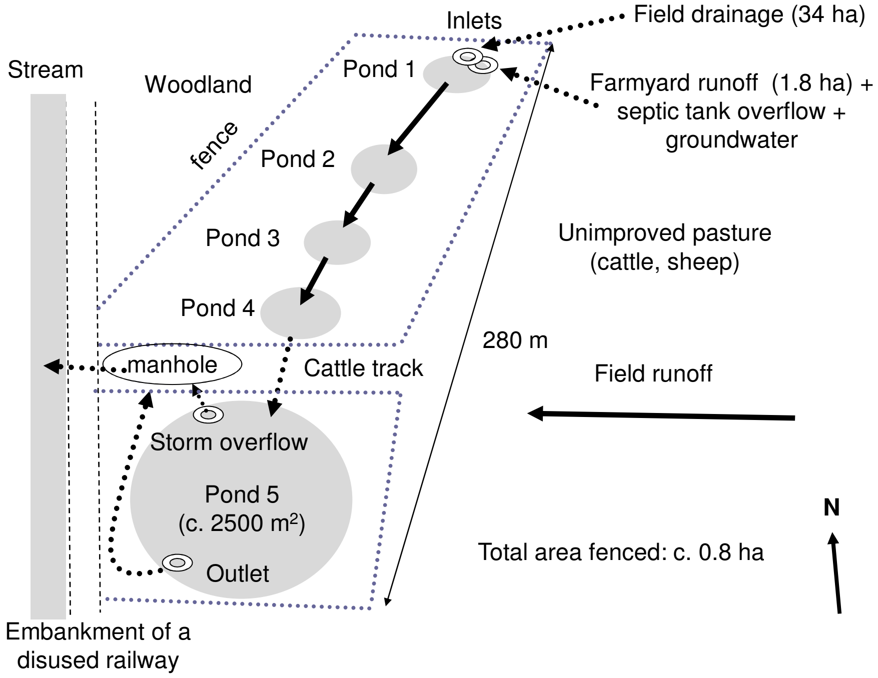


Figure 3.7 Sketch of CFW2 (not to scale). Dashed arrows represent underground piped flow and full arrows represent surface flow.

Wastewater from the farmyard and roofs (formerly piped directly into the burn) enters the wetland in a small "forebay pond" (P1, 50 m^2, up to 1.6 m deep) acting as a silt trap, and field drainage enters P1 through two pipes. Water leaves P1 through a pipe (30 cm diameter), runs through a long shallow vegetated area (c. 40 m long, 15 m wide) and through a series of three ponds (P2: 115 m^2, P3: 105 m^2, P4: 190 m^2, up to 1 m deep) separated by short (c. 20 m) shallow vegetated (grass or watercress) areas. Railway sleepers were placed between the ponds to create a serpentine flow path and increase sediment retention and water residence time. Flow then enters a large and deep pond (P5, c. 2500 m^2**,** up to 1.5 m deep, vegetated) through a pipe passing under a cattle track. Finally, under normal conditions, water discharges from P5 through a pipe (14 cm diameter) located at the south side of the pond running along the railway embankment back into an existing main drain and to a small stream, tributary to the Whiteadder Water. At higher flows, water may leave P5 by a vertical stormwater pipe (30 cm diameter) located at the northern end of the pond, which controls the maximum level and volume of the pond.

Chapter 4: Water Balance and Water Treatment Efficiency of two Constructed Farm Wetlands

The chapter presents the water balance and water treatment efficiency of two constructed farm wetlands (CFW1 and CFW2) located in the Scottish Borders and used to mitigate the impacts of agricultural pollution. It focuses on the hydrology of the wetlands, on water quality of farmyard runoff and field drainage and wetland effluent and investigates the influence of the season and design on performance.

4.1 Introduction

Constructed farm wetlands (CFWs) have been implemented in Scotland for more than 10 years by private or semi-private companies and small organisations, but until 2008, there was neither a clear legislative framework regulating their use, discharge and monitoring, nor a consensus on their design. In reality, in the absence of legislative framework and financial support, design was mainly driven by farmers' resources and willingness to devote land for water treatment. The absence of monitoring of CFWs and subsequent lack of information on performance triggered the need for more in-depth investigation.

The main objectives of this chapter are: 1) To assess the quality and quantity of the inflows into CFW1 and CFW2 in order to describe pollutant loads expected during the year and to study the relationships between farm characteristics and practices, climatic conditions and runoff characteristics; and 2) To evaluate the water treatment performance of the CFWs over time by assessing effluent quality and volume, and to identify the factors related to CFW characteristics and management (e.g. size, age, location in landscape, residence time), to the physical environment (e.g. season, rainfall, temperature), and to farm characteristics (e.g. pollutant and hydraulic loadings) influencing efficiency.

The hypotheses are the following: 1) Water treatment performance varies between CFWs due to the influence of hydraulic and pollutant loadings and design (which influences hydraulic residence time and treatment); 2) The efficiency of CFWs decreases in winter when effluent volumes and pollutant loadings are larger and temperatures lower: the CFW may become a source rather than a sink of pollutants; 3) The long-term performance of a CFW (especially for P removal) decreases with its increasing age: flow patterns change and pollutants accumulate or are flushed out. Sediment dredging is therefore required to increase pond performance.

4.2 Materials and methods

4.2.1 Site description

The main characteristics of the two CFWs investigated have already been detailed in Chapter 3, and are summarized as follows. CFW1 was built in 2005 and comprises a swale followed by a 2200 m^2 surface area, 1500 m^3 volume, up to 1.5 m deep non vegetated pond. It receives farmyard and roof runoff (3.22 ha), field drainage (c. 6 ha), septic tank overflow and groundwater (c. 20 m^3 d^{-1}). CFW2 was built in 2004 and consists of five ponds in series (four small ponds between 50 and 290 m^2 surface area and one final pond of c. 2500 m^2 surface area, up to 1.5 m deep), planted with reeds and separated by grassy areas. It receives runoff from farmyard and roofs (1.8 ha), field drainage (33.6 ha), septic tank overflow and groundwater.

4.2.2 Water balance assessment of the CFWs

The study of the water balance involved the assessment of the inputs into the wetlands, including rainfall-generated farmyard runoff and field drainage, septic tank overflow, rainfall falling on the CFWs, and groundwater, as well as the evaluation of the outputs, i.e. evapotranspiration, infiltration and outflow. In farm settings, difficulties arise when gauging flows in CFWs using conventional hydrological monitoring techniques due to great variations in flow and low water levels, susceptibility to biofilm development, and accumulation of debris (Edwards *et al.*, 2008). Hence a variety of techniques was used to estimate flow inputs and outputs.

4.2.2.1 Quantification of the inputs and outputs at CFW1

Monitoring of the inflow

In CFW1, since instrumentation was only available for one inflow monitoring point, a dam was built from plywood in October 2006 at the beginning of the swale and a 39 cm diameter and 1.8 m long pipe was inserted through it to quantify the combined flow of the two inflow pipes, i.e. field drainage, farmyard runoff and septic overflow. An Isco 730 bubbler module measuring pressure, which seemed particularly suitable for low flow measurement, was located at the base of the pipe for long-term level measurements and attached to an automatic compact water sampler (Model 6712, Teledyne Isco).

The inflow was extrapolated from level measurements given by the bubbler meter which were converted to flow using the flow-level relationship obtained from measurements by an Isco 750 area velocity meter (AV meter) deployed at the same location between 2nd October 2006 and 1st December 2006. For this meter, water depth readings are only accurate when the level is higher than 3 cm and velocity readings when particles and bubbles are present in the water since the meter measures velocity using the Doppler effect. Hence, low flow values given by the AV meter were removed, and only those obtained by time volumetric gauging at each site visit were used. Combining both datasets, a flow-level relationship was established for water depths below 15 cm in the pipe (Figure 4.1) and was applied over the whole study period due to levels being rarely above 15 cm, corresponding to a maximum flow of about 72 l s^{-1}.

The inflow was calculated as:

(Eq. 4.1) $$F = -0.0105 \times L^3 + 0.4501 \times L^2 + 0.4205 \times L$$

Where: F: inflow (l s^{-1}), L: water level (cm); R^2= 0.99.

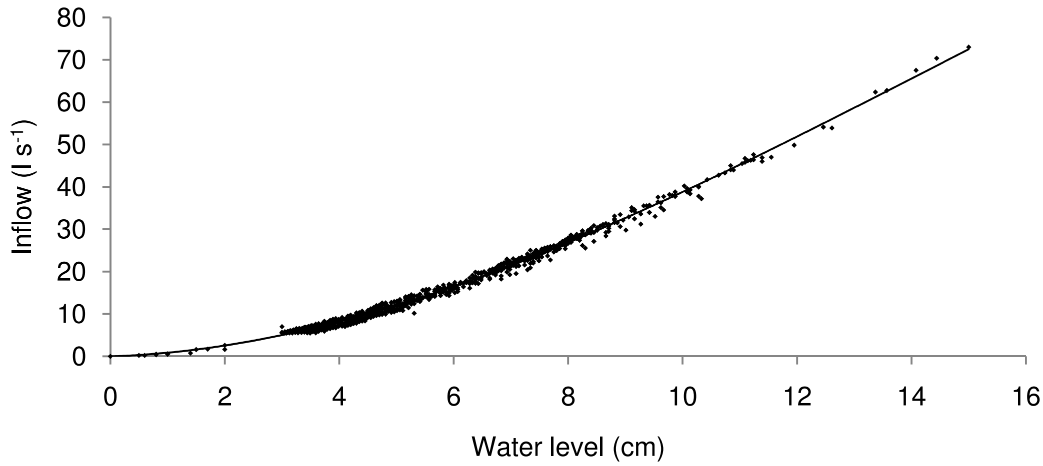


Figure 4.1 Best fit relationship between flow and water level in diameter pipe at the inlet of CFW1, for a level of less than 15 cm (volumetric gauged data where water level < 3 cm; area-velocity meter data 3-15 cm).

Although the bubbler line was cleaned during each field visit, problems occurred with level measurements due to biofilm growth and accumulation of debris on the line. Measurements sometimes drifted gradually upwards over time until the sensor was cleaned. Thus readings were corrected as follows and an example is shown in Figure 4.2. Corrections were applied using levels measured manually in the pipe at each field visit, and level measurements were adjusted to minimum values for periods where no inputs were expected (e.g. periods without rainfall and without inputs from water used for milk cooling), knowing that an average inflow of about 0.01 l s^{-1} exists due to septic tank input, estimated using measurements during dry weather and literature data (Grant and Moodie, 1997). Input from water used to cool the milk was also assessed by bucket and stopwatch during a milking period to compare with bubbler measurements.

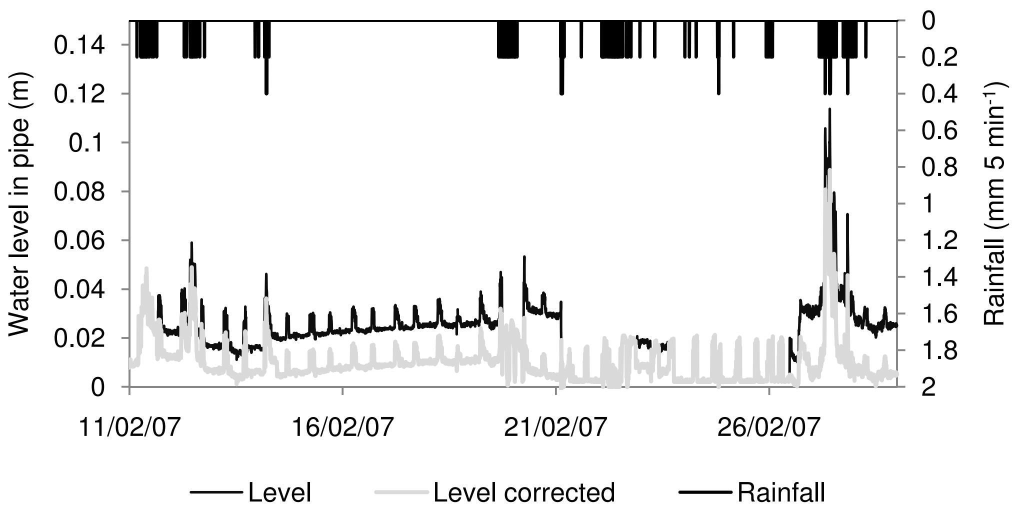


Figure 4.2 Example of a drift in level measurement by bubbler and correction to baseline level measured manually on 12/02/07, 26/02/07 and 28/02/07.

Monitoring of the pond water level and outflow

The outflow from the pond was originally measured using an Isco 4150 area velocity meter, placed on 14th September 2006 in the outlet pipe flowing into the adjacent ditch, but due to repeated instrument failure and non continuity of the measurements, the outflow was estimated thereafter from continuous monitoring of pond water level. A pressure transducer (PDCR 1830, Campbell Scientific) calibrated in the laboratory and connected to a logger (CR10X, Campbell Scientific) was set up on 11th April 2006 in the pond close to the outlet in a stilling well with a stageboard next to it, taking measurements every five minutes. The sensor was cleaned in 2007 and 2008 and calibrated in the laboratory in May 2008, obtaining a pressure-level relationship comparable to the initial one.

Level measurements were corrected manually when obvious and non realistic changes occurred (e.g. 10 cm level rise or drop in 1 h during a dry period due for example to interference with the cable by swans), working on hourly level averages. Outflow from the pond via the pipe only occurs when water level in the pond exceeds c. 70 cm (± 1 cm).

A relationship between outflow and water level in the pond (Figure 4.3) was derived from the pressure transducer data and readings from the stageboard at each visit coupled with volumetric flow measurements made with a bucket and stopwatch on each site visit, and flow measured by the area-velocity meter when available. The maximum outflow measured volumetrically was only about 7 l s^{-1}, so measurements from the AV meter at higher flows were used to deduce flows at higher levels. The maximum outflow detected by the AV meter was 25 l s^{-1}, with a maximum velocity of 1.8 m s^{-1}, and a maximum water level in the pipe of about 12 cm. The relation between flow and level showed a very slow increase in flow during the first few cm of level rise in the pond, followed by a steeper increase. A sigmoid relationship (Figure 4.3) was therefore fitted to take into account that the outflow increases less rapidly when water level in the pond exceeds the level of the outlet pipe (water level at gauge of 84.5 cm).

The equation used is as follows:

(Eq. 4.2) $$F = 0.0685 + \frac{25.5211}{\left(1+exp^{\left(\frac{-(L-82.98)}{0.8566}\right)}\right)^{0.3287}}$$

Where: F: flow (l s^{-1}), L: water level at gauge (cm); R^2= 0.99.

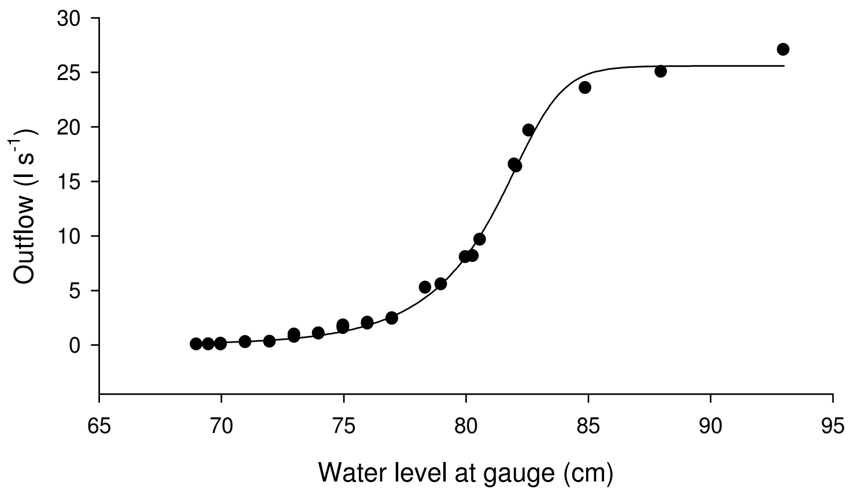


Figure 4.3 Sigmoid relationship between outflow and water level in the pond measured by pressure transducer (after correction) at CFW1.

Meteorological monitoring

Rainfall depth was recorded every 5 min from April 2006 to May 2008 using a ARG100 tipping bucket raingauge (Campbell Scientific) placed 2 m from the pond in an open area, initially calibrated in the laboratory (0.2 mm per tip) and checked regularly in the field by slowly pouring in a given amount of water and counting the tips. Consistency was obtained at ± 0.1 mm. Data were discarded for August 2007 and June 2008 due to grass seeds obstructing the raingauge inlet. To infill missing daily rainfall data at CFW1, regression relationships (R^2= 0.88 and 0.91) were derived between daily rainfall measurements at CFW1 and the closest data available from BADC (daily rainfall at Lochton at 4 km distance, and hourly rainfall at Charterhall, 7 km away). Since the regressions indicate factors of 0.98 and 1.015 and the sensitivity of the raingauge was 0.2 mm, a factor of 1 to 1 was used.

Evapotranspiration and infiltration were estimated using literature for Scotland and actual measurements of water level fluctuations in the pond when there was no outflow. Additionally, a large fibreglass container (61 x 46 x 53 cm) was installed next to the pond as an evaporation pan, with its rim levelled with the ground surface. The bucket was refilled with water on each site visit and water depth below the rim was measured on each occasion. Reliable estimates of evaporation were only obtained between June and December 2007, as the pan became unlevelled with time.

4.2.2.2 Quantification of the inputs and outputs at CFW2

Monitoring of the inflow

At CFW2, the combined inflow from field drainage (via two pipes), septic tank overflow, farmyard runoff and groundwater (via a separate pipe) was monitored at the outlet of P1 for 535 days between May 2006 and May 2008 as continuously as possible at 5 min intervals, using an Isco pressure transducer (Module 720) connected to an Isco compact sampler (Model 6712) and placed at the base of the pipe (29 cm diameter), 40 cm into it to avoid the "level drop" zone.

The sensor could only measure accurately levels above 3 cm and thus did not detect low flows, which are on average between 0.5 and 1 l s^{-1} (estimated by measuring water level in pipe during dry days) all year round due to groundwater and septic tank inputs. Volumetric flow gauging was impossible due to the lack of space beneath inflow and outflow pipes. The sensor was regularly cleaned from debris and biofilms and calibrated by measuring water level above it.

An AV-meter (Isco Module 750) was placed in the pipe at the outlet of P1 in May 2009 to try establish a relationship between velocity (and hence flow) and water level, but no useful relation was obtained due to velocity at low level being underestimated, and to very scattered data. Therefore, Manning's equation was used as the closest approximation to calculate velocity for a slope of 1.5 % and roughness coefficient of 0.011 (Tullis *et al.*, 1990) giving the relationship shown in Figure 4.4.

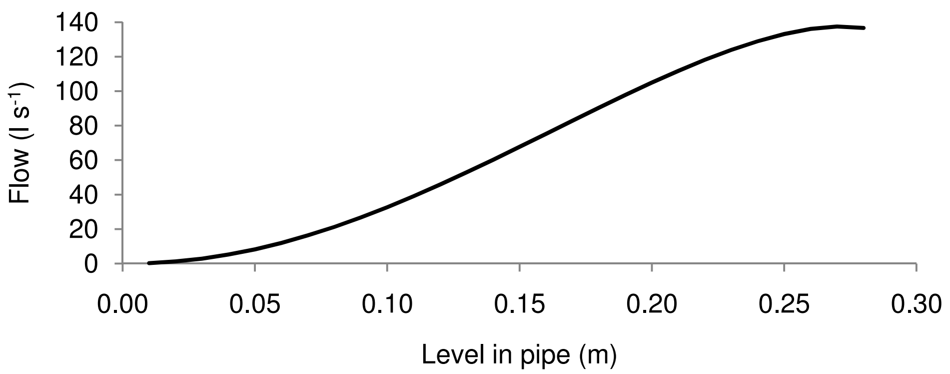


Figure 4.4 Theoretical flow-level relationship in the pipe at the outlet of P1, CFW2, calculated using Manning's equation (S= 1.5%, R= 0.011).

Monitoring of the outflow and level in P5

At CFW2, the outflow was monitored with an Isco 750 AV-meter taking measurement every 5 min, placed in a 14.2 cm diameter pipe at the outlet of P5, and attached to an automatic compact Isco sampler (Model 6712). The sensor detects relatively low water levels (a few mm) but level and velocity are only measured with confidence for levels above 3 cm and when sufficient air bubbles and particles are present in the water to reflect Doppler waves.

Therefore, Manning's equation, using the pipe's slope of 2% and a roughness coefficient R= 0.011 (PVC pipe) was also used to estimate low flows or when velocity was not measured by the sensor.

The storm overflow in P5, through which water was escaping at times in winter, was raised by 8 cm at the end of 2006, and a second riser (adding 10 cm) was installed in February 2007. However, it was removed temporarily from December 2007 to February 2008, when flooding occurred of the track between P4 and P5, caused by water escaping the 1st part of the wetland (due to inadequate levelling). Therefore, unknown quantities of water escaped through the overflow and on the track probably leading to underestimation of outflows, mostly in autumn and winter.

A pressure transducer (PDCR 1830, Campbell Scientific) connected to a CR10X datalogger, taking measurement every 5 min, and calibrated in the laboratory was placed in P5 within a vertical perforated pipe in June 2006 with attached stageboard, to assess water level fluctuations over time, and cleaned in 2007. Gaps in the water level dataset exist between December 2006 and February 2007, in April and May 2007 and June 2008, due to interference by cows and swans (which dragged or cut cables) and battery failures. The sensor was recalibrated at the end of the monitoring period (May 2008) showing a pressure-level relationship similar to the original one.

A sigmoid relationship (Figure 4.5) between flow and level (measured at each field visit by AV-meter) was used to estimate the outflow over the entire study period. The equation used is as follows:

(Eq. 4.3) $$F = \frac{19.7855}{1 + exp^{\frac{(-(L-70.5454)}{3.6869}}}$$

Where: F: flow (l s^{-1}), L: water level at gauge (cm); R^2= 0.98.

Although a higher level in the pond usually implies a higher outflow, the relation is not straightforward, due to the influence of obstacles clogging the outlet and slowing down the rate of increase of the outflow.

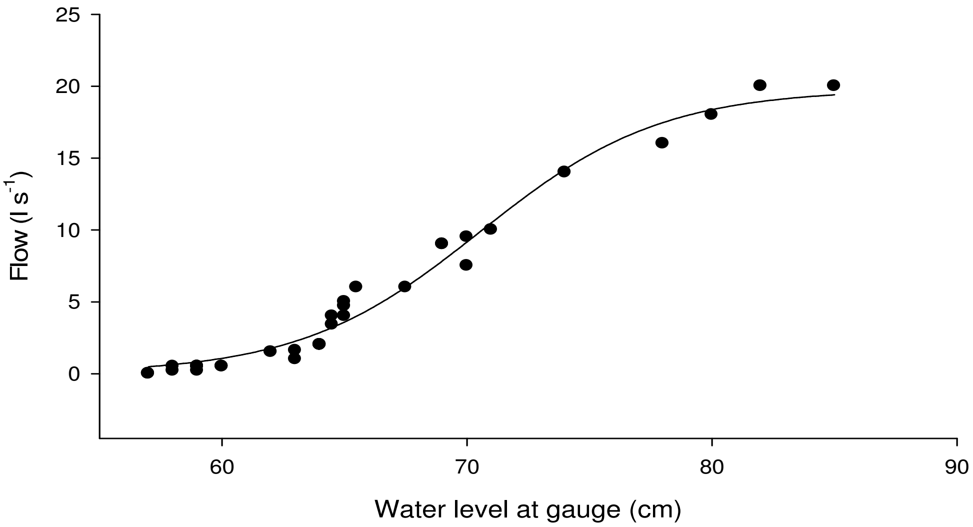


Figure 4.5 Relationship between outflow and water level measured by pressure transducer in P5, CFW2.

Meteorological monitoring

Rainfall was measured every 5 min from April 2006 to June 2008 using an ARG100 tipping bucket (0.2 mm) raingauge located close to P5, connected to a CR10X logger, calibrated in 2007 and 2008 and regularly cleaned. Gaps in the rainfall dataset exist between 8/11/06 and 16/11/06, 28/12/06 and 31/12/06, 1/01/07 and 9/02/07, 20/05/08 and 28/05/08, 21/06/08 and30/06/08, due to damage to the raingauge by cows and swans.

A strong linear relationship (y= 0.99 x, R^2= 0.89, n= 522) between daily rainfall measured at CFW2 and daily rainfall at the nearest BADC raingauge (Harelaw 2, 1.8 km from CFW2) allowed infilling of the gaps using a coefficient of 1.

Evapotranspiration and infiltration were estimated using existing literature on ponds and vegetated wetlands (Nkemdirin, 1970; Gavin and Agnew, 2000; Fermor *et al.*, 2001; Soulsby *et al.*, 2001, Jacobs *et al.*, 2002), and from measured water level fluctuations in P5 during periods without outflow.

The water balance estimated from the above measurements is subject to an unknown but probably large uncertainty, since not all the inputs and outputs to the system occurred through the pipes monitored. Diffuse inputs along the length of the wetland system were observed during site visits, including field drainage emerging within the system. Conversely outflow from the system also occurred along the track between P4 and P5 due to clogging with vegetation of the pipe between these ponds and insufficient bank height to retain water within the system. Overflow from storm overflow in P5 when the riser was removed could also not be assessed.

4.2.3 Water quality monitoring in the CFWs

4.2.3.1 Water sampling

Water quality and pollutant removal capacity of CFW1 and CFW2 was assessed in two ways, combining manual grab samples during low (e.g. < 2 l s^{-1}) and high flow conditions and automatic storm event samples during rainy periods. Samples were taken at the locations (names written in bold) shown in Figures 4.6 and 4.7.

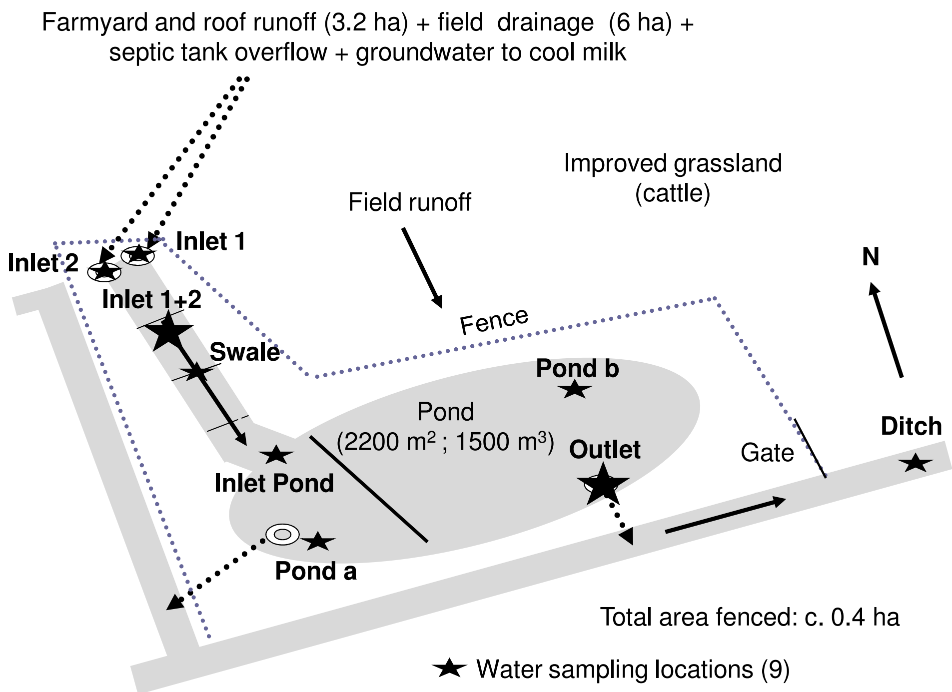


Figure 4.6 Schematic of CFW1 (Not to scale). Stars indicate water sampling locations, the two largest being the locations sampled during storm events.

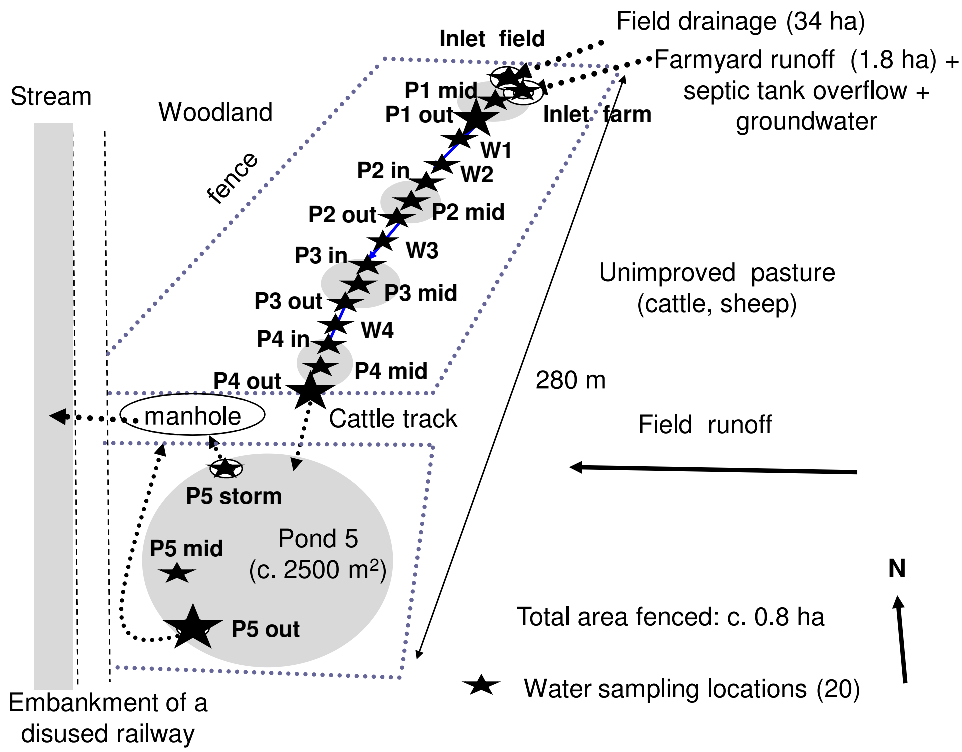


Figure 4.7 Schematic of CFW2 (Not to scale). Stars indicate water sampling locations, the three largest being the locations sampled during storm events.

Manual sampling allowed for the analysis of spatial and temporal variability in water quality within the CFWs, but its frequency was constrained by vehicle availability. On monthly to bi-monthly field visits from 2006 to 2008, grab samples were collected by hand in single-use HDPE vials on transects from the inlet to the outlet of the systems, 5 to 15 cm below the water surface, transported in a cool box to the laboratory and then refrigerated or frozen (if analysis was not possible within four days) until analysis for N and P species by automated colorimetry.

At several points along the wetlands, two samples were also taken in clean 1 litre polyethylene bottles, to be analysed for five days biochemical oxygen demand (BOD_5) and suspended solids (SS) respectively, within 24 h. At CFW1, samples were mainly taken at inlet 1 and inlet 2, inlet pond and outlet, from March 2006 till May 2008. At CFW2, samples for BOD_5 analysis were collected mainly at P1 out, P4 out, and P5 out. Since greater variability in water quality was expected during storm events, sampling frequency and spatial coverage for all pollutants were sometimes reduced during relatively dry conditions.

On three occasions (3/04/07, 6/08/07 and 28/01/08) during or after rainy conditions, water samples were taken at CFW1 (inlet 1+2, inlet pond and outlet), and at CFW2 (P1 out, P4 out and P5 out) in sterile 500 ml glass bottles to be analysed by SEPA for faecal coliforms and *streptococci*. The number of samples analysed was constrained by staff availability.

Water samples were also taken at the inlet and outlet of the CFWs during a limited number of storm events (4 complete events at CFW1 and 3 events at CFW2) when maximum pollution was anticipated to occur, using Isco compact automatic samplers (Isco 6712, Teledyne Isco) at the inlet and outlet of CFW2 and inlet of CFW1, and a full size sampler at the outlet of CFW1.

The sampling strategy was initially intended to be flow-proportional. However, following several failures in flow measurements and incorrect sampling timing, Isco samplers were programmed to take composite water samples (three 150 ml samples per 500 ml bottle in compact samplers and four 250 ml samples per 1000 ml bottle in the full size sampler) in each pre-cleaned polyethylene bottle at regular time intervals during rainy periods, with purge and rinsing of the suction line between samples to reduce cross-contamination. Samplers were programmed within 24 h of periods of heavy rainfall predicted by the Met Office, to start before the predicted rainy period, and then samples were retrieved after the rainfall event. In summer, to compensate for the lack of an on-site cooling unit, water and ice packs were placed around the sampling bottles to preserve samples in field conditions. Nevertheless, sample degradation may have occurred due to insufficient cooling, but was not quantified.

At CFW1, “inlet” samples were taken at the beginning of the swale (inlet 1+2), 1.5 m below the confluence of the two inputs in order for the combined flow (farmyard and field) to be sampled, by placing the suction line attached to the surface of a concrete slab in a zone of flowing water. “Outlet” samples were taken just before the outlet of the pond, the suction line placed above a glass fibre plate to minimise clogging.

At CFW2, “inlet” samples were taken at the outlet of the first pond (P1 out), 10 to 20 cm below the water surface and “outlet” samples were taken at the outlet of P5, 10 to 20 cm below the water surface. “Inlet” samples were considered to be representative of the combined runoff from all inlets since the volume of the first pond is small and residence time is negligible during storm events. However, these samples may contain some phosphorus and sediment resuspended at high flows.

4.2.3.2 Water analysis

Reactive phosphorus (RP) and total phosphorus (TP) were analysed at the University of Edinburgh laboratory using automated colorimetry (Bran & Luebbe AA2 autoanalyser), within four days from collection for unfrozen samples (or several weeks for frozen samples). RP was measured in unfiltered samples using a reaction with molybdate in the presence of ascorbic acid, and TP was measured after digestion with UV oxidation and a persulphate catalyst (absorption measured at 690 nm). For many of the samples, TP concentrations were lower than RP concentrations, probably due to TP being measured later than RP and samples being refrozen for several weeks before analysis, to P being absorbed to the vials or within biofilms, to microbial uptake or to instrument failure. To address this issue, several samples were analysed in 2008 for RP and TP at the same time and better results were obtained. Organic phosphorus (OP) was calculated as the difference between TP and RP.

Ammonium (referred to as NH_4 hereafter) and nitrate and nitrite combined (referred to as nitrate or NO_3) were analysed by the same laboratory, using automated colorimetry (Bran & Luebbe AA3 autoanalyser) within four days from collection for unfrozen samples (or several weeks for frozen samples). Ammonium determination involves a reaction between salicylate and dichloro-isocyanuric acid which forms, with the catalyst nitroprusside, a blue compound (absorption measured at 660 nm). Nitrate is first reduced to nitrite by hydrazine in alkaline solution with copper as catalyst, and nitrite then reacts with sulphanilamide and *N*-(1-naphthyl)-ethylenediamine dihydrochloride to form a pink compound measured at 550 nm. In 2007, several water samples taken in both wetlands by grab or storm sampling were analysed for nitrite only, to investigate the proportion of nitrite compared to nitrate.

At CFW2, the concentration of nitrite appeared negligible compared to the concentration of nitrate (0.3% at inlet and 0.5% at outlet on average, with maximum of 2.2% and 0.7% respectively, n= 62). At CFW1, nitrite represented a higher proportion (4% at inlet and 10% at outlet on average, with maximum of 15% and 13.5% respectively, n= 73).

Suspended solids (SS) were determined by vacuum-filtering water samples (between 50 ml and 1 l, depending on the expected amount of solids and the risk of clogging the filters) through a GF/C filter (pre-dried in the oven at 105°C and weighed) and by oven-drying at 105°C the wet filter and sediment to reach a constant weight. To remove larger fragments (e.g. straw) in a relatively consistent manner, samples were sieved (2 mm sieve) before filtration, and living organisms (e.g. worms) were removed from the filters after filtration. SS concentration (mg l^{-1}) was calculated as:

(Eq. 4.4) $$SS = \frac{(FS-F)}{V}$$

Where: FS: mass of dry filter and sediment (mg), F: mass of dry filter (mg), V: volume of sample filtered (l).

BOD_5 was measured within 24 h of sampling using the Oxitop® system (GmbH products, Germany) involving agitated incubation in brown bottles for 5 days at 20°C of oxygenated samples with nitrate inhibitor.

Faecal coliforms (FC) and faecal *streptococci* (FS) were analysed by SEPA (Riccarton Office, Edinburgh) using a two-membrane filtration technique (EA, 2002a, b). Amounts of FC (yellow colonies counted after incubation for 4 h at 30°C and 14 h at 37°C) and FS (red colonies counted after incubation for 18 h at 44°C) were expressed as colony forming units (cfu) per 100 ml. The number of storm event samples that could be analysed for faecal indicators was limited to three, due to the unpredictability of rainfall events and SEPA staff availability.

Water temperature and pH were measured *in situ* between 2 and 5 cm depth along the CFWs using a combined portable temperature probe and pH meter (HANNA Instrument HI 9025) calibrated with pH 4 and 7 buffer solutions and automatically temperature-corrected. Internal temperature of the CR10X datalogger storage box was recorded every 5 min and used as a surrogate for air temperature, probably overestimated due to shelter effect. Conductivity was measured in several storm samples using a portable conductivity meter (HANNA Instruments HI 9033).

4.3 Data analysis

4.3.1 Characterisation of the water balance and hydrology

The simplified overall water budget for the CFWs over the whole monitoring period was expressed as:

(Eq. 4.5) $$Vi + P + Gi = Vo + ET + I$$

Where: V_i: volume of influent, *P*: precipitation amount intercepted by the CFW, G_i: groundwater input, V_o: volume of effluent, *ET*: evapotranspiration, *I*: infiltration to groundwater.

The water balance was first assessed for the two sites for the overall monitoring period, using field measurements of inflow, outflow, rainfall, water levels, and estimations of evapotranspiration and infiltration. However, inaccuracy of the flow measurements over long periods of time (the longer the sensors stay uncleaned and uncalibrated, the more the readings diverge from actual flows) introduced large uncertainties into water budget calculations. Consequently, water budgets were also estimated for shorter time periods by quantifying each input and output source separately, using flow measurements shortly after sensor cleaning and calibration, especially during storm events, and manual measurements or literature values. This allowed to some extent precipitation characteristics and time of year to be related to pollutant loadings.

4.3.2 Study of the spatial and temporal variability in water quality

Water chemistry data from grab and storm samples were analysed separately or combined, after being tested for normality and homogeneity of variances. Since most of the data were non-normally distributed and transformations were unsuccessful, non-parametric statistical tests (e.g. Kruskal-Wallis) were used. Summary statistics were calculated for the different sampling points and correlation coefficients (Spearman) were analysed to look for relationships between rainfall, temperature and concentrations. Data were analysed to: 1) describe longitudinal gradients in pollutant concentration at different dates and seasons (spatial and temporal variability); 2) assess the effect of sampling location and timing on water quality; 3) evaluate the influence of rainfall, temperature and agricultural activities (e.g. slurry spreading) on water quality; 4) assess the treatment efficiency by concentration and mass; 5) investigate the probability for given water quality parameters of exceeding standards at the outlet, information of relevance for environmental policy; 6) examine correlations between pollutant concentrations to see if some parameters can be used as surrogates to determine others and thereby reduce monitoring costs.

4.3.3 Characterisation of the water treatment efficiency

Treatment efficiency by concentration

Water treatment efficiency was first calculated in terms of Concentration Reduction Efficiency (CRE, %) between inlet and outlet, expressed as:

(Eq. 4.6) $$CRE = 100 \times \frac{(Ci - Co)}{Ci}$$

Where: C_i: mean inlet concentration (mg l^{-1}), C_o: mean outlet concentration (mg l^{-1}).

CRE was calculated for the whole study period using the mean concentration of the pollutants from all the samples (grab and storm) collected, at inlets and outlets of the CFWs, and was also assessed for each storm event, to look for temporal variability in efficiency, aware of the limited meaning of such a short-term efficiency.

Estimation of pollutant fluxes, loads and treatment efficiency by mass

Relationships between pollutant concentrations and flows (for grab and storm samples) were investigated to assess whether concentrations could be extrapolated from flow data and thereby improve the accuracy of long-term loading assessments into and out of the CFWs (Verhoff *et al.*, 1980; Walling and Webb, 1985).

During storm events, between three and four samples were taken per bottle, and therefore, the average flow measured over the sampling period for a given bottle was used to investigate concentration-flow relationships. However, there were no significant relationships between concentrations and flows at the inlet or outlet of both wetlands and thus, assessment of loadings and treatment efficiency was done using averaged water chemistry and averaged inflow/outflow data. The Mass Reduction Efficiency (MRE, %) for a given pollutant was calculated as:

(Eq. 4.7) $$MRE = 100 \times \frac{(Mi - Mo)}{Mi} = 100 \times \frac{(Vi \times Ci) - (Vo \times Co)}{(Vi \times Ci)}$$

Where: M_i: mass of pollutant entering the CFW (g), M_o: mass of the same pollutant leaving the CFW, V_i: total influent volume, V_o: total effluent volume, C_i: mean inflow concentration, C_o: mean outflow concentration.

For a few selected storm events (when sampling periods were comparable), mean fluxes (F, mg s^{-1}) of NH_4, NO_3 and RP at the inlets and outlets of the CFWs were calculated using the following expression, adapted from Verhoff *et al.* (1980) and Walling and Webb (1985):

(Eq. 4.8) $$F = \frac{\sum_{s=1}^{n} Cs\ Qs}{n}$$

Where: C_s: concentration of the pollutant in water sample *s* at inlet or outlet (mg l^{-1}), Q_s: flow measured during collection of sample *s* (l s^{-1}).

Subsequently, Flux Reduction Efficiency (FRE, %) was expressed as:

(Eq. 4.9) $$FRE = \frac{(Fi - Fo)}{Fi} \times 100$$

Where: F_i: mean inlet pollutant flux (mg s^{-1}), F_o: mean outlet pollutant flux (mg s^{-1}).

Investigation of hysteresis patterns

Hysteresis patterns during storm events were investigated to identify sources of pollutants and to understand mobilisation processes, e.g. early and fast mobilisation ("first flush") from the yards or delayed and slower mobilisation from the fields.

4.4 Results for CFW1

4.4.1 General description of the monitoring period at CFW1

CFW1 was studied between 16th February 2006 and 30th June 2008. The water quality monitoring began in February 2006, rainfall, temperature and water level measurements started in April 2006, and flow measurements in September 2006.

The total amount of rainfall recorded between April 2006 and 30 June 2008 was 1600 mm, with August and September 2006, June and July 2007, and January and April 2008 being the wettest months, and June 2006, April 2007 and February and May 2008 being the driest (Figure 4.8). Short-term trends (2000-2007) at Lochton station (located 4 km from CFW1) show that average total rainfall in the area of CFW1 (c. 675 mm yr^{-1}) is only slightly higher in autumn-winter (345 mm) than in spring-summer (329 mm). However, long-term trends for south-east Scotland (Meteorological Office, 1981) indicate slightly higher precipitations in spring-summer (370 mm) than in autumn-winter (360 mm).

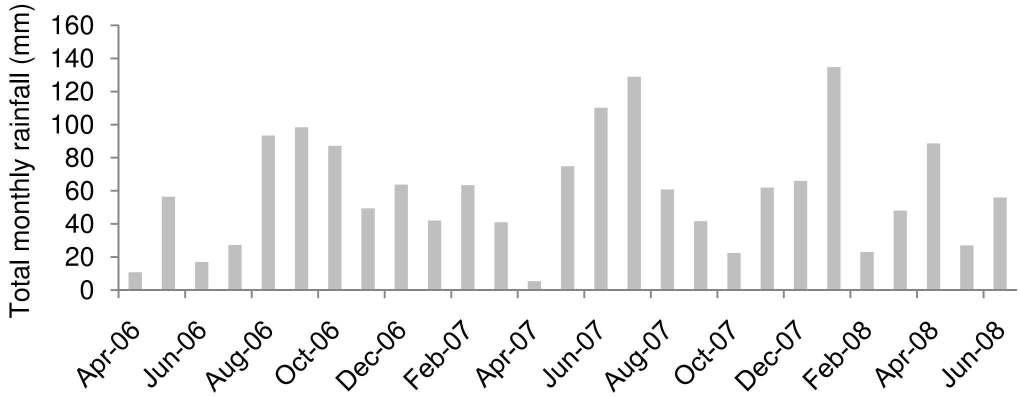


Figure 4.8 Total monthly rainfall over the monitoring period at CFW1.

Only a few heavy rainfall events were recorded. The highest daily rainfall totals were: 57.8 mm on 18th August 2006, 32.2 mm on 14th September 2006, 26.6 mm on 24th September 2006, 25.0 mm on 22nd July 2007, and 26.6 mm on 24th September 2007. Rainfall intensity reached 13 mm h^{-1} on 18th August 2006 and 12.4 mm h^{-1} on 14th September 2006. The maximum inflow recorded was approximately 70 l s^{-1} on 21st June 2007 (5.5 mm rainfall in 25 min), and the maximum outflow was 28 l s^{-1} on 22nd July 2007 (37 mm rainfall in 17 h).

Air temperature measured in the datalogger box varied between 22°C in summer (July 2006) and – 9°C on one occasion in winter (21st December 2007), with 2007 and 2008 being slightly cooler years than 2006 (Figure 4.9). Mean temperature over the study period was 9°C, which is slightly higher than the long-term (1971-2000) mean annual temperature of 7-8°C for east Scotland (MetOffice, 2009).

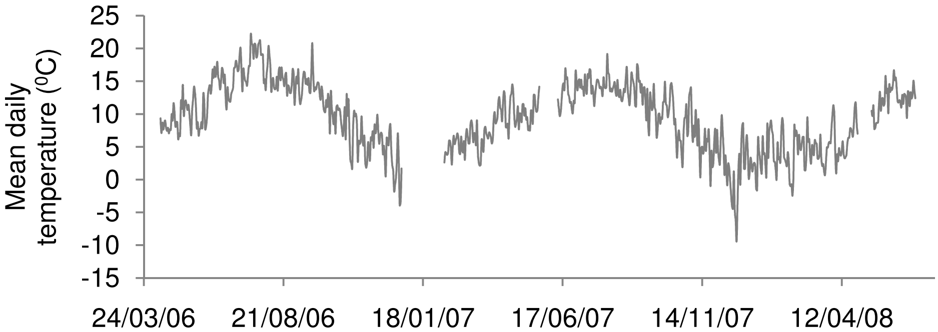


Figure 4.9 Mean daily temperature recorded inside the datalogger box at CFW1 during the monitoring period.

4.4.2 Water balance at CFW1

4.4.2.1 Water balance overview

Table 4.1 summarizes the estimated inputs and outputs for CFW1, and calculations are explained in more detail below. Despite uncertainties linked to errors in level measurements caused by sensor drifts or failures and gaps in the monitoring period, estimates of inputs and outputs are close, which suggests that all the components of the water balance have been relatively satisfactorily accounted for, allowing mass flux estimates to be assessed with more confidence. Smaller uncertainties are attached to the outflow assessment, due to the more accurate and continuous water level monitoring.

Table 4.1 Water balance overview for CFW1.

Period	Monitoring between 2/10/06 and 30/06/08 Data for 430 days at inlet and 560 days at outlet (208 and 78 days missing respectively from a total of 638 days)
Total rainfall	1296 mm
Rainfall volume intercepted	2852 m^3, c. 4.5 $m^3\ d^{-1}$
Inflow (Farmyard runoff, field drainage, septic tank overflow and water to cool the milk)	Overall input volume: c. 62 271 m^3 , i.e. c. 98 $m^3\ d^{-1}$ Average overall inflow: c. 1.1 l s^{-1} , (Summer/spring: 1.1 l s^{-1}; Autumn/winter: 1.2 l s^{-1}) Farmyard input: 26 594 m^3; c. 42 $m^3\ d^{-1}$ Field input: 20 577 m^3, c. 32 $m^3\ d^{-1}$
Groundwater inputs (from below)	Not assessed; Assumed to be negligible
Outflow	Overall outflow volume: 39 444 m^3, i.e. c. 62 $m^3\ d^{-1}$ Average outflow: 0.7 l s^{-1} Spring/summer: 0.7 l s^{-1}, c. 60 $m^3\ d^{-1}$ Autumn/winter: 0.8 l s^{-1}, c. 67 $m^3\ d^{-1}$
Evapotranspiration	Daily mean: 1.2 mm d^{-1}, 2.6 $m^3\ d^{-1}$
Infiltration	Daily mean: 15 mm d^{-1}, 33 $m^3\ d^{-1}$
Total inputs	c. 103 $m^3\ d^{-1}$
Total outputs	c. 98 $m^3\ d^{-1}$

4.4.2.2 Individual inputs

Inflow sources at CFW1, identified from discussions with the pond designer and farmers, include farmyard runoff from 3.22 ha (2.28 ha impermeable, roof area: c. 40%), field drainage from c. 6 ha of improved pasture, overflow from the septic tank serving the farmhouse (four to six people) and groundwater pumped from a borehole in the farm and used to cool the milk. Direct overland flow, infiltration through the banks and groundwater inputs into CFW1 could not be quantified but are considered to be limited since the pond has a clay base, its banks are raised above ground level and a field drain intercepts drainage a few meters above it.

Direct input from rainfall

The volume of rainfall intercepted directly by the pond between October 2006 and June 2008 was calculated by multiplying the total rainfall amount during this period (1296 mm) by the surface area of the pond (c. 2200 m^2), i.e. c. 2852 m^3.

Septic tank overflow

Effluent quantity and quality varies greatly between septic tanks and fluctuates during the day, depending on the design and operation of the tank and water use by the owners. According to literature (Grant and Moodie, 1997), based on a typical septic tank built for 5 population equivalents (comparable to the number of people at F1 including staff), the volume of water input is estimated at 400 l per person per day, i.e. 2000 l d^{-1} in total, which corresponds to an average flow of about 0.023 l s^{-1}. Using the above information and by taking water samples and measuring inflow during periods of no rainfall, septic tank overflow was characterized at CFW1. It varied between 0 l s^{-1} and 0.05 l s^{-1}. A maximum volume of 2 m^3 d^{-1} (i.e. if all the septic effluent goes to the CFW) contributes approximately 1276 m^3 for the whole period of 638 days (c. only 2% of the whole input).

Water used to cool the milk

Cows are milked twice a day (5-9 am and 4-8 pm), and during the milking process, groundwater is pumped to cool the milk and is released into CFW1. Using information from the farmer and flow measured by the Isco bubbler probe (Figure 4.10) and bucket/stopwatch, the daily volume of groundwater diverted to the pond was estimated to be on average 22 m^3 (± 1.5 m^3) (between 20 and 28 m^3), the flow varying between 1.5 and 2.2 l s^{-1} for 1.5 to 2 h, generating c. 14 036 m^3 over 638 days. This daily input was significant considering the volume of CFW1. Groundwater sampled during milking time (e.g. 14 May 2008) at inlet 2, during a dry period, contained a high concentration of NO_3 (> 20 mg l^{-1}, indicating groundwater contamination) but negligible concentration of RP.

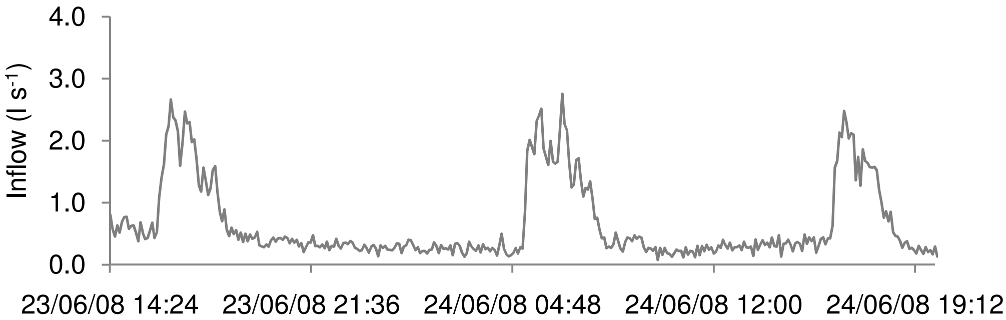


Figure 4.10 Inflow fluctuations due to water used for milk cooling at CFW1.

Overall Inflow (septic tank, groundwater, field drainage, farmyard runoff)

Since the flow monitoring equipment was removed in December and January of 2006-2007 and 2007-2008 to avoid damage from freezing conditions, flow during these periods was estimated as follows. It was assumed that 50% of rain falling on the fields entered the CFW as drainage during the first period considered (115 mm rainfall in 71 d), and 70% during the second much wetter period (138 mm in 44 d), and also that 90% of rain falling on the farmyard entered CFW1, taking into account losses through horizontal runoff and evapotranspiration. The input for the whole period was therefore 62 271 m^3, c. 98 m^3 d^{-1}, i.e. an average inflow of 1.1 l s^{-1} with autumn/winter mean inflow of 1.2 l s^{-1} and spring/summer mean inflow of 1.1 l s^{-1}.

Field drainage and farmyard runoff

Field drainage and farmyard runoff inputs were significant in the long-term compared to septic tank overflow and water used to cool the milk. Field drainage was observed to be much larger in autumn and winter when the soil was saturated (at field capacity) and evapotranspiration much lower. The overall contribution of runoff from rainfall (1296 mm) over impermeable areas (2.28 ha) was estimated to be around 26 594 m^3 over the whole period (assuming 90% runoff of total rainfall onto impermeable area), and therefore, field drainage could be estimated as the difference between the total inflow and runoff from the impervious areas, subtracting input from septic tank and from water used to cool the milk, i.e. 20 577 m^3, 42 $m^3\ d^{-1}$.

4.4.2.3 Individual outputs

Outflow

An overall outflow volume of 39 444 m^3 (over 638 d) was measured, i.e. 62 $m^3\ d^{-1}$, corresponding to an average outflow of c. 0.7 l s^{-1}. Data suggested lower outputs during spring/summer than during autumn/winter taking into account evaporation, i.e. 10 to 15% less, with mean outflows of 0.7 and 0.8 l s^{-1} respectively.

Evapotranspiration

Losses by evaporation and infiltration were estimated using pond level fluctuations when no rainfall and no outflow was occurring, taking into account potential inputs (groundwater and septic tank). Losses by transpiration were considered negligible due to the limited presence of macrophytes. Evaporation, which is influenced by air temperature and humidity, wind speed, solar radiation, as measured by level fluctuations in the pan, ranged between 0 mm d^{-1} in winter and 2.8 mm d^{-1} maximum in June 2007 (Table 4.2).

Table 4.2 Measurements of evaporation using level fluctuations in an evaporation pan at CFW1.

Measurement period	Evaporation (mm d^{-1}) (estimated error corresponding to reading accuracy: ± 1 mm / number of days in measurement period)
15/06/07 to 17/06/07	2.8
17/06/07 to 6/08/07	0.7
6/08/07 to 09/08/07	2.7
09/08/07 to 13/08/07	1.8
13/08/07 to 21/08/07	0.7
21/08/07 to 5/10/07	0.8
5/10/07 to 17/11/07	0.6
17/11/07 to 21/11/07	0.0
21/11/07 to 16/12/07	0.0
28/01/07 to 10/03/08	0.3

Evaporation was positively correlated (r_s= 0.83, p= 0.01) to temperature recorded by the datalogger (Figure 4.11).

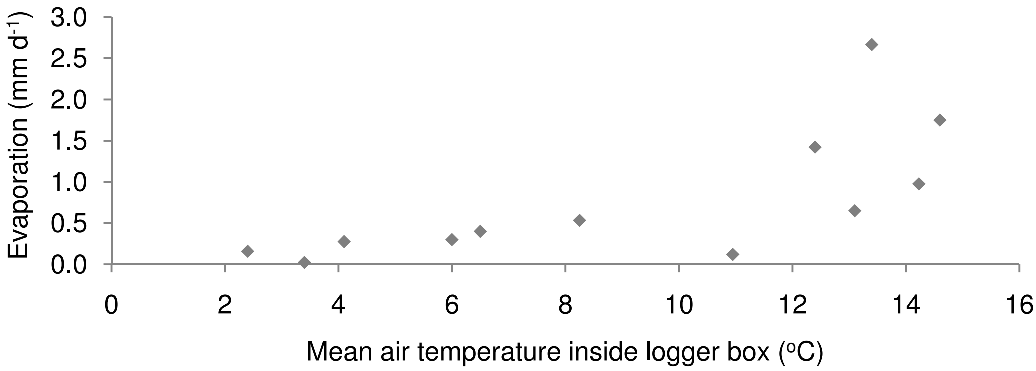


Figure 4.11 Relationship between evaporation measured in the pan and mean temperature recorded by datalogger at CFW1.

However, the limited number of values and the fact that wind speed and humidity were not assessed, did not allow a meaningful linear relationship to be established and hence, long-term data for the Scottish Borders (Table 4.3) were used to estimate evaporation. Data suggest a mean daily evaporation of 1.3 mm and monthly evaporation rate of 40 mm, with winter evaporation of 13.3 mm per month and summer evaporation of 66.7 mm per month (up to 2.8 mm d^{-1} on average). Applying the mean daily evaporations for each month between October 2006 and June 2008 gives a mean evaporation of 1.2 mm d^{-1}, and therefore the volume lost by evaporation was estimated at 1672 m^3, i.e. 2.6 m^3 d^{-1}.

Table 4.3 Long-term rainfall and evaporation estimates for south-east Scotland (Meteorological Office, 1981) (FC: field capacity).

Month	J	F	M	A	M	J	J	A	S	O	N	D	Total
Rainfall (mm)	65	50	40	45	55	55	65	85	65	65	75	65	730
Mean daily rainfall	2.1	1.7	1.3	1.5	1.8	1.8	2.1	2.7	2.2	2.1	2.5	2.1	-
Potential evaporation (mm)	5	10	30	50	75	85	85	65	40	25	5	5	480
Mean daily evaporation	0.2	0.3	1.0	1.7	2.4	2.8	2.7	2.1	1.3	0.8	0.2	0.2	-
Cumulated water deficit (mm)	FC	FC	FC	5	25	55	75	55	30	FC	FC	FC	-

Infiltration

Infiltration was estimated from changes in pond water level when no outflow occurred and during periods of no rainfall, taking into accounts inputs of milk cooling water and septic tank overflow and outputs in the form of evaporation. At an early stage after the pond's construction, in 2005 and the beginning of 2006, infiltration was significant (up to 17.6 mm d^{-1}), but it appears to have decreased over time (last evaluation of 13 mm d^{-1} in June 2008), due to self-sealing of the pond. Calculations (for June, July, August 2006, April 2007, February and June 2008) show an average infiltration rate of 15 mm (± 2 mm) per day (between 12 and 18 mm d^{-1}), assuming a daily volume of 24 m^3 used to cool the milk and evaporation rates from Meteorological Office (1981). Considering that CFW1 was built on soils of the Whitsome series, comprising clay loam topsoil over clay loam subsoil, and that its base was compacted, this infiltration rate was relatively high, but in the range of hydraulic conductivity values found in the literature, i.e. between 3.6 mm d^{-1} (clay, silty clay and sandy clay) and 36.5 mm d^{-1} (clay loam) (INSTAAR, 2002). It could be overestimated due to assumptions made regarding pond inputs and outputs. Ideally, the use of lysimeters such as the one used under ICWs (circular collecting pans covered with gravels) and installed during construction would help assess infiltration rate more accurately.

4.4.3 Water quality and treatment efficiency at CFW1

4.4.3.1 Overview of the water sampling period

Figure 4.12 shows the hourly rainfall and average hourly inflow into and outflow from the pond over the whole monitoring period. Grab sampling along the length of CFW1 took place on 37 occasions and storm event sampling was carried on 10th and 27th February 2007, 12th, 13th and 17th June 2007, 6th to 9th and 13th to 15th August 2007 and 17th to 21st November 2007.

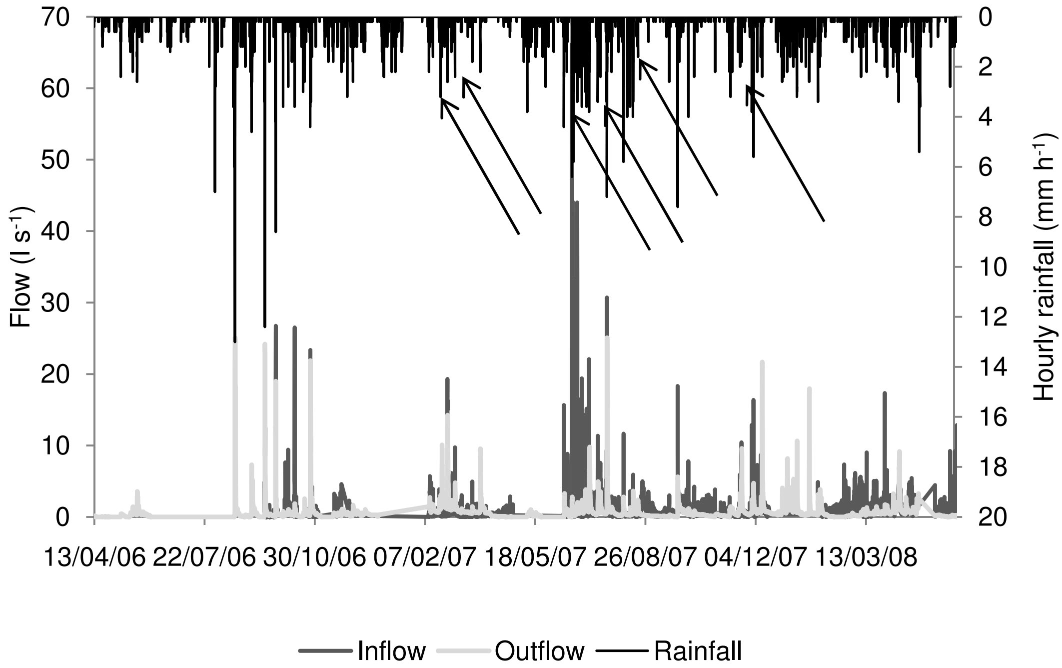


Figure 4.12 Mean hourly inflow, mean hourly outflow and hourly rainfall at CFW1 between April 2006 and May 2008. The time of storm event sampling are indicated by arrows.

Table 4.4 summarizes chemistry data for samples taken from the inlet to the outlet of CFW1, and in the ditch downstream of the pond to illustrate the influence of adjacent land use on water quality. Inlet 1+2 and outlet include average concentrations obtained from samples taken during seven storm events.

Table 4.4 Water quality along CFW1 between 2006 and 2008 (n: number of samples, SE: standard error).

Water quality parameter		Water sampling location								
		Inlet 1	Inlet 2	Inlet 1+2	Swale	Inlet Pond	Pond a	Pond b	Outlet	Ditch
NH_4 (mg l^{-1})	n	31	23	29	28	38	18	29	45	16
	Mean	16.2	22.8	16.6	15.7	12.4	10.1	8.97	9.58	4.49
	SE	2.5	5.6	2.4	2.4	2.1	1.5	1.68	1.15	1.85
	Median	11.9	11.6	14.0	13.6	8.94	10.1	6.42	9.06	2.60
	Max	54.9	100	65.1	47.7	60.3	20.8	36.4	31.7	30.3
NO_3 (mg l^{-1})	n	38	23	29	28	38	18	29	45	16
	Mean	28.3	22.9	22.1	26.6	13.7	9.14	7.33	7.13	21.4
	SE	5.4	4.1	4.8	6.4	3.1	2.81	1.93	1.36	21.2
	Median	16.9	20.2	10.3	14.4	5.13	4.32	3.33	3.54	12.6
	Max	152	66.4	92.3	154	64.1	46.9	46.0	45.0	66.2
RP (mg l^{-1})	n	29	23	19	18	29	18	17	36	16
	Mean	2.42	0.606	1.53	1.62	1.70	1.46	1.18	1.34	0.672
	SE	0.47	0.134	0.31	0.38	0.29	0.17	0.16	0.09	0.248
	Median	1.39	0.325	1.33	1.16	1.46	1.53	1.22	1.42	0.259
	Max	9.24	2.39	4.56	5.10	7.94	2.56	2.61	2.32	3.87
OP (mg l^{-1})	n	7	6	-	-	-	-	-	43	-
	Mean	0.850	0.360	-	-	-	-	-	0.390	-
	SE	0.120	0.060	-	-	-	-	-	0.010	-
	Median	0.240	0.220	-	-	-	-	-	0.290	-
	Max	2.08	1.11	-	-	-	-	-	1.32	-
BOD_5 (mg l^{-1})	n	22	11	6	-	21	5	2	26	-
	Mean	85.0	109	392	-	73.7	12.6	7.50	19.8	-
	SE	19.4	34.1	26.3	-	20.1	5.3	3.50	2.4	-
	Median	52.0	130	370	-	30.0	8.00	7.50	17.5	-
	Max	430	270	560	-	400	30.0	4.00	50.0	-
SS (mg l^{-1})	n	20	15	18	-	23	4	-	34	-
	Mean	111	89.1	481	-	179	90.1	-	40.5	-
	SE	39	27.0	65	-	61	11.2	-	5.4	-
	Median	68.2	39.3	238	-	65.4	90.2	-	30.5	-
	Max	796	366	1700	-	1190	112	-	160	-
FC (cfu 100 ml^{-1})	n	-	-	3	-	3	-	-	3	-
	Mean	-	-	72 500	-	71	-	-	4450	-
	SE	-	-	41 400	-	40	-	-	3910	-
	Max	-	-	150 000	-	150	-	-	12 250	-
FS (cfu 100 ml^{-1})	n	-	-	3	-	3	-	-	3	-
	Mean	-	-	5900	-	8320	-	-	3230	-
	SE	-	-	263	-	2360	-	-	1810	-
	Max	-	-	15 000	-	15	-	-	9500	-
pH	n	31	26	15	19	29	22	17	29	15
	Mean	7.51	7.89	7.71	7.61	7.76	8.01	8.02	7.89	7.50
	SE	0.05	0.09	0.10	0.10	0.09	0.12	0.12	0.10	0.07
	Max	8.05	8.97	8.76	8.80	9.38	9.38	9.35	9.52	8.08
Conductivity (µS cm^{-1})	n	4	3	24	-	2	-	-	18	-
	Mean	1370	1310	1570	-	1350	-	-	1180	-
	SE	175	570	110	-	166	-	-	27	-
	Max	1600	2430	2490	-	1520	-	-	1270	-

- Data not available.

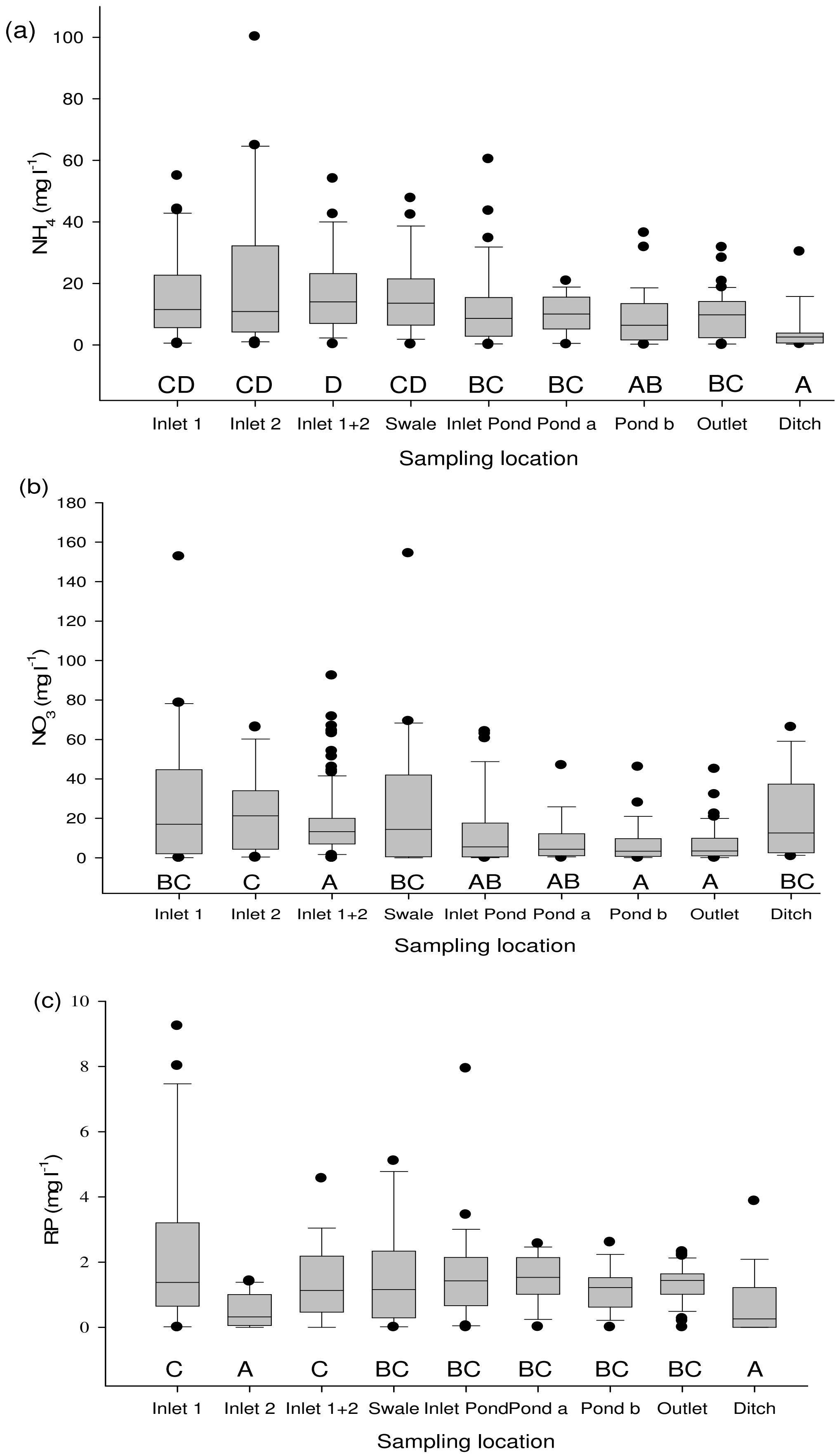
Contamination at the inlet was significant for all pollutants, concentrations reaching 152 mg l^{-1} for NO_3, 100 mg l^{-1} for NH_4, 9.2 mg l^{-1} for RP, 560 mg l^{-1} for BOD_5, 1690 mg l^{-1} for SS, 150 000 cfu 100 ml^{-1} for FC, and 15 000 cfu 100 ml^{-1} for FS. Inflow water pH was on average slightly alkaline, which suggests no strong contamination by acidic effluent such as silage runoff, and pond pH reached relatively high values (c. 9.5) in summer, probably due to the presence of green algae consuming CO_2.

Outlet concentrations of all pollutants were on average lower that at inlet, but remained high compared to river quality standards, reaching 45 mg l^{-1} for NO_3, 32 mg l^{-1} for NH_4, 2.3 mg l^{-1} for RP, 50 mg l^{-1} for BOD_5, 160 mg l^{-1} for SS, 12 250 cfu 100 ml^{-1} for FC and 9500 cfu 100 ml^{-1} for FS. For BOD_5, from 24 samples at outlet, nine (i.e. 38 %) were above 20 mg l^{-1}. Concentrations of OP represented a significant fraction of the TP, this fraction being larger on average at the inlet than the outlet: 52% at inlet 1, 67% at inlet 2, 35% at inlet 1+2 (maximum of 100%), and 20% at outlet (maximum of 53%).

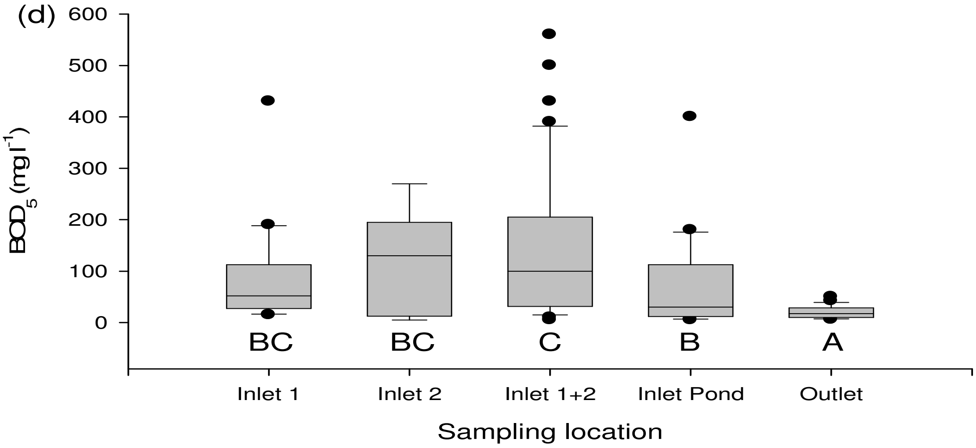
4.4.3.2 Spatial and temporal fluctuations in water quality

Spatial heterogeneity in water quality along CFW1

Kruskal Wallis tests and pairwise comparisons showed significant differences in median concentration between water sampling locations for NO_3 (H= 32.19, p< 0.0001), NH_4 (H= 48.81, p< 0.0001), RP (H= 49.76, p< 0.0001), BOD_5 (H= 19.89, p< 0.0005) and SS (H= 13.46, p< 0.0092) (Figure 4.13). Median NH_4 concentration was significantly lower at outlet (9.1 mg l^{-1}) than at inlet 1+2 (14 mg l^{-1}), but no difference existed between outlet and inlet pond (8.9 mg l^{-1}). Differences existed between the lowest concentrations of BOD_5 and SS at outlet (18 mg l^{-1} and 27 mg l^{-1} respectively) and the concentrations at inlet 1+2 (100 mg l^{-1} and 71 mg l^{-1}), inlet 1 (52 mg l^{-1} and 68 mg l^{-1}) and inlet pond (30 mg l^{-1} and 65 mg l^{-1}). Median NO_3 concentration was not significantly lower at outlet (3.5 mg l^{-1}) than at inlet pond (5.1 mg l^{-1}), i.e. no significant treatment effect of the pond, maybe due to dilution. Nitrate concentration in the ditch was also significantly higher than in the outflow, due to the impact of surrounding fields, whose runoff was not intercepted by CFW1.



(a)
NH_4 (mg l^{-1})
CD CD D CD BC BC AB BC A
Inlet 1 Inlet 2 Inlet 1+2 Swale Inlet Pond Pond a Pond b Outlet Ditch
Sampling location
(b)
NO_3 (mg l^{-1})
BC C A BC AB AB A A BC
Inlet 1 Inlet 2 Inlet 1+2 Swale Inlet Pond Pond a Pond b Outlet Ditch
Sampling location
(c)
RP (mg l^{-1})
C A C BC BC BC BC BC A
Inlet 1 Inlet 2 Inlet 1+2 Swale Inlet Pond Pond a Pond b Outlet Ditch
Sampling location



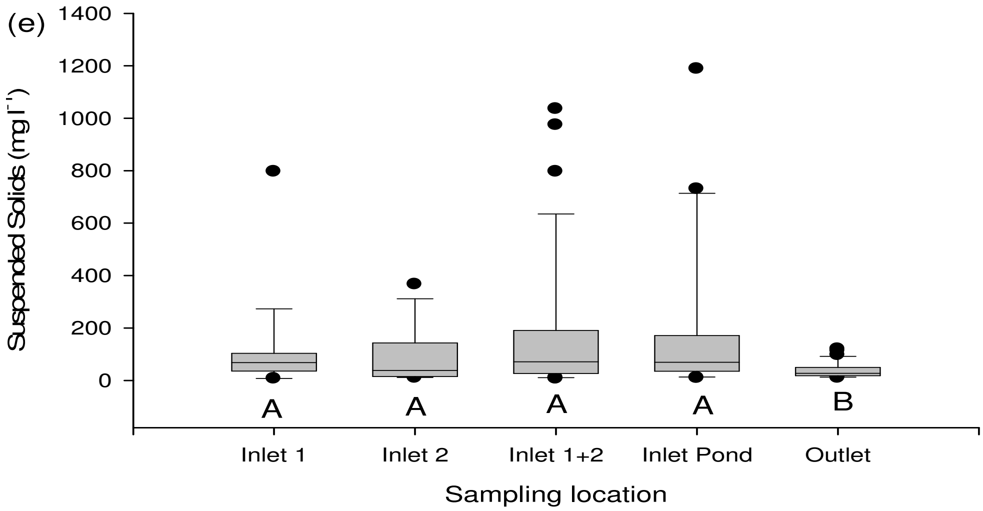
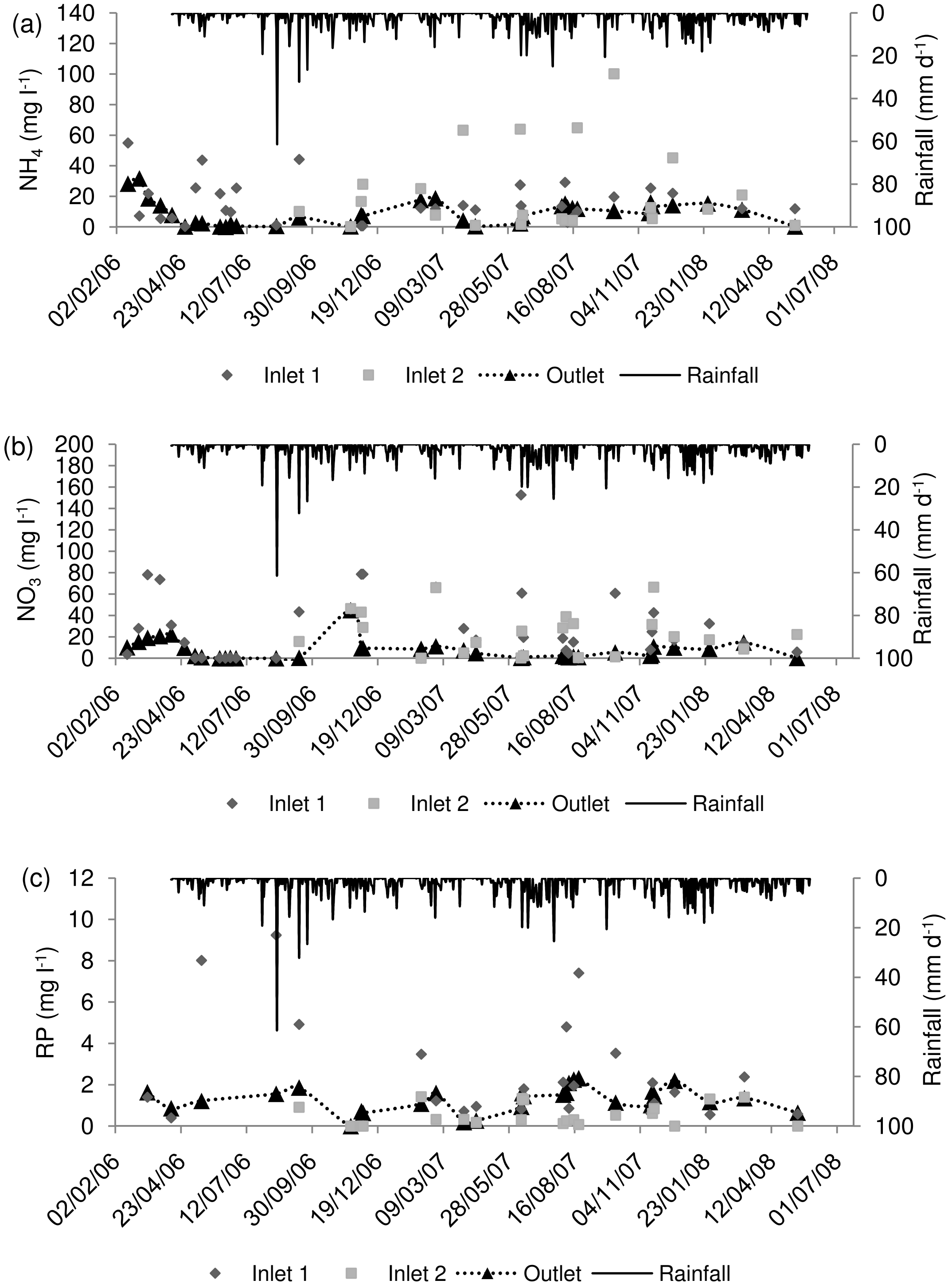
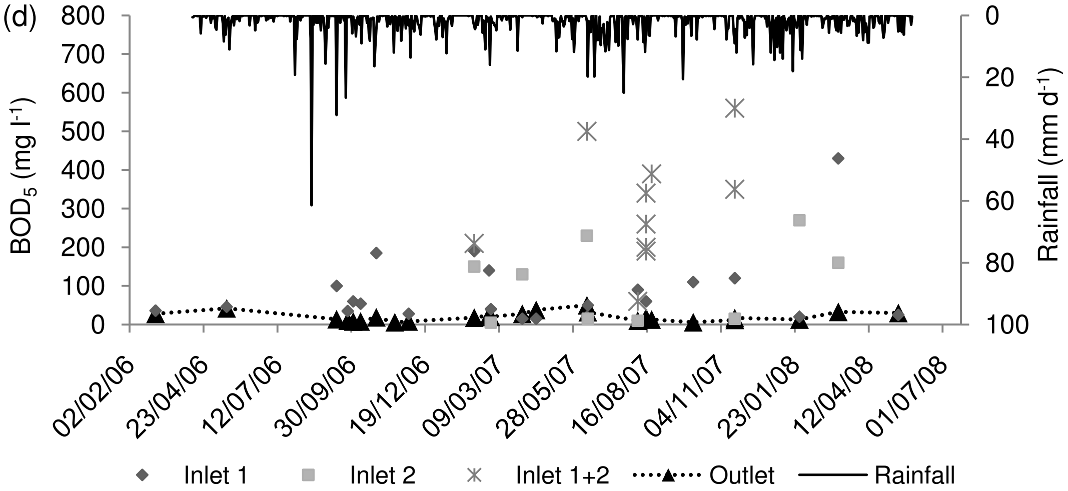


Figure 4.13 Variations in the concentration of (a) NH_4, (b) NO_3, (c) RP, (d) BOD_5 and (e) SS along CFW1. Boxes delimitate the lower and upper quartiles, horizontal bars indicate the medians, whiskers the minima and maxima, and dots the extreme values. Locations which do not share a capital letter have significantly different median concentrations (p< 0.05).

Long-term fluctuations in effluent water quality

As shown in Figure 4.14 (dashed lines indicates uncertainties between sampling dates due to the low sampling frequency), NH_4 concentration at the outlet varied between < 0.01 and 31.7 mg l^{-1}, was most of the time < 20 mg l^{-1} and was higher in autumn/winter and during the wet summer of 2007. Nitrate concentration fluctuated between < 0.017 and 45 mg l^{-1}, peaked in autumn/winter and was most of the time < 20 mg l^{-1}. RP varied between < 0.003 and 2 mg l^{-1} and remained high over long periods of time, mostly between 1 and 2 mg l^{-1}. BOD_5 ranged from < 1 to 50 mg l^{-1}, and mean BOD_5 was higher in spring/summer (28 mg l^{-1}) than in autumn/winter (16 mg l^{-1}), while at the pond inlet, it was higher in autumn/winter (86 mg l^{-1}) compared to spring/summer (63 mg l^{-1}). SS varied from 9.6 to 120 mg l^{-1} and mean SS was higher in summer (45 mg l^{-1}) than in autumn/winter (30 mg l^{-1}).





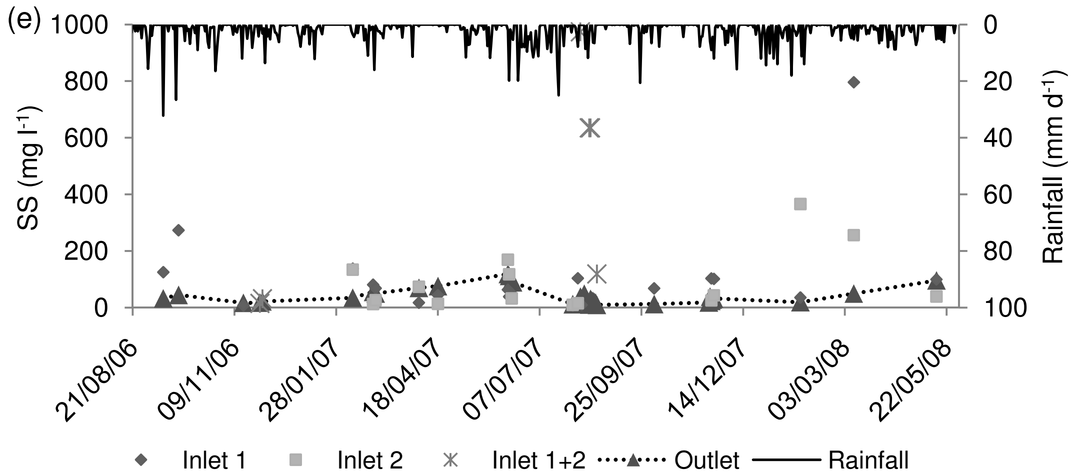


Figure 4.14 Daily rainfall and fluctuations of (a) NO_3, (b) NH_4, (c) RP, (d) BOD_5 and (e) SS concentration at inlet and outlet of CFW1 in grab samples.

Spatial water quality fluctuations on selected dates

From inlet to outlet at CFW1, different patterns in concentration fluctuations were observed, mainly influenced by sampling timing (e.g. summer/winter, dry/rainy period) (Figure 4.15). For example, a reduction in NO_3 occurred on 14/09/06 under dry antecedent conditions, and on 13/06/07 (outflow: 2.4 l s^{-1}) shortly after rainfall (limited inputs from field drainage at this stage), but not on 16/11/06 when sampling immediately followed heavy rain. On 14/09/06 NH_4 decreased, but increased on 28/02/07. A reduction in RP occurred on the 14/09/06 between inlet 1 and outlet. For all the pollutants, concentrations at outlet, pond inlet and within the pond were comparable, showing rapid water mixing.

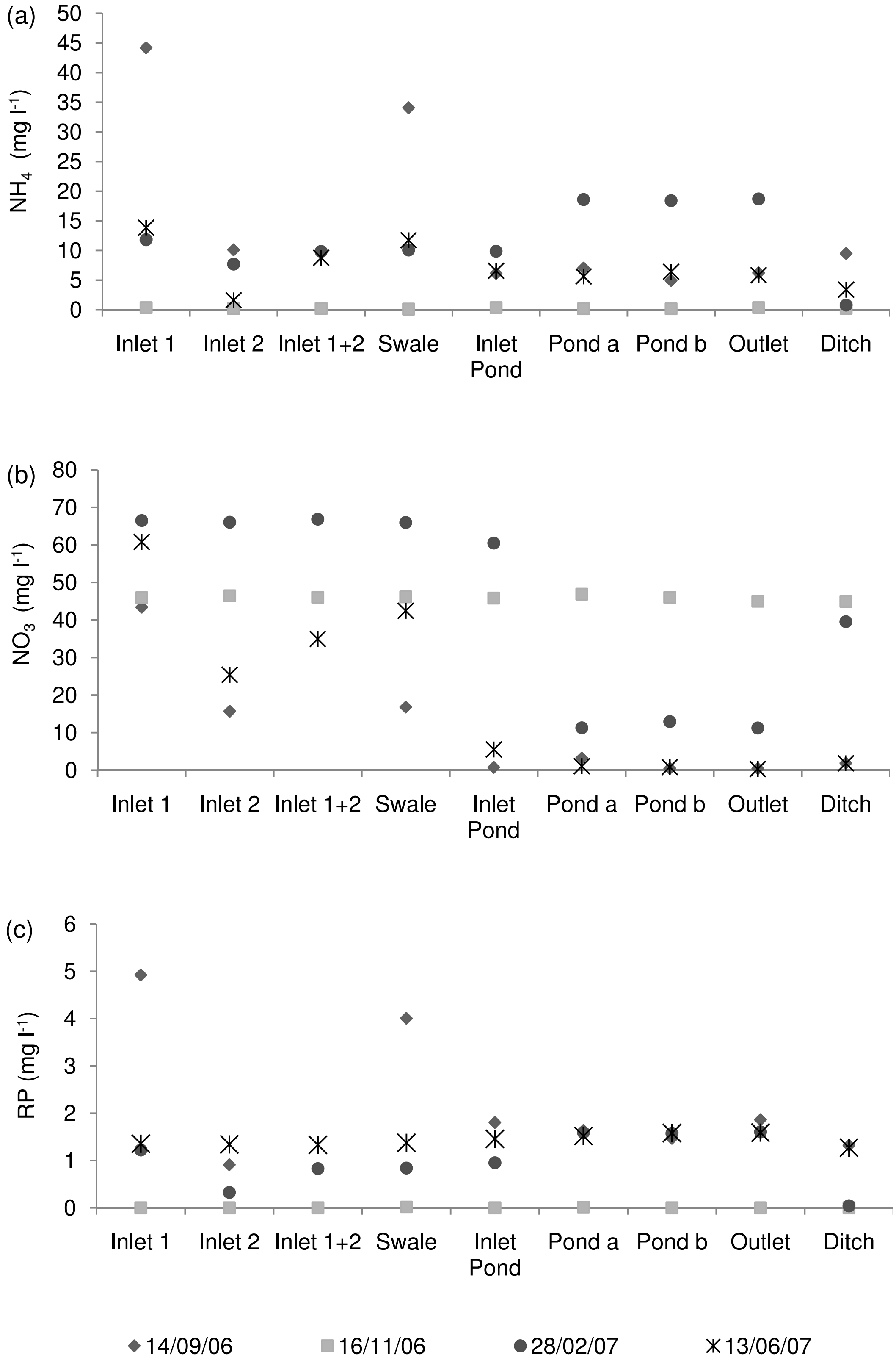


Figure 4.15 Concentration of (a) NH_4, (b) NO_3 and (c) RP from inlet to outlet of CFW1 on different dates.

Seasonal fluctuations in water quality

Table 4.5 gives the mean and median concentrations for different water quality parameters, for samples collected in autumn and winter (October to March) grouped as “AUTWIN”, and spring and summer (April to September) (“SPRSUM”) between the inlet and outlet of CFW1 and also in the ditch located downstream.

Mann Whitney tests and pairwise comparisons indicated significant differences between seasons for median NO_3 concentration at inlet 1 (U= 242, p= 0.05), inlet 1+2 (U= 9.15, p= 0.0025), inlet pond (U= 278, p= 0.003), outlet (U= 418, p= 0.001) and ditch (U= 59, p= 0.004), for median NH_4 concentration at inlet pond (U= 285, p= 0.001) and outlet (U= 412, p= 0.001), for BOD_5 at outlet only (U= 24.5, p= 0.015). No differences were detected for RP and SS at any of the locations.

NH_4 and NO_3 concentrations were higher in winter than in summer at all locations, which corresponds to a combination of higher N inputs from the fields and farmyard in winter and to a better treatment within the CFW in summer influenced by higher temperatures. Median outlet concentrations of $BOD_{5,}$ SS and RP to a lesser extent were higher in summer than in winter, which could be explained both by a smaller dilution (less rainfall and field drainage) and by the presence of algae, whose growth is enhanced in warmer months, since water samples were unfiltered before analysis.

In the ditch (200 m downstream from CFW1), concentrations of RP and NH_4 were higher in summer than in winter due to smaller dilution of farmyard runoff by field drainage, while NO_3 concentration was much higher in winter than in summer, due to larger quantities (fields are saturated over longer periods) of nitrate-rich (smaller nitrate uptake by grassland) field drainage.

Table 4.5 Seasonal differences in water quality at different sampling locations at CFW1 (n: number of samples; SE: standard error).

Location	Season		Water quality parameter				
			NH_4	NO_3	RP	BOD_5	SS
Inlet 1	AUTWIN	n	13	17	14	12	9
		Mean (mg l^{-1})	18.8	38.5	1.73	118	155
		SE	4.1	7.1	0.47	33	81.1
		Median (mg l^{-1})	13.4	32.4	1.31	85.0	80.9
	SPRSUM	n	18	21	15	10	11
		Mean (mg l^{-1})	14.3	20.0	3.06	46.0	75.4
		SE	3.2	7.5	0.78	10.0	23.1
		Median (mg l^{-1})	11.3	7.35	1.81	41.0	49.2
Inlet 2	AUTWIN	n	12	12	12	5	7
		Mean (mg l^{-1})	24.8	28.4	0.737	120	124
		SE	7.7	6.7	0.221	50	52
		Median (mg l^{-1})	18.8	24.5	0.569	150	43.5
	SPRSUM	n	11	11	11	4	8
		Mean (mg l^{-1})	20.7	16.9	0.464	96.0	58.9
		SE	8.4	4.1	0.145	52.0	20.4
		Median (mg l^{-1})	5.29	15.7	0.309	73.0	35.9
Inlet 1+2	AUTWIN	n	12	12	9	14	12
		Mean (mg l^{-1})	17.6	34.4	1.62	162	188
		SE	3.9	7.5	0.50	41	87.2
		Median (mg l^{-1})	15.9	32.0	2.01	125	66.1
	SPRSUM	n	17	17	10	17	16
		Mean (mg l^{-1})	15.8	13.4	1.46	150	243
		SE	3.1	5.47	0.40	35	70
		Median (mg l^{-1})	12.7	5.76	1.13	73	118
Inlet Pond	AUTWIN	n	17	17	14	11	11
		Mean (mg l^{-1})	17.7	25.3	1.68	93	130
		SE	3.4	5.50	0.30	36	62
		Median (mg l^{-1})	14.4	16.6	1.58	50	50.0
	SPRSUM	n	21	21	15	10	12
		Mean (mg l^{-1})	8.05	4.23	1.72	52	224
		SE	2.27	1.28	0.49	15	102
		Median (mg l^{-1})	6.15	0.990	1.40	30	85.5
Outlet	AUTWIN	n	20	20	17	14	14
		Mean (mg l^{-1})	14.6	11.4	1.28	16	29.9
		SE	1.7	2.09	0.13	2	3.7
		Median (mg l^{-1})	14.1	9.36	1.32	15	26.1
	SPRSUM	n	25	25	19	12	23
		Mean (mg l^{-1})	5.59	3.74	1.39	25	45.4
		SE	1.08	1.52	0.14	4	7.4
		Median (mg l^{-1})	3.80	1.05	1.52	23	30.8
Ditch	AUTWIN	n	9	9	9	-	-
		Mean (mg l^{-1})	4.82	34.4	0.574	-	-
		SE	3.21	6.59	0.416	-	-
		Median (mg l^{-1})	1.47	30.7	0.216	-	-
	SPRSUM	n	7	7	7	-	-
		Mean (mg l^{-1})	4.07	4.71	0.798	-	-
		SE	1.33	1.79	0.223	-	-
		Median (mg l^{-1})	3.39	1.91	1.06	-	-

- Data not available.

4.4.3.3 Water quality and hysteresis during storm events

Table 4.6 summarizes the main characteristics of the storm events studied at CFW1.

Table 4.6 Summary of the hydrological characteristics of the storm events investigated at CFW1 ("runoff": total inflow minus water volume to cool the milk).

Storm event No. Date/Time	Total rainfall Max. intensity	Inflow characteristics	Outflow volume (% total volume attenuation)	Notes
Storm event 1 10/02/07 20:00 to 13/02/07 03:00	8.6 mm 0.2 mm 5 min^{-1}	Max. 14 l s^{-1} Volume: 533 m^3 Runoff: 485 m^3 56 m^3 mm^{-1}	356 m^3 (33%)	Outflow volume underestimated due to sensor failure
Storm event 2 27/02/07 04:00 to 28/02/07 04:00	16.2 mm 0.4 mm 5 min^{-1}	Max. 40 l s^{-1} Volume: 726 m^3 Runoff: 702 m^3 43 m^3 mm^{-1}	730 m^3 (0%)	-
Storm event 3 12/06/07 21:30 to 13/06/07 14:00	21.2 mm; 1.0 mm 5 min^{-1}	Max. 43 l s^{-1} Volume: 325 m^3 Runoff: 313 m^3 15 m^3 mm^{-1}	110 m^3 (66%)	Water level below outlet level at start (storage: 80 m^3)
Storm event 4 15/06/07 12:00 to 16/06/07 16:00	8.6 mm 0.2 mm 5 min^{-1}	Max. 10 l s^{-1} Volume: 144 m^3 Runoff: 120 m^3 14 m^3 mm^{-1}	110 m^3 (24%)	-
Storm event 5 5/08/07 to 6/08/07	13.8 mm 1.5 mm 5 min^{-1}	Max. 23 l s^{-1} Volume: 194 m^3 Runoff: 170 m^3 12 m^3 mm^{-1}	155 m3 (20%)	-
Storm event 6 14/08/07 16:00 to 15/08/07 16:00	12.6 mm 0.8 mm 5 min^{-1}	Max. 15 l s^{-1} Volume: 245 m^3 Runoff: 221 m^3 18 m^3 mm^{-1}	188 m^3 (23%)	-
Storm event 7 18/11/07 2:00 to 21/11/07 13:00	31.2 mm 0.4 mm 5 min^{-1}	Max. 13 l s^{-1} Volume: 761 m^3 Runoff: 667 m^3 22 m^3 mm^{-1}	627 m^3 (18%)	Two-stage event. Long duration and large rainfall

The results obtained for storm events 1, 3, 5, 6 and 7 are presented and illustrated in more detail below to illustrate the key findings.

Storm event 1: 10th - 13th February 2007

The first stage of the storm event (Figure 4.16) was characterised by a rise in flow shortly (c. 15 min) after rainfall started, and by an increase in NH_4 (up to 27 mg l^{-1}) and RP (up to 5 mg l^{-1}) accompanied by a decrease in NO_3 concentration (from 20 mg l^{-1} to < 0.01 mg l^{-1}), which may correspond to the dominance of farmyard runoff over deep groundwater and field drainage at an early stage of the event. Later on, a decrease in NH_4 and RP and an increase in NO_3 were observed, probably explained by inputs from the fields. On 11th February, NH_4 concentration was lower in the sample (diluted by groundwater) taken during milking time (4-8 pm).

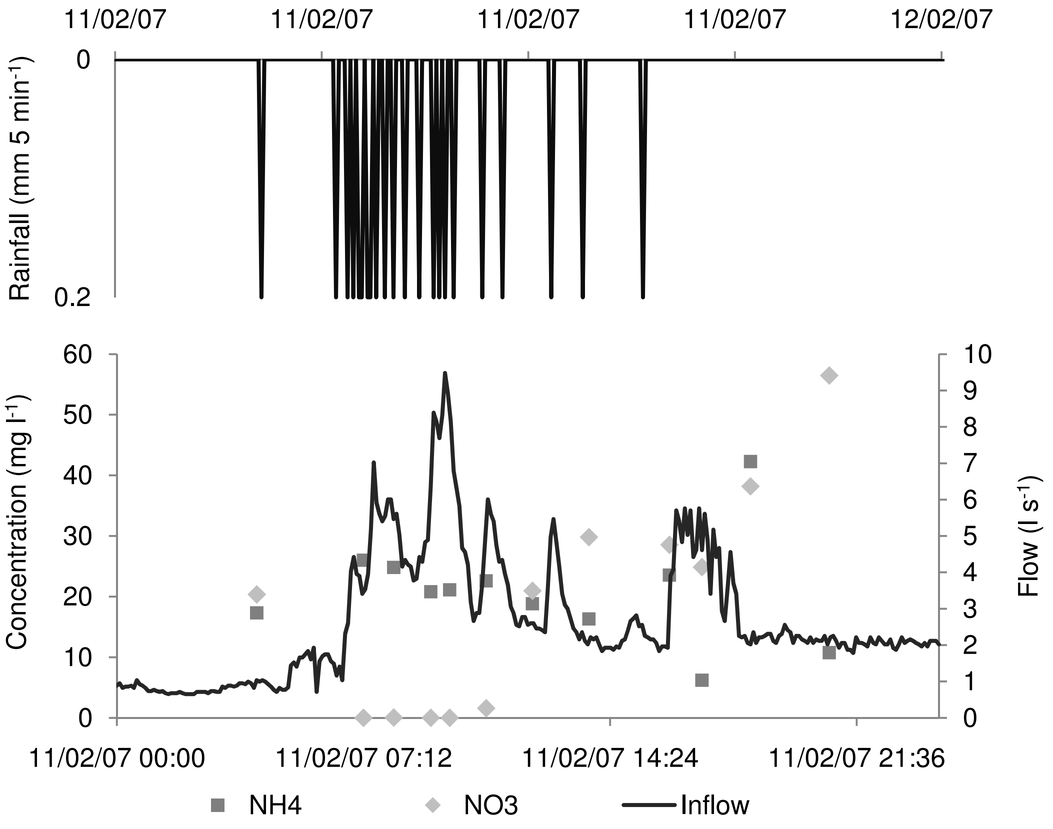


Figure 4.16 Concentration of NH_4 and NO_3 and flow at the inlet of CFW1 during a storm event in February 2007.

At the inlet, a flush of RP at the beginning of the event and progressive decrease in concentration were observed (Figure 4.17), which could correspond to source exhaustion on the farmyard or dilution by field drainage.

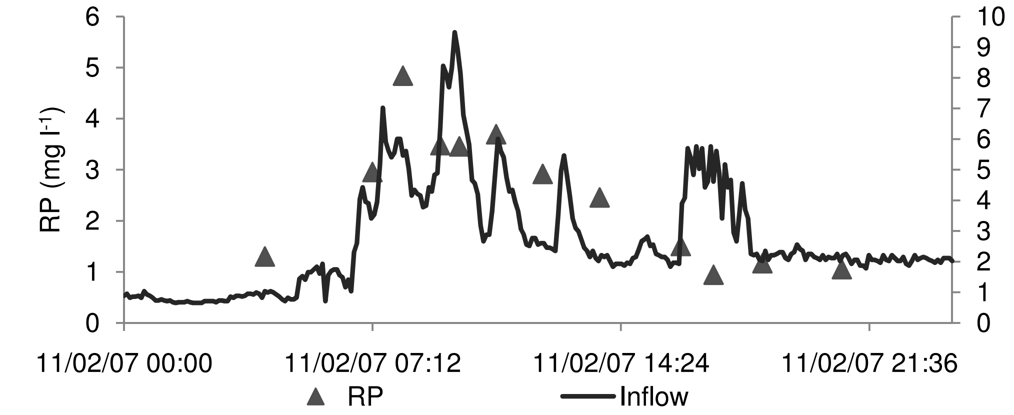


Figure 4.17 Concentration of RP and flow at the inlet of CFW1.

At the outlet (Figure 4.18), no significant concentration increase was observed in response to the storm event and no relation between flow and concentration could be identified, which suggests a strong dilution effect and mixing within the pond. Mean concentration reduction efficiency (± standard error) between inlet 1+2 and outlet for this event was 11% (± 11%) for NH_4, 51% (± 15%) for NO_3, and 45% (± 8%) for RP, and mean flux reduction efficiency was 56% (± 10%) for NH_4, 63% (± 12%) for NO_3 and 77% (± 7%) for RP.

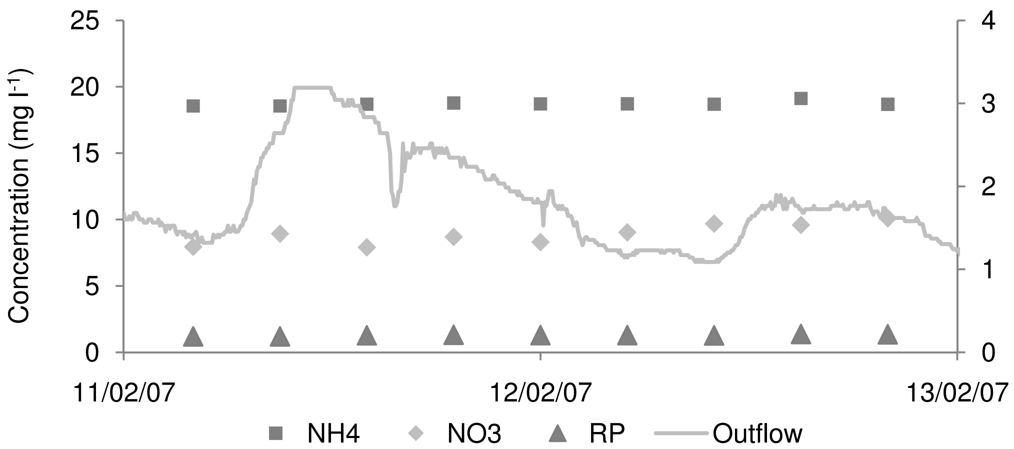


Figure 4.18 Concentration of NH_4, NO_3 and RP, and flow at the outlet of CFW1 during a storm event in February 2007.

Storm event 3: 12th - 13th June 2007

In contrast with Event 1, rainfall amount (21 mm) and intensity (up to 1 mm 5 min^{-1}) were higher, which triggered a larger flow response (Figure 4.19). The outflow only started around 1 am on 13th June, due to level in the pond being 3 to 4 cm below the pipe at the start of the storm event, giving c. 80 to 90 m^3 storage.

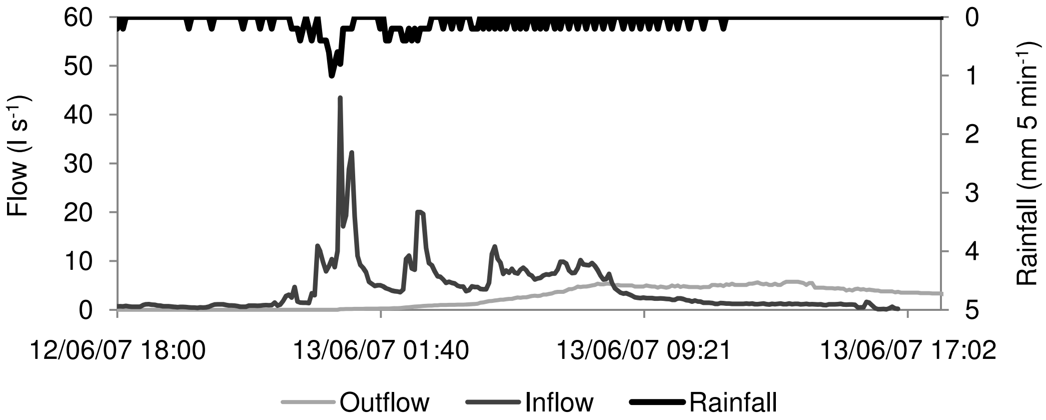


Figure 4.19 Fluctuations in the inflow and outflow during a storm event at CFW1 in June 2007.

Only the first part of the event was sampled due to clogging of the suction line with straw and plastic after bottle 12. At the inlet, a drop in NH_4 concentration was visible at higher flow, due to dilution and maybe source exhaustion, i.e. reduction of the quantities of faeces and urine available for mobilisation (Figure 4.20).

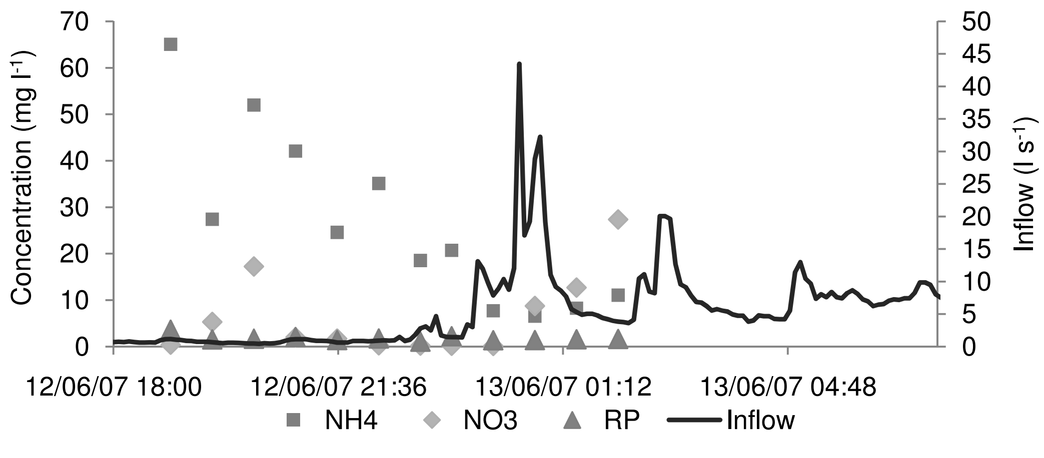


Figure 4.20 Concentration of NH_4, NO_3 and RP and flow at the inlet of CFW1 during a storm event in June 2007.

The concentration of all pollutants at outlet was lower than at inlet and rose only slightly, up to 5.6 mg l^{-1} for NH_4, 2.8 mg l^{-1} for NO_3 and 1.3 mg l^{-1} for RP (Figure 4.21).

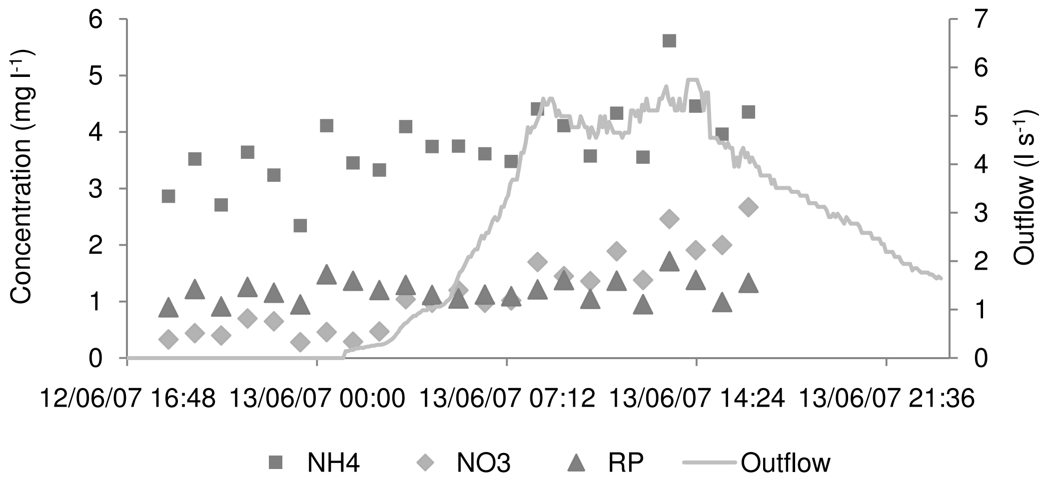


Figure 4.21 Concentration of NH_4, NO_3 and RP and flow at the outlet of CFW1 during a storm event in June 2007.

Mean concentration reduction efficiency (± standard error) between inlet 1+2 and outlet was 86% (± 3%) for NH_4, 82% (± 8%) for NO_3, and 32% (± 8%) for RP, while mean flux reduction efficiency was 82% (± 5%) for NH_4, 88% (± 7%) for NO_3 and 58% (± 17%) for RP.

Storm event 5: 3rd - 13th August 2007

Only a very small part of Event 5 was monitored due to the early start of the sampling cycle (incorrect estimation of the start of the rainfall) at inlet and late start at outlet (incorrect setting of the sampler). Flow response was very small, due to low precipitation and/or underestimation by the sensor. Concentrations of NH_4, NO_3 and RP at the inlet were relatively high, even during dry periods, due to input from septic tank and groundwater (Figure 4.22). During the light rainfall, NH_4 concentration increased while NO_3 concentration decreases. Total phosphorus concentration was only slightly higher than RP concentration.

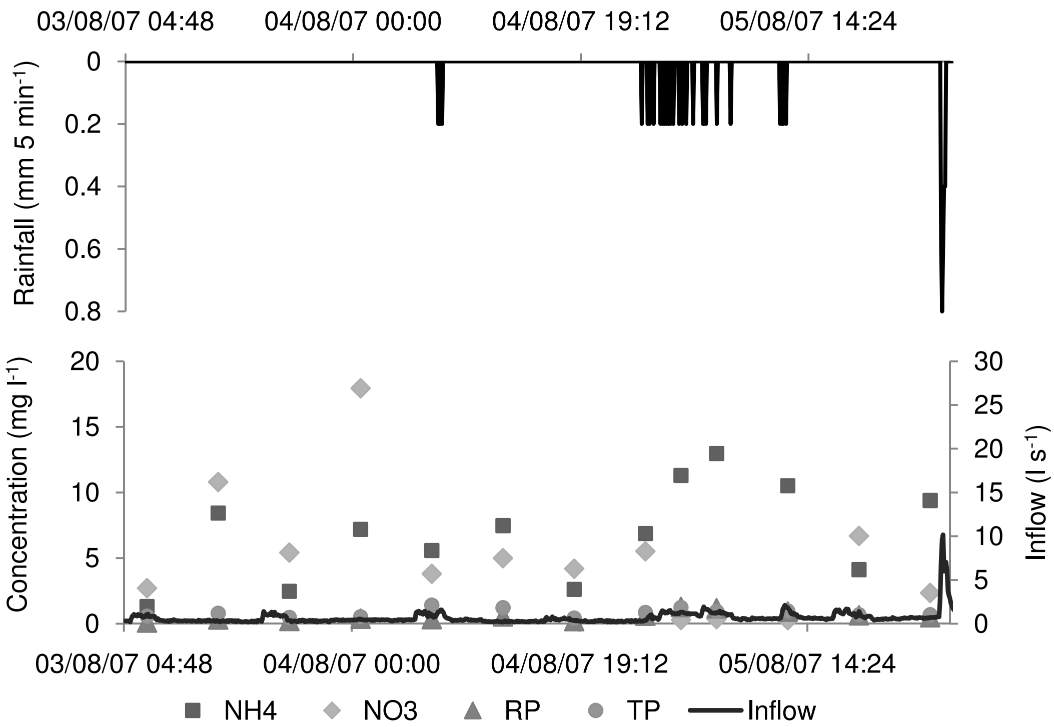


Figure 4.22 Concentration of NH_4, NO_3 and RP and flow at the inlet of CFW1 during a small storm event in August 2007.

Outlet concentrations were constant (Figure 4.23), except for SS, which dropped from 40 mg l^{-1} to 20 mg l^{-1}, i.e. a 50% reduction in 48 h, illustrating the rapid settling of sediment.

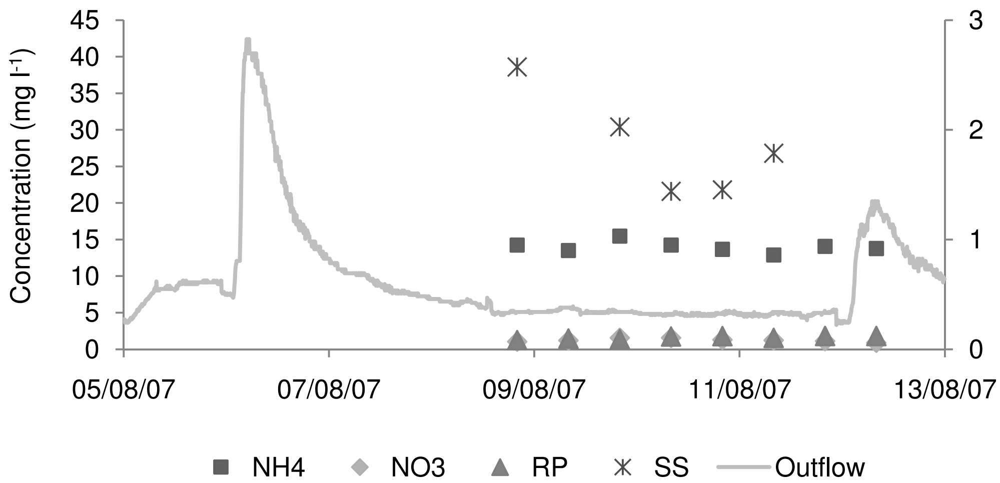


Figure 4.23 Concentration of NH_4, NO_3 and RP and flow at the outlet of CFW1 during a storm event in August 2007.

Storm event 6: 15th - 20th August 2007

NH_4 concentration at inlet reached high levels at the beginning of the monitoring period (c. 30 mg l^{-1}), before the increase in flow, which could correspond to septic tank input (also corresponds with the increase in RP), and decreased later on during the period of higher flows due to dilution. NO_3 concentration decreased at the beginning (to < 0.1 mg l^{-1}) due to the inflow source changing from groundwater to farmyard runoff, and increased later on, when field inputs were dominant (Figure 4.24). Reactive phosphorus and TP concentrations reached about 5 and 6 mg l^{-1} and decreased at the end of the event, due to source exhaustion or to limited mobilisation (lower rainfall intensity).

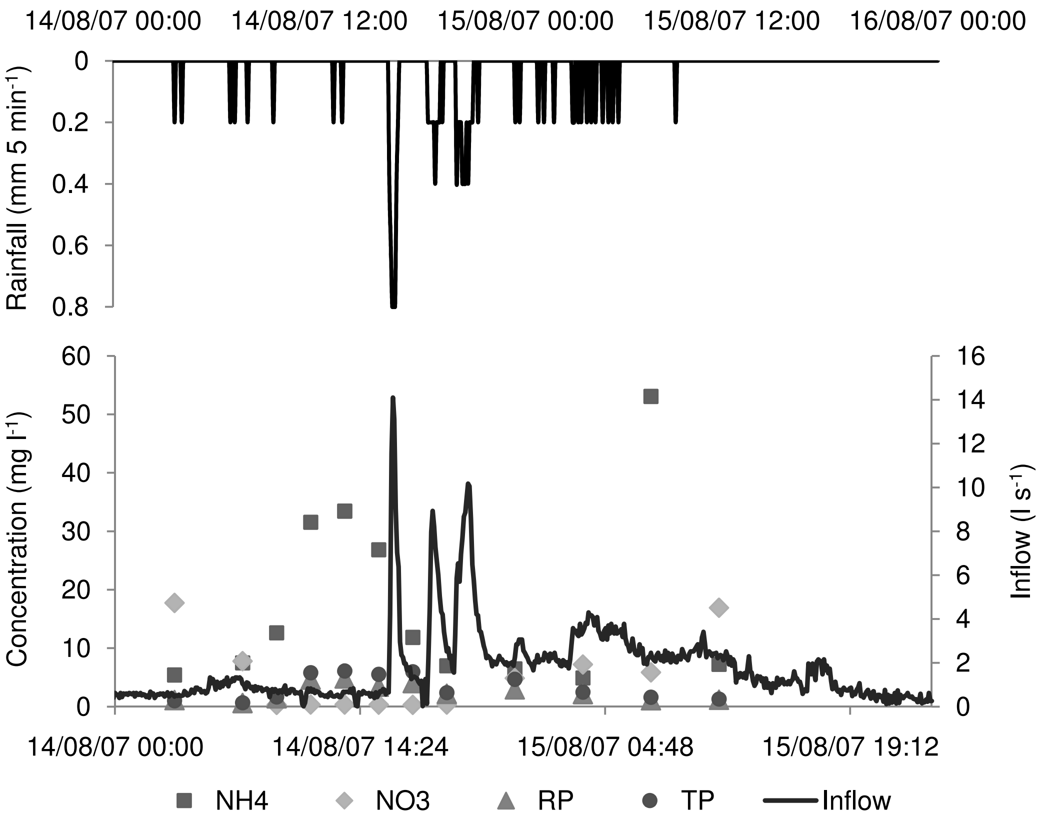


Figure 4.24 Concentration of NH_4, NO_3, RP and TP and flow at the inlet of CFW1 during a storm event in August 2007.

Outflow concentrations of NH_4, NO_3 and RP stayed relatively constant throughout the event, suggesting dilution and mixing within the pond (Figure 4.25), but no short-term treatment. However, SS decreased from 31 to 17 mg l^{-1} (45% reduction in 96 h).

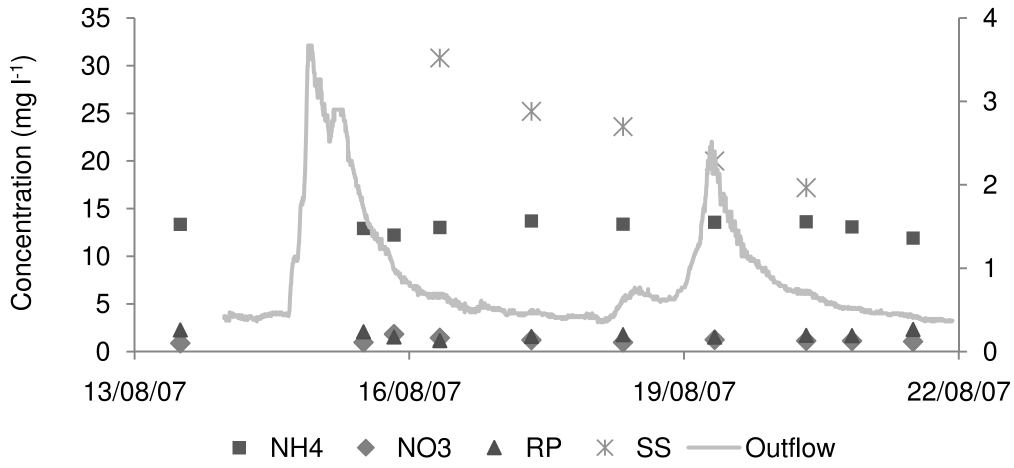


Figure 4.25 Concentration of NH_4, NO_3, RP and SS, and flow at the outlet of CFW1 during a storm event in August 2007.

Storm event 7: 17th - 21st November 2007

This event was characterized by the largest rainfall (31.2 mm) and largest runoff volume, estimated to be around 670 m^3, but inflow stayed below 18 l s^{-1}, due to the small rainfall intensity (Figure 4.26).

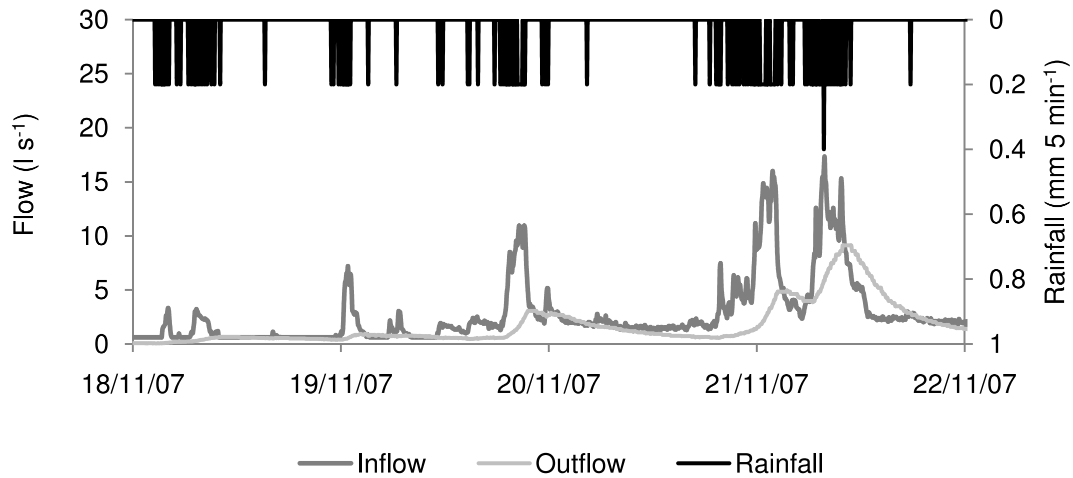


Figure 4.26 Fluctuations in the inflow and outflow during a storm event at CFW1 in November 2007.

Considerable variability in concentrations of NH_4 (7.8 to 30 mg l^{-1}), NO_3 (< 0.017 to 30 mg l^{-1}), RP (< 0.003 to 4 mg l^{-1}) and TP (< 0.01 to 5 mg l^{-1}) occurred during the event (Figure 4.27). RP and TP concentrations were higher at the beginning of the rainy period, when flow was between 3 and 4 l s^{-1}.

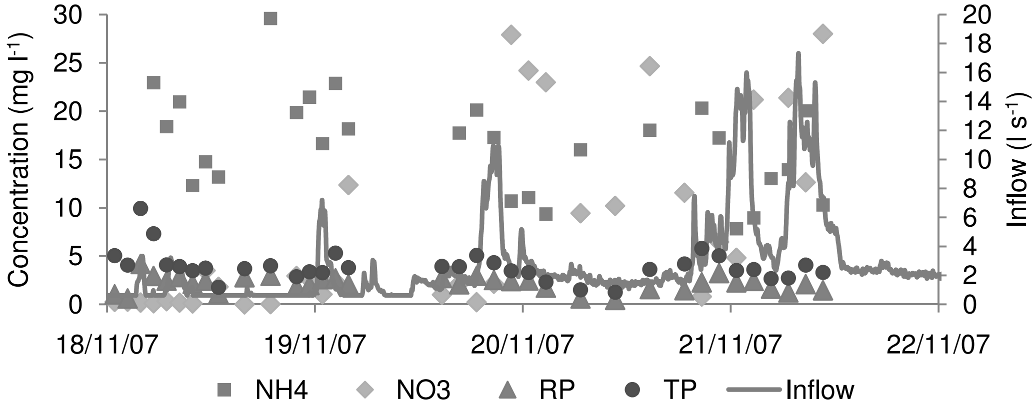


Figure 4.27 Concentration of NH_4, NO_3, RP and TP and flow at the inlet of CFW1 during a storm event in November 2007.

No significant change in outflow NH_4 or RP concentrations occurred (Figure 4.28), but NO_3 concentration increased at the end of the period and reached 11 mg l^{-1}, maybe due to large field inputs during the second part of the event. Mean concentration reduction efficiency (± SE) between inlet 1+2 and outlet was 39% (± 7%) for NH_4, 58% (± 12%) for NO_3, and 41% (± 6%) for RP, and mean flux reduction efficiency was 70% (± 10%) for NH_4, 87% (± 7%) for NO_3 and 76% (± 9%) for RP.

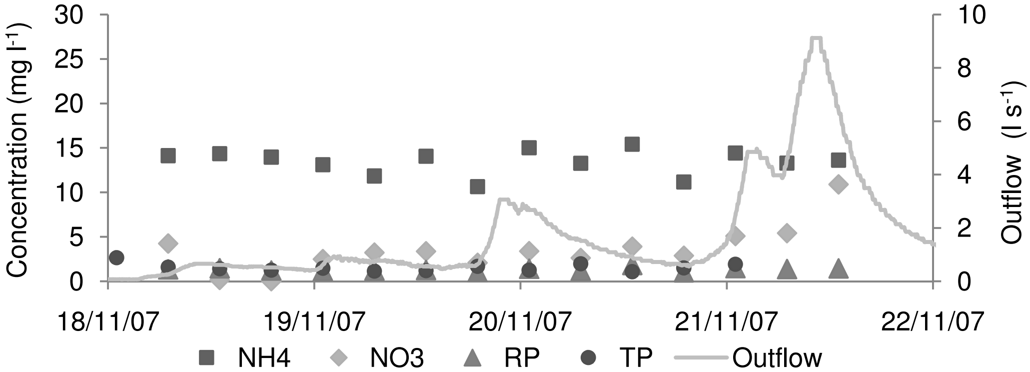


Figure 4.28 Concentration of NH_4, NO_3, RP and TP, and flow at the outlet of CFW1 during a storm event in November 2007.

4.4.3.4 Influence of antecedent rainfall on water quality

Correlation analyses were carried out to examine the influence of antecedent rainfall (defined as the total amount of rainfall before a given sampling date) on inflow and outflow quality. Several antecedent rainfalls were tested, but 2-day AR at inlet and 5-day AR at outlet (which give sufficient time for the water to flow through the ponds) shown some significant trends.

Since NH_4, BOD_5 and RP concentrations at the inlet were mainly controlled by farmyard runoff, an inverse relationship between concentration and AR was expected at the inlet due to the accumulation of contaminants (e.g. faeces, urine) on impervious surfaces between storm events. However, a positive relationship could be expected at the inlet for NO_3 because a high AR will tend to saturate the fields and trigger field drainage input, rich in NO_3. At the outlet, larger AR was expected to cause higher concentrations of all pollutants, due to more contaminants being flushed, although this effect could also be masked by the dilution of pond water by roof runoff and rainfall.

No correlations existed between NH_4 concentration and 2-day AR at inlet 1 and inlet 2, which could be due to the daily contamination of the farmyard by fresh faces and its permanent "dirty state" (i.e. rainfall rarely washes out all the faeces). A significant positive correlation existed between 5-day AR and NH_4 concentration at outlet (r_s= 0.45, p= 0.0046) which indicates more inputs during rainy periods. A significant positive correlation between NO_3 at inlet 1 and 2-day AR existed (r_s= 0.35, p= 0.05), but none existed between NO_3 concentration at outlet and 5-day AR, due to NO_3 entering the pond everyday with the water used to cool the milk. Reactive phosphorus concentrations at inlet 2 (field) and outlet were positively correlated to 2-day AR (r_s= 0.48, p= 0.03) and 5-day AR (r_s= 0.51, p= 0.0025) respectively. Figures 4.29, 4.30 and 4.31 illustrate the relationships between concentrations of pollutants and antecedent rainfall at inlet and outlet.

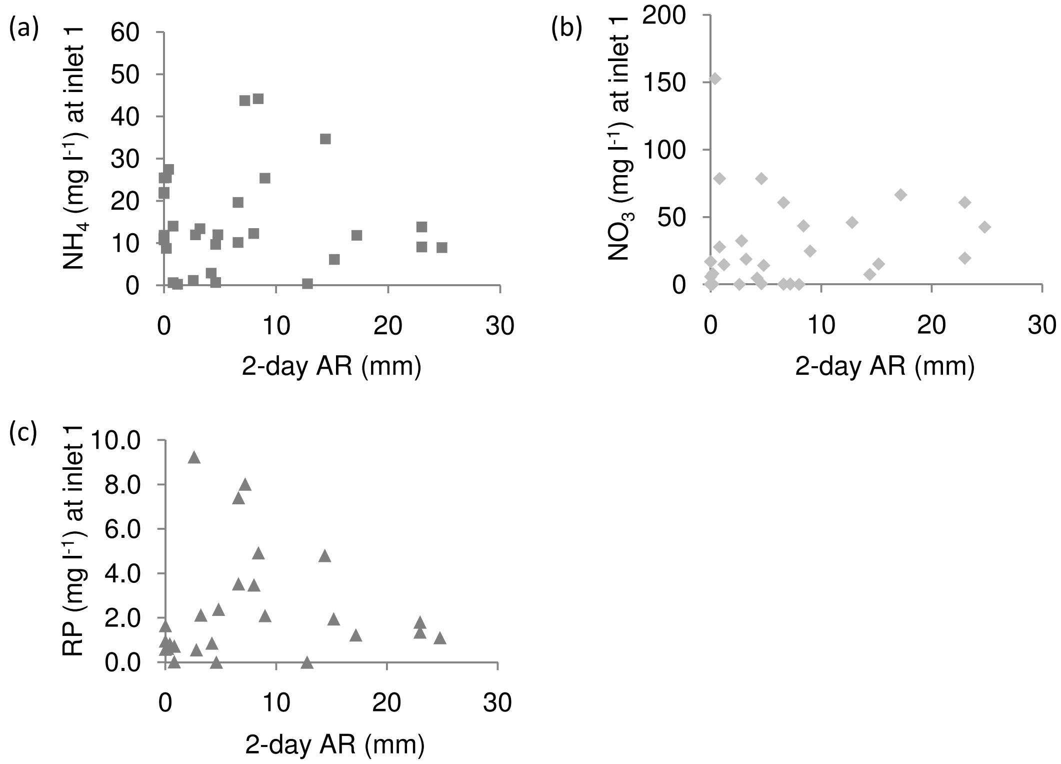


Figure 4.29 Concentration of (a) NH_4, (b) NO_3, and (c) RP at inlet 1 as a function of 2-day antecedent rainfall (AR), CFW1.

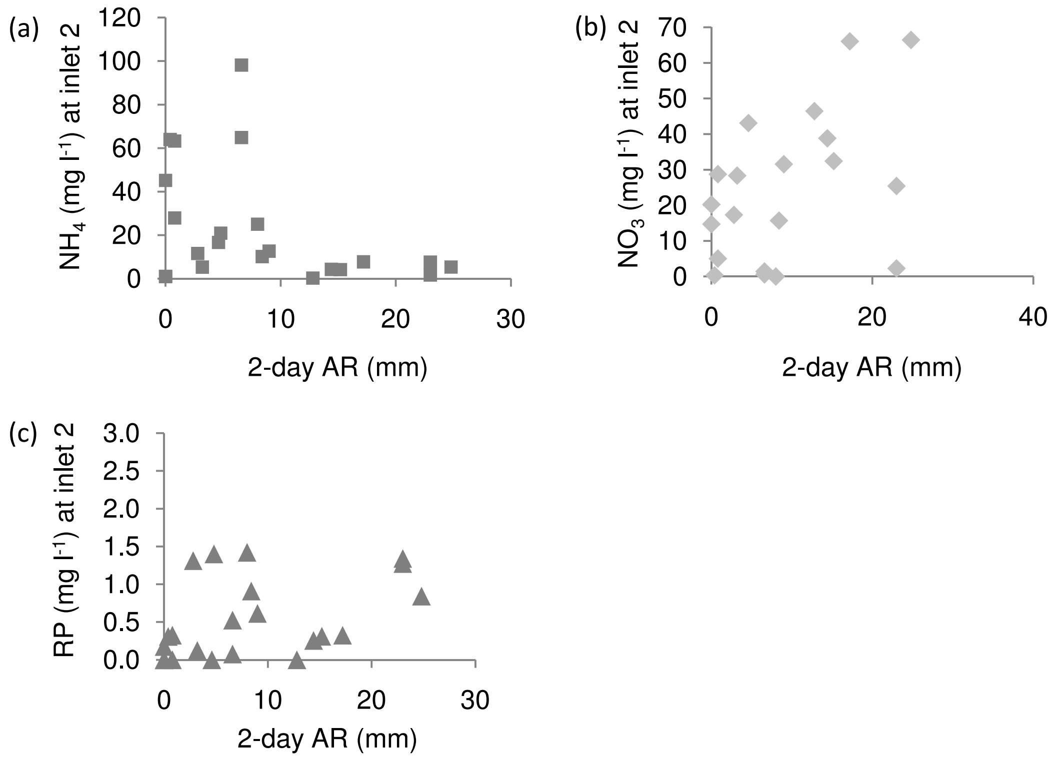


Figure 4.30 Concentration of (a) NH_4, (b) NO_3 and (c) RP at inlet 2, as a function of 2-day antecedent rainfall (AR), CFW1.

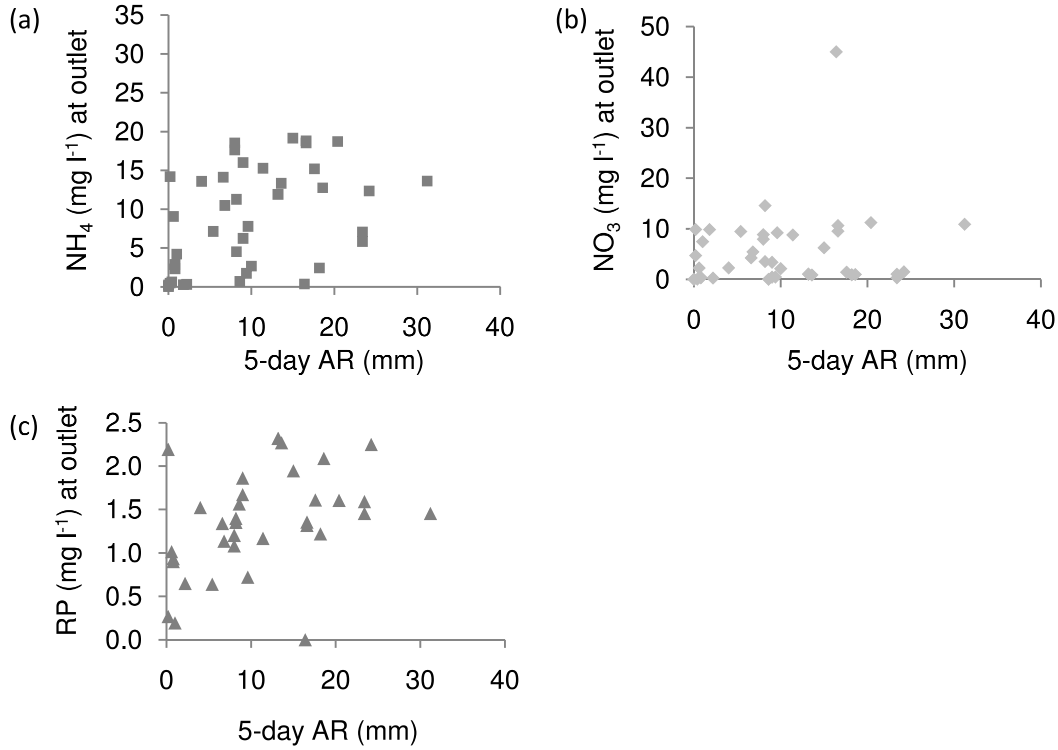


Figure 4.31 Concentration of (a) NH_4, (b) NO_3 and (c) RP at outlet as a function of 5-day antecedent rainfall (AR), CFW1.

4.4.3.5 Treatment efficiency by concentration

An estimate of the treatment efficiency by mean and median concentration between inlet (samples taken at the weir) or pond inlet and outlet using grab and storm samples is shown in Table 4.7 for the different pollutants. Large differences existed depending on whether efficiency was expressed using the mean or median concentration and whether it was calculated between inlet 1+2 and outlet or between pond inlet and outlet. The latter was usually lower, except for RP whose mean concentration was slightly higher close to pond inlet than at the inlet of the swale.

Overall, maximum concentration reduction efficiencies were for faecal coliforms (94% - only three samples analysed), BOD_5 (87%), SS (86%), followed by NO_3 (68%), faecal *streptococci* (45%), NH_4 (42%) and RP (13%). Average removal by median concentration was lower than by mean concentration.

Table 4.7 Mean and median concentration reduction efficiency between inlet 1+2 and outlet, and inlet pond and outlet at CFW1 (n: number of samples; ± standard error in brackets).

Water quality parameter		Sampling location			Concentration Reduction Efficiency (%)	
		Inlet 1+2	Inlet Pond	Outlet	Inlet 1+2 to Outlet	Inlet Pond to Outlet
NH_4 (mg l^{-1})	n	29	38	45		
	Mean	16.6 (± 2.4)	12.4 (± 2.1)	9.58 (± 1.15)	42 (± 11)	23 (± 16)
	Median	14.0	8.94	9.06	35	< 0
NO_3 (mg l^{-1})	n	29	38	45		
	Mean	22.1 (± 4.8)	13.7 (± 3.1)	7.13 (± 1.36)	68 (± 9)	48 (± 16)
	Median	10.3	5.1	3.54	66	36
RP (mg l^{-1})	n	19	29	36		
	Mean	1.53 (± 0.31)	1.70 (± 0.30)	1.34 (± 0.09)	12 (± 19)	21 (± 15)
	Median	1.33	1.5	1.42	< 0	7
BOD_5 (mg l^{-1})	n	31.0	21	26		
	Mean	155 (± 26)	73.7 (± 20.1)	19.8 (± 2.4)	87 (± 2)	73 (± 8)
	Median	105	30	18	83	40
SS (mg l^{-1})	n	38	23	34		
	Mean	281 (± 65)	179 (± 61)	40.6 (± 5.4)	86 (± 4)	77 (± 9)
	Median	118	65.4	30.5	74	53
FC (cfu 100 ml^{-1})	n	3	3	3		
	Mean	72 500 (± 41 400)	71 700 (± 40 800)	4450 (± 39110)	94 (± 7)	94 (± 7)
FS (cfu 100 ml^{-1})	n	3	3	3		
	Mean	5900 (± 2630)	8320 (± 2360)	3230 (± 1810)	45 (± 40)	61 (± 24)

Treatment efficiency also varied significantly between seasons (Table 4.8), and was c. 4 times higher in summer than in winter for NH_4, similar for NO_3 and 4 times higher in autumn/winter for RP, while it was slightly lower for BOD_5 and SS.

Table 4.8 Seasonal differences in mean concentration reduction efficiency between inlet 1+2 and outlet at CFW1 ($n_{inlet\ 1+2}$ and n_{out}: number of samples at inlet 1+2 and outlet respectively; ± standard error in brackets).

Season		Water quality parameter				
		NH_4	NO_3	RP	BOD_5	SS
Autumn Winter	$n_{inlet\ 1+2}$ / n_{out}	12 / 20	12 / 20	9 / 17	14 / 14	12 / 14
	Mean concentration reduction efficiency (%)	17 (± 21)	67 (± 10)	21 (± 26)	90 (± 3)	84 (± 8)
Spring Summer	$n_{inlet\ 1+2}$ / n_{out}	17 / 25	17 / 25	10 / 19	17 / 12	16 / 23
	Mean concentration reduction efficiency (%)	65 (± 10)	72 (± 17)	5 (± 28)	83 (± 5)	81 (± 6)

4.4.3.6 Treatment efficiency by mass

To assess mass loadings in the long-term by interpolating concentration when flow was known, it was first investigated whether there were any correlations and relationships between concentrations of NH_4, NO_3 and RP measured at inlet and outlet during grab and storm sampling, and flow. Attempts were made to transform concentration data but were unsuccessful in normalizing the data. Figure 4.32 below illustrates the lack of relationship between concentrations (untransformed) and flow at inlet 1+2 ($R^2 < 0.1$), which might be explained by the fact that several sources are mixed, which have different chemical characteristics, and also respond differently to rainfall or do not depend from it. For example, low flows might correspond to septic tank overflow with relatively high concentrations, but could also occur at the end of a storm event and contain smaller amounts of pollutants. Nevertheless, there appeared to be some dilution effect at higher flows.

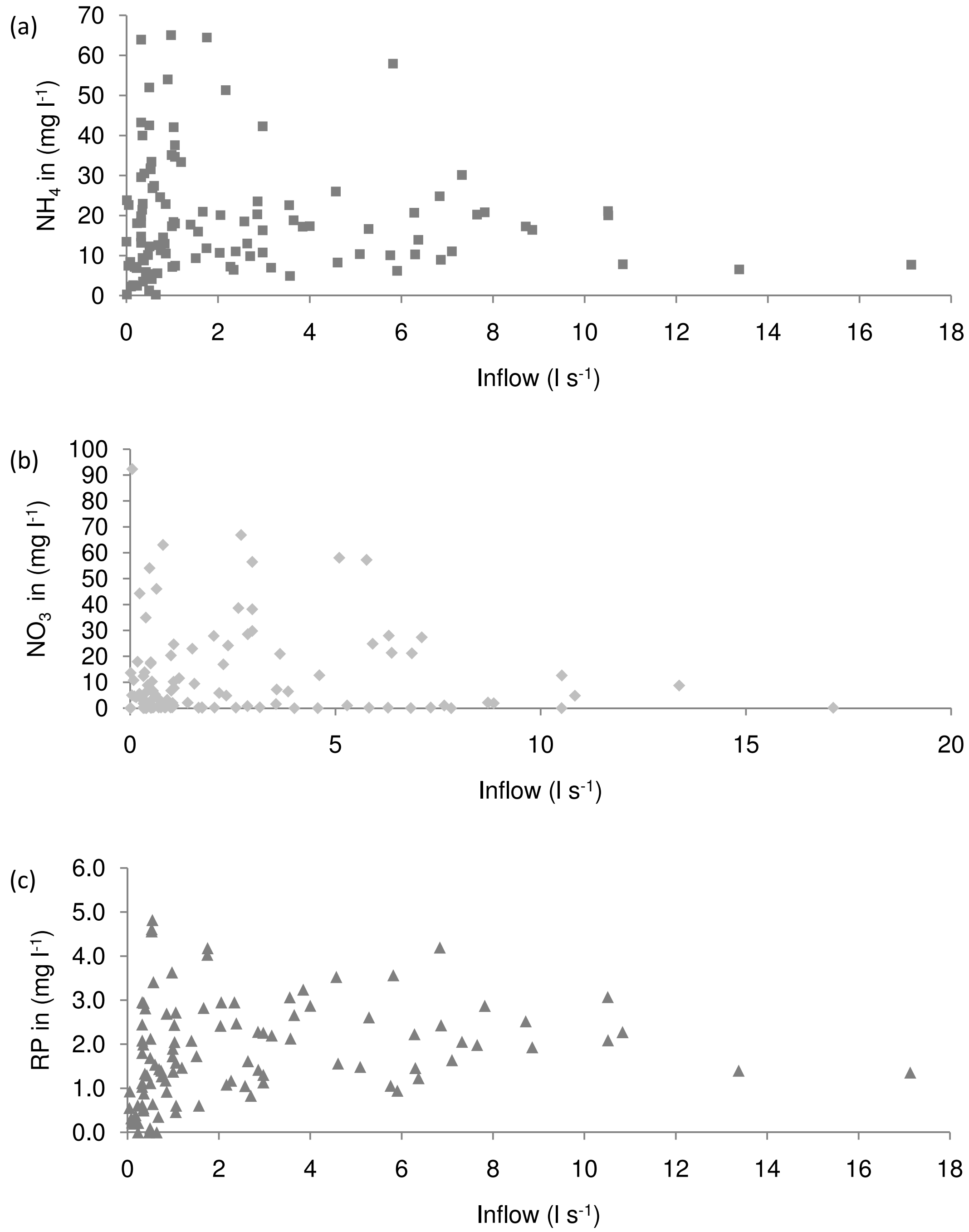


Figure 4.32 Relationships at inlet 1+2 between the concentration of (a) NH_4, (b) NO_3 and (c) RP and inflow, CFW1.

Figure 4.33 illustrates the lack of systematic significant relationships between concentrations and flow at outlet (R^2< 0.1), which could be because the pond had little impact on concentration (small volume) and outflow was changing very quickly (small volume and large outlet), at a much shorter scale than pollutants concentrations.

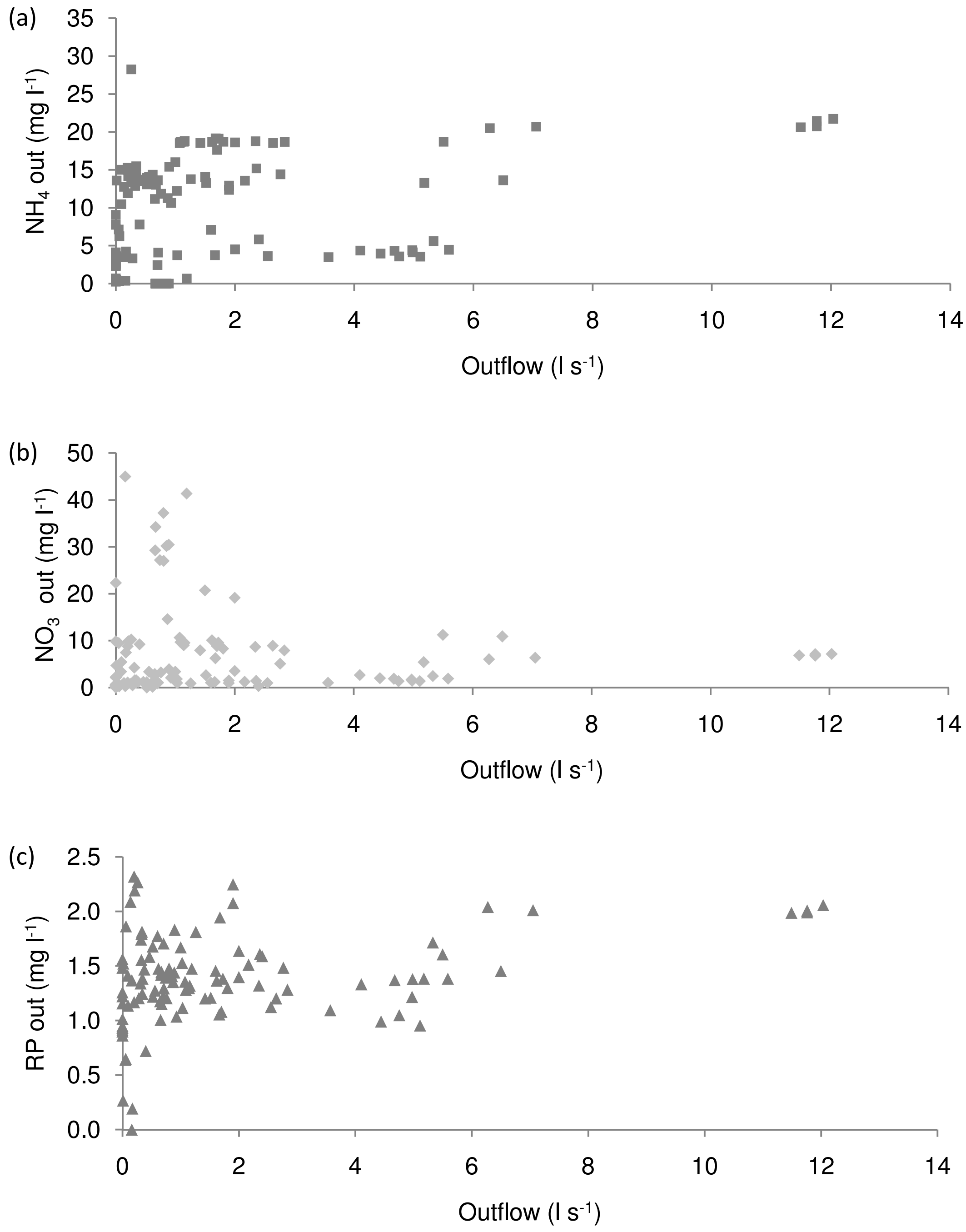


Figure 4.33 Relationships at the outlet between the concentration of (a) NH_4, (b) NO_3 and (c) RP and outflow, CFW1.

Since no relationships existed between concentration and flow, the interpolation approach could not be used to assess long-term loadings. Consequently, efficiency by mass was assessed using mean concentrations of pollutants at inlet and outlet.

Treatment efficiency by mass using long-term flow data and mean concentrations of pollutants:

Treatment efficiencies and mass removal per surface area for the different pollutants between inlet 1+2 and outlet are shown in Table 4.9, calculated considering a mean overall daily inflow of 98 m^3 d^{-1} and an outflow of 62 m^3 d^{-1} (c. 65% of the input).

Table 4.9 Mean mass reduction efficiency between inlet 1+2 and outlet at CFW1 using mean concentrations at inlet and outlet (± standard error in brackets).

	Water quality parameter				
	NH_4	NO_3	RP	BOD_5	SS
Mean conc. in (mg l^{-1})	16.6 (± 2.4)	22.1 (± 4.8)	1.53 (± 0.31)	155 (± 26)	281 (± 65)
Mass in (g d^{-1})	1630 (± 235)	2170 (± 470)	150 (± 30)	15 200 (± 2550)	27 500 (± 6330)
Mean conc. out (mg l^{-1})	9.58 (± 1.15)	7.13 (± 1.36)	1.34 (± 0.09)	19.8 (± 2.4)	40.6 (± 5.4)
Mass out (g d^{-1})	594 (± 71)	442 (± 84)	83.1 (± 5.6)	1230 (± 149)	2520 (± 335)
Mass out (kg yr^{-1})	217 (± 26)	161 (± 31)	30.3 (± 2.0)	448 (± 54)	919 (± 122)
Mass intercepted (g d^{-1})	1036 (± 245)	1720 (± 478)	66.8 (± 31.0)	13 960 (± 2550)	25 030 (± 6340)
Daily mass intercepted per m^2 of wetland (g m^{-2} d^{-1})	0.640 (± 0.110)	0.783 (± 0.220)	0.030 (± 0.010)	6.35 (± 1.16)	11.4 (± 2.9)
Mass reduction efficiency (%)	64 (± 7)	80 (± 6)	45 (± 12)	92 (± 2)	91 (± 3)

The highest treatment efficiency by mass was for BOD_5 (92%), followed by SS (91%), NO_3 (80%), NH_4 (64%) and RP (45%).

Following the same approach, mass reduction efficiency between pond inlet and outlet was lower than between inlet 1+2 and outlet, except for RP, due to dilution effects. Highest efficiency was obtained for SS (86%), followed by BOD_5 (84%), NO_3 (68%), RP (54%) and NH_4 (52%). Efficiency by mass was higher than by concentration for all pollutants.

Using mean concentrations of pollutants in samples taken in spring/summer and autumn/winter, and following the same calculations as above, an estimation of seasonal fluxes and efficiencies are given in Table 4.10, using a mean inflow of 95 $m^3\ d^{-1}$ and mean outflow of 60 $m^3\ d^{-1}$ in spring/summer, and mean inflow of 105 $m^3\ d^{-1}$ and mean outflow of 67 $m^3\ d^{-1}$ in autumn/winter.

Table 4.10 Seasonal pollutant fluxes and differences in mass reduction efficiency between inlet 1+2 and outlet at CFW1 ($n_{inlet\ 1+2}$ and n_{out}: number of samples at inlet 1+2 and outlet respectively; ± standard error in brackets).

Season		Water quality parameter				
		NH_4	NO_3	RP	BOD_5	SS
Autumn Winter	$n_{inlet\ 1+2}$ / n_{out}	12 / 20	12 / 20	9 / 17	14 / 14	12 / 14
	Mass in (kg d^{-1})	1.85 (± 0.41)	3.61 (± 0.79)	0.170 (± 0.010)	17.0 (± 4.3)	19.7 (± 9.2)
	Mass out (kg d^{-1})	0.976 (± 0.111)	0.762 (± 0.140)	0.086 (± 0.008)	1.07 (± 0.13)	2.00 (± 0.25)
	Daily mass intercepted (kg d^{-1})	0.874 (± 0.425)	2.85 (± 0.80)	0.084 (± 0.013)	15.9 (± 4.3)	17.7 (± 9.2)
	Daily mass intercepted per unit area (g $m^{-2}\ d^{-1}$)	0.397 (± 0.193)	1.29 (± 0.36)	0.038 (± 0.006)	7.24 (± 1.96)	8.05 (± 4.18)
	Mass reduction efficiency (%)	47 (± 13)	79 (± 6)	49 (± 6)	94 (± 2)	90 (± 5)
	Concentration reduction efficiency (%)	17 (± 21)	67 (± 10)	21 (± 26)	90 (± 3)	84 (± 8)
Spring Summer	$n_{inlet\ 1+2}$ / n_{out}	17 / 25	17 / 25	10 / 19	17 / 12	16 / 23
	Mass in (kg d^{-1})	1.50 (± 0.29)	1.27 (± 0.52)	0.139 (± 0.038)	14.3 (± 3.3)	23.1 (± 6.7)
	Mass out (kg d^{-1})	0.335 (± 0.065)	0.224 (± 0.091)	0.083 (± 0.008)	1.50 (± 0.20)	2.72 (± 0.44)
	Daily mass intercepted (kg d^{-1})	1.17 (± 0.30)	1.05 (± 0.53)	0.056 (± 0.039)	12.8 (± 3.3)	20.4 (± 6.7)
	Daily mass intercepted per unit area (g $m^{-2}\ d^{-1}$)	0.529 (± 0.135)	0.475 (± 0.240)	0.025 (± 0.018)	5.81 (± 1.50)	9.26 (± 3.05)
	Mass reduction efficiency (%)	78 (± 6)	82 (± 10)	40 (± 18)	90 (± 3)	88 (± 4)
	Concentration reduction efficiency (%)	65 (± 10)	72 (± 17)	5 (± 28)	83 (± 5)	81 (± 6)

Treatment efficiency by mass was 170% higher in spring/summer than in autumn/winter for NH_4, 110% higher NO_3 and 120% lower for RP, which could be explained by higher temperature (and hence higher bacterial activity) and larger plant uptake of NH_4. Efficiency was similar for SS, due to the main removal process being sedimentation, which is not temperature-dependant, and also similar for BOD_5, which may be due to algae contribution masking the improved removal expected at higher temperatures (since bacterial activity is a temperature-dependant process).

4.4.3.7 Correlations between inlet and outlet pollutant concentrations

Correlations between concentrations of pollutants at inlet 1+2 and outlet of CFW1 were investigated to examine if inflow and outflow quality are "coupled", which could indicate insufficient residence time and treatment. A significant positive correlation existed for NO_3 (r_s= 0.56, p< 0.001) and RP (r_s= 0.62, p= 0.01) but not for NH_4 (r_s= 0.32, p= 0.06). Figure 4.34 illustrates the relationships between the concentrations at inlet 1+2 and concentrations at outlet of NH_4, NO_3 and RP.

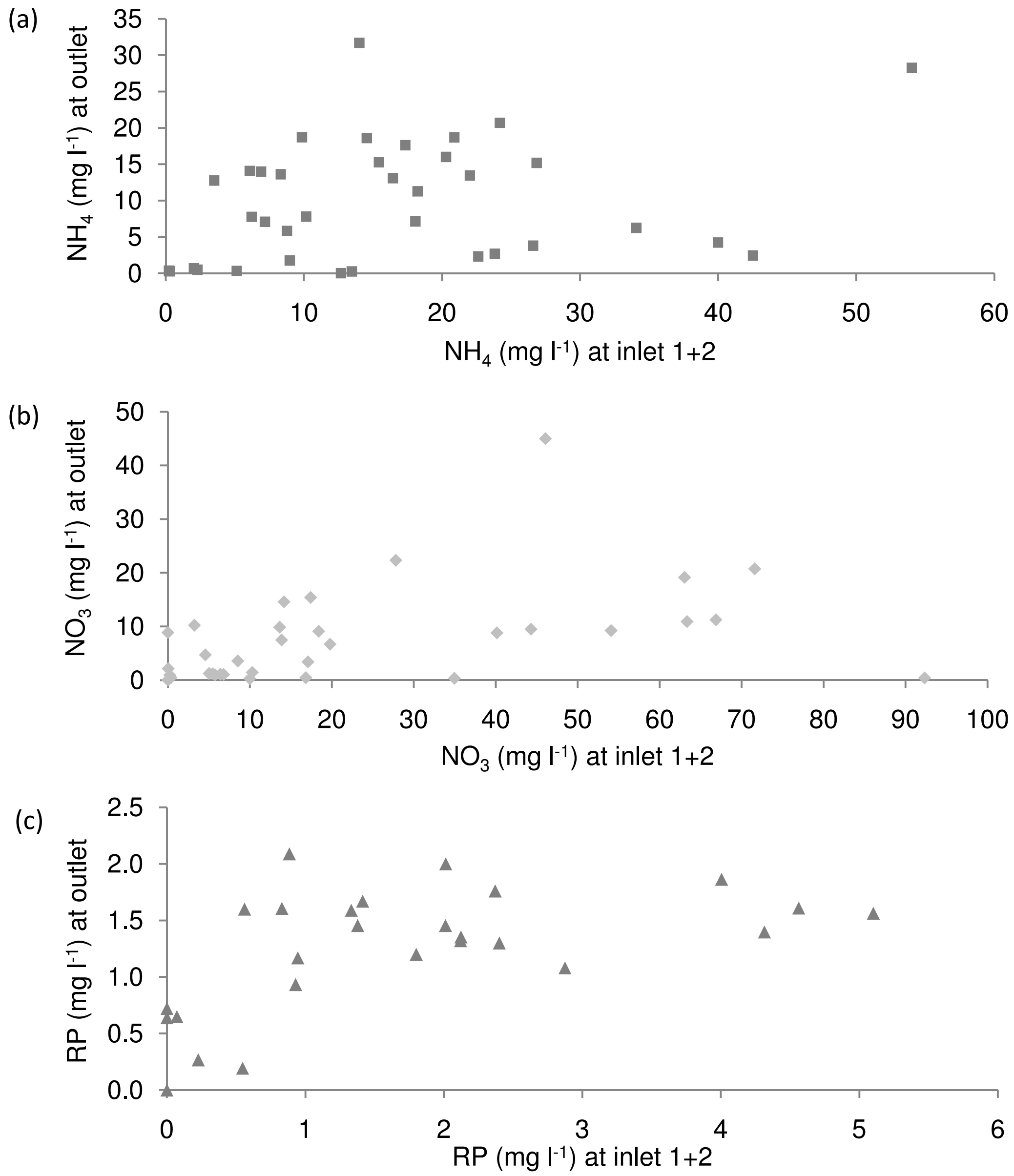


Figure 4.34 Relationships between the concentrations of (a) NH_4, (b) NO_3 and (c) RP at inlet 1+2 and outlet, CFW1.

4.4.3.8 Correlations between water quality parameters at the outlet

Correlations between NH_4, NO_3, RP, BOD_5 and pH (for which enough values were obtained) at the outlet were investigated to identify if some parameters could be predicted using others as surrogates, and the relationships between the concentrations of pollutants at outlet are illustrated in Figure 4.35.

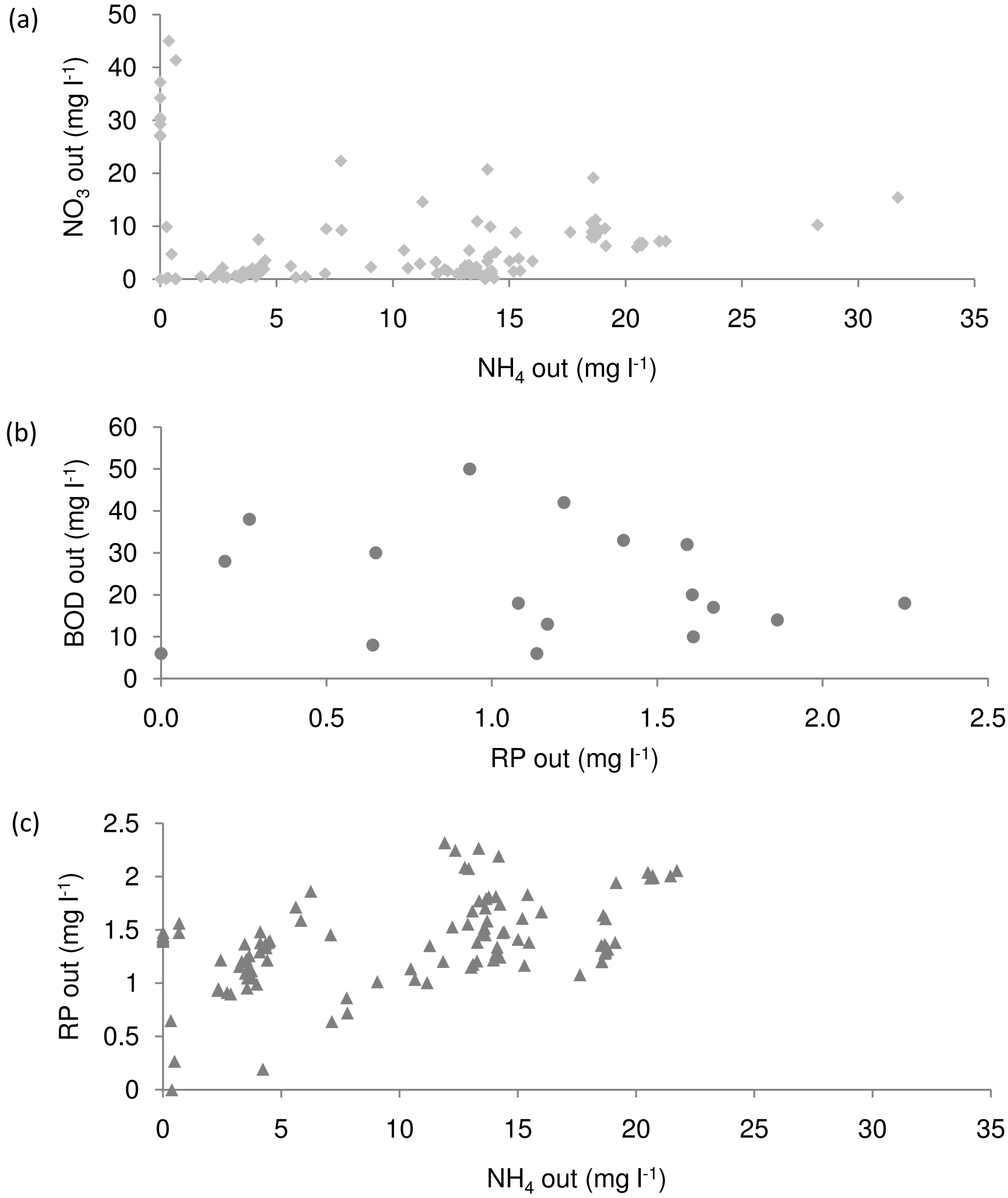


Figure 4.35 Relationships between the concentrations of (a) NO_3 and NH_4, (b) BOD_5 and RP, and (c) RP and NH_4 at outlet, CFW1.

No correlations appeared between pH or BOD_5 and any of the parameters, between NO_3 and NH_4 (while we could expect an inverse relationship due to transformation from one form to the other). However, RP concentration appeared to be positively correlated to NH_4 at outlet (r_s= 0.41, p< 0.001), suggesting that RP and NH_4 were removed to a lesser extent. High concentrations of RP were observed at low BOD_5 concentrations, suggesting that BOD_5 might not be a good surrogate for overall water quality in the case of CFWs, i.e. not a suitable water quality standard.

4.4.4 Key results for CFW1

- CFW achieved a high concentration and mass treatment efficiency for BOD_5 and SS, medium to high efficiency for NO_3 and NH_4 and limited treatment efficiency for RP. More importantly, the effluent it discharged still contained high concentration of all pollutants, above river water quality standards.
- Treatment efficiency by concentration appeared to be higher in spring/summer compared to autumn/winter for NO_3 and NH_4, but lower for RP (slightly higher RP concentration at inlet and outlet were observed in summer due to smaller dilution by field drainage).
- Calculation of treatment efficiency by mass may have overestimated CFW performance due to the large losses by infiltration which are not accounted for.
- Farmyard runoff was largely diluted by roof runoff, roof surface area representing c. 40% of the impervious area, which decreased the strength of the wastewater, and decreased treatment efficiency by concentration.
- The treatment volume available was too small compared to the amount of wastewater entering the system. Indeed, CFW1 was not designed for all current inputs and retention time was therefore limited during rainy periods. Moreover, daily inputs of groundwater "pushed out" contaminated pond water, creating a small but continuous pollution source.

4.5 Results for CFW2

4.5.1 General description of the monitoring period at CFW2

The monitoring of CFW2 took place between 16th February 2006 and 20th June 2008, the water quality monitoring started in April 2006, and flow measurements started in September 2006. The total amount of rainfall from February 2006 to end of June 2008 was 1792 mm, with August and October 2006, June and July 2007, and January and April 2008 being the wettest months, and April 2006, April and October 2007 and May 2008 being the driest (Figure 4.36). The highest daily rainfall totals were 33.4 mm on 20th June 2007, 27.6 mm on 2nd August 2006, 26 mm on 30th June 2008, 23.6 mm on 20th October 2006, 23.0 mm on 18th August 2006 and 22.8 mm on 4th

January 2008. Rainfall intensity reached a maximum of 13.8 mm h^{-1} on 20th October 2006, 11.8 mm h^{-1} on 20th June 2007, 11.4 mm h^{-1} on 24th September 2007 and 10.2 mm h^{-1} on 20th June 2007.

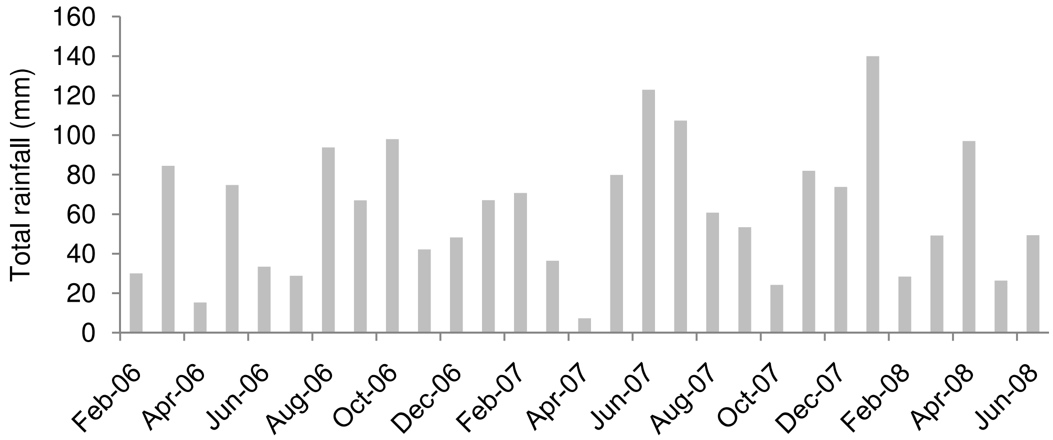


Figure 4.36 Total monthly rainfall at CFW2 from February 2006 to June 2008.

Mean annual rainfall for the area is 830 mm (BADC Harelaw station, 3 km from CFW2, 2000-2006) and rainfall is well distributed across the year, although October is wetter on average (mean rainfall of 120 mm). Temperatures recorded inside the datalogger box varied between 21°C in summer (18/07/2006) and - 8°C in winter (17/12/2007), with the year 2007 cooler on average than 2006 (Figure 4.37). Climatic conditions during the study period were comparable to long-terms trends in south-east Scotland (MetOffice, 2008)

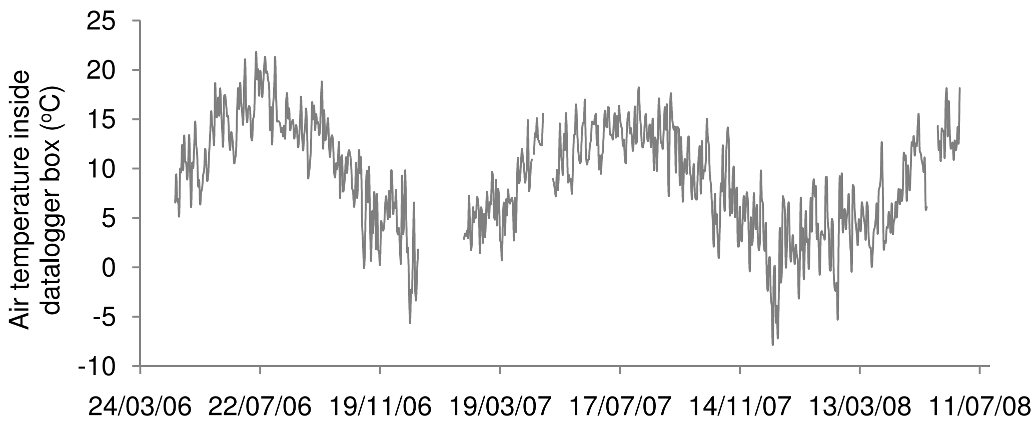


Figure 4.37 Mean daily temperature recorded inside the datalogger box over the monitoring period at CFW2.

4.5.2 Water balance at CFW2

4.5.2.1 Water balance overview

Table 4.11 summarizes the different inputs and outputs into and from CFW2 over the monitoring period. Corresponding calculations and assumptions are explained below in more detail. A discrepancy exists between inputs (12% more) and outputs, suggesting underestimation of the outflows or overestimation of the inflows.

Table 4.11 Water balance overview for CFW2.

Monitoring period	Between May 2006 and June 2008
Total rainfall	1667.6 mm
Rainfall volume intercepted by the wetland (0.5 ha)	8335 m^3, i.e. 11 $m^3\ d^{-1}$
Inflow (Farmyard runoff, field drainage, septic tank overflow)	Mean Daily Volume: 415 $m^3\ d^{-1}$; Mean inflow: 4.8 $l\ s^{-1}$ Spring/summer: 390 $m^3\ d^{-1}$ Autumn/winter: 439 $m^3\ d^{-1}$
Groundwater inputs	Not measured individually but significant. Mean inflow of 0.5 $l\ s^{-1}$, i.e. 43 $m^3\ d^{-1}$ on average (between 0.05 $l\ s^{-1}$ during dry periods and 1.0 $l\ s^{-1}$ during wet periods, i.e. between 4 $m^3\ d^{-1}$ and 86 $m^3\ d^{-1}$)
Outflow (through normal outflow)	Mean outflow volume: 328 $m^3\ d^{-1}$; Mean outflow: 3.8 $l\ s^{-1}$ Spring/summer: 256 $m^3\ d^{-1}$ (2.9 $l\ s^{-1}$) Autumn/winter: 403 $m^3\ d^{-1}$ (4.7 $l\ s^{-1}$)
Evapotranspiration	About 21 $m^3\ d^{-1}$ over 5000 m^2 throughout the year
Infiltration	Average of 13 mm d^{-1}, i.e. 65 $m^3\ d^{-1}$
Total daily inputs	469 $m^3\ d^{-1}$
Total daily outputs	414 $m^3\ d^{-1}$

4.5.2.2 Individual inputs

Inputs from the rain over the surface of the wetland

Direct precipitation input to the wetland surface (the surface directly interacting with wastewater and draining from one pond to another estimated at c. 5000 m^2) between 1st May 2006 and 30th June 2008 was estimated at 8335 m^3 (i.e. 11 m^3 d^{-1}).

Field drainage, farmyard runoff, septic tank overflow and groundwater

Field drainage (34 ha) and farmyard runoff (1.8 ha) represented the most significant inputs in terms of volume and pollutant loads. Field drainage was larger in autumn/winter when the soil was saturated and evaporation reduced. Flow measurement between May 2006 and June 2008 at the outlet of P1, gave a mean inflow of 4.8 l s^{-1}, i.e. 415 m^3 d^{-1}, with mean inflow in spring and summer of 4.5 l s^{-1}, i.e. 390 m^3 d^{-1}, and mean inflow in winter of 5.1 l^{-1}, i.e. 439 m^3 d^{-1}.

For a typical septic tank used for 16 population equivalents in summer and 4 the rest of the time, with a daily volume of about 400 l per day per person, c. 2.8 m^3 d^{-1} are expected on average for the whole year. This corresponds to an average flow of about 0.03 l s^{-1}. Septic overflow characteristics could not be investigated due to the mixing with large volumes of groundwater into the incoming pipe. BOD_5 concentrations measured in the 1st pond were surprisingly very low all over the monitoring period, with a mean of 1.3 mg l^{-1} and maximum of 10 mg l^{-1} (in November 2007).

An unknown quantity of groundwater entered CFW2 through a drain, which flowed even during dry conditions, creating a minimum inflow of around 0.05 to 1 l s^{-1} during dry days. Considering a mean inflow of 0.5 l s^{-1}, groundwater inputs were estimated at c. 43 m^3 d^{-1} on average over the year.

4.5.2.3 Individual outputs

Outflow from CFW2 (P5)

Figure 4.38 shows the water level in P5 (in cm) recorded by pressure transducer and used to extrapolate the outflow, with a noticeable dry summer in 2006 and wet summer in 2007.

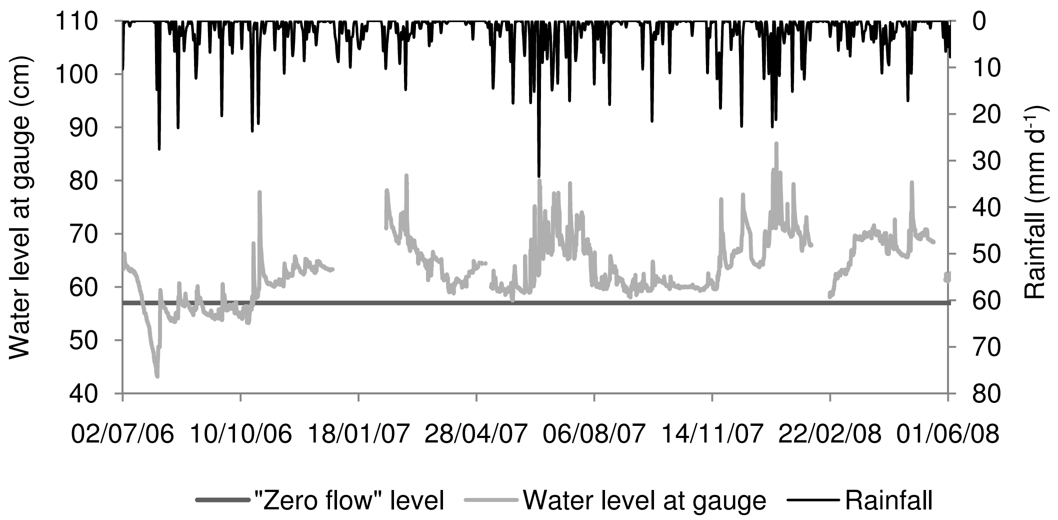


Figure 4.38 Water level in P5 (CFW2) detected by pressure transducer, and daily rainfall. "Zero flow" level corresponds to the level of the bottom of the outlet pipe.

The mean daily outflow volume over the monitoring period, estimated by extrapolating flow from level in P5, was 328 $m^3 d^{-1}$ (3.8 $l s^{-1}$), with on average 403 $m^3 d^{-1}$ (4.7 $l s^{-1}$) in autumn/winter, and 256 $m^3 d^{-1}$ (i.e. 2.9 $l s^{-1}$) in spring/summer.

Lateral outflow

Water escaped the system frequently during rainy periods before entering P5, due to clogging of the outlet pipe from P4 and improper levelling on the east edge of the wetland, causing flooding of the track between ponds 4 and 5. The amount leaving the CFW in this manner could not be assessed, but is expected to be significant, especially in 2007 and 2008, when conditions were wetter.

Evapotranspiration

Losses by evapotranspiration (especially in summer) were expected to be more important at CFW2 than at CFW1 because the system was highly vegetated (dominant species being *P. australis*) and the area of open water was also larger, especially in the last pond. Fermor *et al*. (2001), based on experiments on reedbeds in North East England, found an average evapotranspiration rate of 2.28 mm d^{-1} (over the whole year), with average rates of 3.32 mm d^{-1} in spring/summer (maximum: 4.35 mm d^{-1}) and 1.24 mm d^{-1} in autumn/winter (minimum: 0.29 mm d^{-1}). From these data, the average volume of water lost by evapotranspiration from the CFW surface area of 5000 m^2 was estimated to be 11.5 m^3 d^{-1}.

Infiltration

Infiltration within P5, which has a loamy-clay base (soil from the Eckford series, with fluvio-glacial sand as parent material), was calculated using level fluctuations on given dates when no outflow was occurring, and taking into account the constant groundwater inflow (estimates of 0.2 l s^{-1} in summer and 0.5 l^{-1} in winter, estimated indirectly by measuring water level in the pipe at P1 out on the same dates) and evapotranspiration (Fermor *et al*., 2001). Groundwater recharge entering the wetland from beneath could not be assessed. Infiltration was estimated to be c. 13 mm d^{-1}, i.e. 65 m^3 d^{-1}, considering a wet saturated surface area of 5000 m^2, assuming a spatially uniform infiltration rate over this surface, from P1 to P5.

4.5.3 Water quality and treatment efficiency at CFW2

4.5.3.1 Overview of the water sampling period

Figure 4.39 shows inflow into and outflow from the wetland over the whole monitoring period, as well as rainfall. Grab sampling took place on 38 occasions and storm sampling was carried out on 27th February 2007, 12th-13th June 2007, 3rd-6th and 15th-21st August 2007, and 17th-21st November 2007.

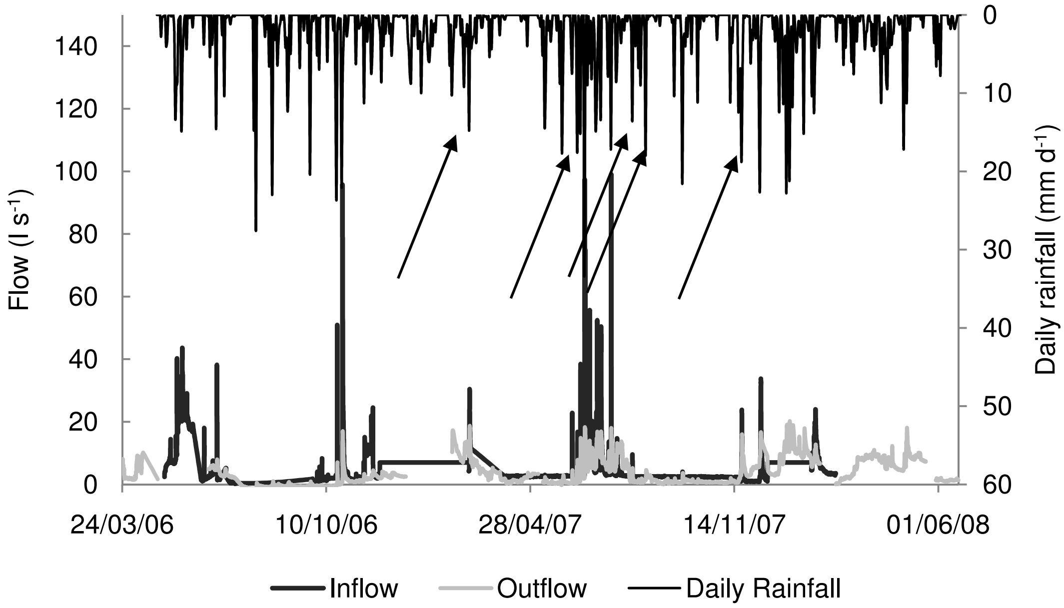


Figure 4.39 Mean hourly inflow, mean hourly outflow and total daily rainfall at CFW2 between April 2006 and June 2008. Storm event sampling dates are located by arrows.

Table 4.12 summarizes the water quality data obtained for CFW2 (see Figure 4.7 for sampling locations) between 2006 and 2008. The only pollutant reaching critical concentrations at inlet as well as outlet during the monitoring period was NO_3, reaching 58.0 mg l^{-1} in P1 and 50.3 mg l^{-1} in P5 (measured once in February 2007). RP concentration was relatively low throughout the year compared to CFW1, i.e. < 0.9 mg l^{-1} at inlet and < 0.3 mg l^{-1} at outlet, but still above river water quality standards (< 0.1 mg l^{-1}). Ammonium concentration reached 3.5 mg l^{-1} at inlet and 4.5 mg l^{-1} at outlet, and BOD_5 concentration remained around background levels, always < 10 mg l^{-1} at inlet and outlet. Average pH along the wetland was alkaline, increasing slightly from the inlet (mean pH 7.5) to the outlet (mean pH 8.3).

Table 4.12 Water quality along CFW2 between 2006 and 2008 (n: number of samples; SE: standard error; P: pond; W: wetland) (2nd part on the next page).

Water quality parameter		Water sampling location									
		Inlet farm	Inlet field	P1 mid	P1 out	W1	W2	P2 in	P2 mid	P2 out	W3
NH_4 (mg l^{-1})	n	22	9	24	36	24	25	26	15	27	25
	Mean	0.758	0.630	0.719	0.554	0.275	0.297	0.280	0.470	0.361	0.616
	SE	0.166	0.250	0.136	0.112	0.040	0.046	0.033	0.117	0.071	0.119
	Median	0.450	0.414	0.467	0.325	0.232	0.259	0.284	0.394	0.278	0.439
	Max	2.99	2.53	2.43	3.51	0.659	0.931	0.645	1.69	1.54	1.95
NO_3 (mg l^{-1})	n	22	9	24	36	24	25	26	15	27	25
	Mean	31.5	37.5	33.8	31.4	34.7	31.3	31.3	27.0	29.2	31.0
	SE	2.9	5.7	2.6	1.9	2.6	2.8	2.9	3.8	2.5	3.6
	Median	33.8	40.2	36.9	30.5	39.2	33.5	33.2	27.8	27.6	30.7
	Max	51.9	59.8	54.2	58.0	53.6	53.7	53.3	48.4	52.9	75.6
RP (mg l^{-1})	n	16	7	12	28	11	12	15	2	16	12
	Mean	0.081	0.043	0.081	0.078	0.065	0.060	0.079	0.025	0.076	0.080
	SE	0.041	0.026	0.034	0.025	0.028	0.027	0.039	0.025	0.035	0.030
	Median	0.011	0.000	0.034	0.033	0.017	0.033	0.000	0.025	0.000	0.006
	Max	0.606	0.158	0.346	0.898	0.279	0.291	0.508	0.049	0.513	0.280
BOD_5 (mg l^{-1})	n	-	-	-	12	-	-	-	-	5	-
	Mean	-	-	-	1.0	-	-	-	-	1.0	-
	SE	-	-	-	0.9	-	-	-	-	0.9	-
	Median	-	-	-	0	-	-	-	-	0	-
	Max	-	-	-	10	-	-	-	-	5	-
SS (mg l^{-1})	n	6	4	-	20	-	-	-	-	3	-
	Mean	8.18	33.2	-	65.6	-	-	-	-	2.93	-
	SE	3.30	18.6	-	34.2	-	-	-	-	0.50	-
	Median	6.20	26.5	-	21.3	-	-	-	-	3.20	-
	Max	20.7	78.5	-	703	-	-	-	-	3.60	-
FC (cfu 100 ml^{-1})	n	-	-	-	3	-	-	-	-	-	-
	Mean	-	-	-	1300	-	-	-	-	-	-
	SE	-	-	-	1100	-	-	-	-	-	-
	Median	-	-	-	300	-	-	-	-	-	-
	Max	-	-	-	3500	-	-	-	-	-	-
FS (cfu 100 ml^{-1})	n	-	-	-	3	-	-	-	-	-	-
	Mean	-	-	-	210	-	-	-	-	-	-
	SE	-	-	-	110	-	-	-	-	-	-
	Median	-	-	-	100	-	-	-	-	-	-
	Max	-	-	-	430	-	-	-	-	-	-
pH	n	18	6	8	20	10	9	10	4	15	9
	Mean	7.54	6.81	7.11	7.82	7.70	7.69	7.84	7.23	8.12	8.03
	SE	0.61	0.29	0.16	0.64	0.31	0.26	0.30	1.08	0.53	0.40
	Max	8.40	7.69	7.55	9.90	9.20	9.20	9.40	10.1	11.0	11.1
Conductivity (µS cm^{-1})	n	-	-	-	26	-	-	-	-	-	-
	Mean	-	-	-	697	-	-	-	-	-	-
	SE	-	-	-	5	-	-	-	-	-	-
	Median	-	-	-	738	-	-	-	-	-	-
	Max	-	-	-	752	-	-	-	-	-	-

Water quality parameter		Water sampling location									
		P3 in	P3 mid	P3 out	W4	P4 in	P4 mid	P4 out	P5 storm	P5 mid	P5 out
NH_4 (mg l^{-1})	n	26	15	20	25	25	15	32	14	23	38
	Mean	0.631	0.301	0.435	0.424	0.469	0.382	0.476	0.623	0.454	0.366
	SE	0.174	0.052	0.114	0.084	0.148	0.074	0.140	0.267	0.161	0.046
	Median	0.353	0.258	0.340	0.346	0.325	0.291	0.257	0.422	0.239	0.316
	Max	3.59	0.599	2.49	1.89	2.86	1.13	4.37	4.02	3.84	4.27
NO_3 (mg l^{-1})	n	26	15	21	25	25	15	32	14	23	38
	Mean	28.7	25.4	27.1	27.3	24.3	14.9	20.5	16.2	21.9	21.7
	SE	3.30	4.14	3.81	3.27	3.40	4.34	2.69	3.87	3.56	2.68
	Median	30.2	25.5	23.9	25.7	23.9	4.88	20.1	14.4	19.7	20.9
	Max	53.3	49.4	61.3	57.1	52.4	46.0	51.9	45.6	48.0	50.3
RP (mg l^{-1})	n	13	4	16	12	14	2	22	14	11	29
	Mean	0.032	0.126	0.043	0.065	0.089	0.000	0.103	0.110	0.124	0.054
	SE	0.024	0.116	0.021	0.033	0.040	0.000	0.051	0.073	0.090	0.020
	Median	0.000	0.016	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	Max	0.288	0.472	0.271	0.275	0.484	0.000	1.003	1.004	0.987	0.464
BOD_5 (mg l^{-1})	n	-	-	3	-	-	-	13	-	-	16
	Mean	-	-	0.0	-	-	-	1.0	-	-	2
	SE	-	-	0.0	-	-	-	0.3	-	-	0.8
	Median	-	-	0	-	-	-	0	-	-	1
	Max	-	-	0	-	-	-	4	-	-	10
SS (mg l^{-1})	n	-	-	2	-	-	-	9	-	-	21
	Mean	-	-	2.95	-	-	-	5.59	-	-	7.80
	SE	-	-	0.35	-	-	-	1.80	-	-	2.60
	Median	-	-	3.00	-	-	-	4.10	-	-	4.00
	Max	-	-	3.20	-	-	-	17.0	-	-	55.0
FC (cfu 100 ml^{-1})	n	-	-	-	-	-	-	3	-	-	3
	Mean	-	-	-	-	-	-	1600	-	-	50
	SE	-	-	-	-	-	-	1500	-	-	49
	Median	-	-	-	-	-	-	100	-	-	100
	Max	-	-	-	-	-	-	4600	-	-	150
FS (cfu 100 ml^{-1})	n	-	-	-	-	-	-	3	-	-	3
	Mean	-	-	-	-	-	-	300	-	-	300
	SE	-	-	-	-	-	-	200	-	-	200
	Median	-	-	-	-	-	-	100	-	-	100
	Max	-	-	-	-	-	-	700	-	-	700
pH	n	10	5	14	11	10	4	25	11	11	26
	Mean	8.15	7.50	8.19	8.12	8.01	7.14	8.20	8.06	8.19	8.28
	SE	0.42	0.83	0.52	0.39	0.34	1.02	0.70	0.35	0.32	0.70
	Max	11.4	10.8	11.5	10.7	10.2	9.70	10.5	9.60	9.50	9.40
Conductivity (µS cm^{-1})	n	-	-	-	-	-	-	-	-	-	18
	Mean	-	-	-	-	-	-	-	-	-	661
	SE	-	-	-	-	-	-	-	-	-	15
	Median	-	-	-	-	-	-	-	-	-	680
	Max	-	-	-	-	-	-	-	-	-	718

- Data not available.

The quality of the farmyard runoff alone could not be assessed since it was mixed with septic tank overflow and groundwater. However, it concentrations of BOD_5, NH_4 and RP were very low compared to CFW1, which may suggest a less contaminated farmyard, less available forms of pollutants (dry dung instead of fresh faeces), a failure to collect and transport dirty yard runoff to the wetland, dilution of farmyard runoff with relatively clean shallow groundwater throughout the year, or attenuation of pollutant in the pipes by attached microbial films. At CFW2, farmyard runoff was transported over a greater distance before reaching the wetland and therefore attenuation of contaminants might have happened through dilution and interaction with biofilms in the drains. The rather high NO_3 concentrations could be due to the oxidation of NH_4 in the runoff between the farmyard and the wetland.

Field drainage from pipe 2 contained relatively high concentrations of NO_3 and low concentrations of other pollutants perhaps due to the absence of slurry application. Although drainage water was not analysed for FIOs, coliforms coming from faeces deposited by grazing animals were expected. Reduction in FIOs was generally observed between inlet and outlet of CFW2, but a high concentration of FC (> 150 000 cfu ml^{-1}) was measured in summer 2007, probably due to inputs from wildfowl (mainly swans and ducks). Groundwater quality could not be assessed on its own, since it was mixed with septic tank overflow, but was expected to contain NO_3 which had leached from the fields through the ground. Septic tank overflow could not be assessed directly either since it was mixed early on in the pipe with groundwater.

4.5.3.2 Spatial and temporal fluctuations in water quality

Spatial heterogeneity in water quality along CFW2

Kruskal Wallis tests and pairwise comparisons were carried out to assess differences in concentrations between locations. No differences existed for NH_4 (H= 10.834, p= 0.929) and RP (H= 27.87, p= 0.086) (therefore, results are not illustrated), but a difference existed for NO_3 (H= 52.488, p= 0.0001), whose median concentration was significantly higher closer to the inlet of the wetland (Figure 4.40).

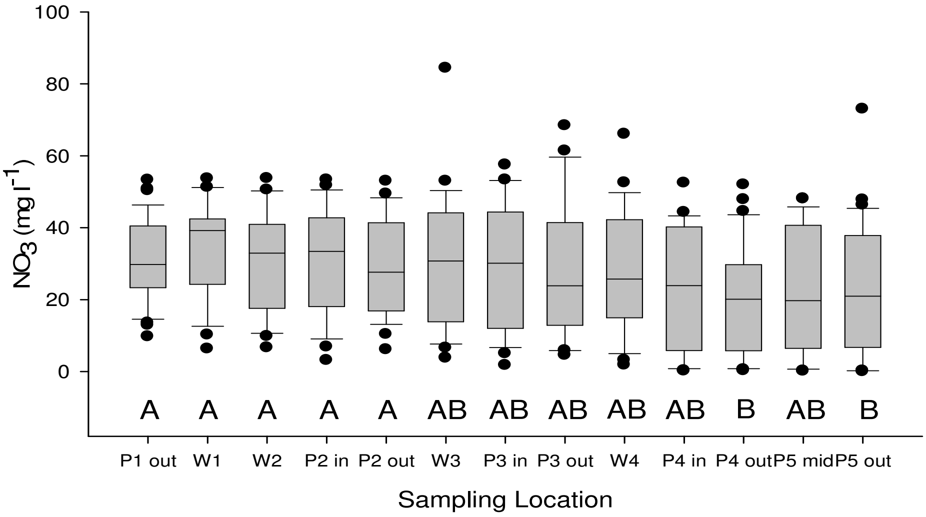


Figure 4.40 Variations in NO_3 concentration along CFW2. Boxes delimitate the lower and upper quartiles, horizontal bars indicate the medians, whiskers the minima and maxima, and dots the extreme values. Locations which do not share a capital letter have significantly different median concentrations (p< 0.05).

No conclusion could be drawn regarding the contribution of ponds or vegetated grass areas in NO_3 removal, since concentration appeared significantly different only between the inlet and outlet of CFW2. However, a small decrease in nitrate concentration was observed between inlet and outlet of the smaller individual ponds (which was not observed in P5), and a small increase occurred in W1, W2, W3 and W4 compared to previous samples from the ponds outlets, maybe due to nitrification or other inputs (e.g. groundwater) unaccounted for.

Long-term fluctuations in effluent quality

At P1 out, NH_4 concentration fluctuated between < 0.01 and 3.5 mg l^{-1} and was higher in June and August 2006, during dry days, maybe due to septic overflow in use by summer workers (Figure 4.41). At the outlet, it varied between < 0.01 and 1.5 mg l^{-1}, the highest values occurring throughout the year. Inlet NO_3 concentration fluctuated between 10 and 60 mg l^{-1}, higher values being in autumn/winter when biological activity was lower, which fits in with seasonal patterns usually observed in freshwaters (Mason, 2002), and ranged at the outlet from < 0.01 to 48 mg l^{-1}. RP concentration in the wetland effluent remained < 0.5 mg l^{-1} during the monitoring period, but still above the river water quality standards.

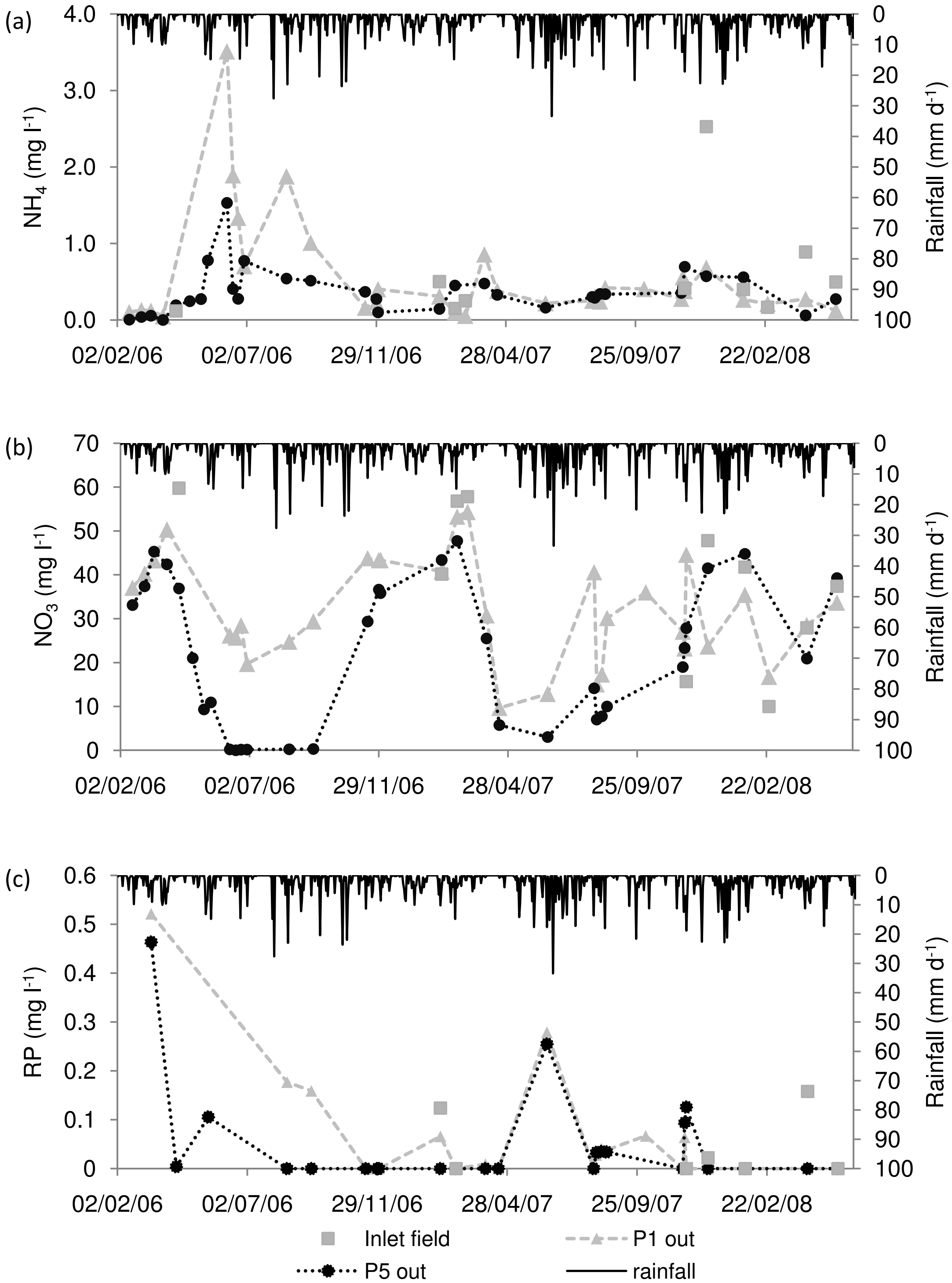


Figure 4.41 Daily rainfall and fluctuations in the concentration of (a) NH_4, (b) NO_3 and (c) RP at inlet and outlet of CFW2 in grab samples.

Spatial water quality fluctuations on selected dates

Figure 4.42 shows concentrations of NO_3 along the wetland for selected dates in summer and winter representing different hydrological and environmental conditions (14/09/06: dry conditions; 28/02/07: during rainy period with saturated fields; 06/08/07: following rainfall). In spring/summer, water quality improvements were often observed between inlet and outlet while, in winter, the concentration remained at 40-50 mg l^{-1} throughout the system. However, as shown by sampling following a rainy period in August 2007, concentrations remained high at the outlet even in summer, if residence time was insufficient.

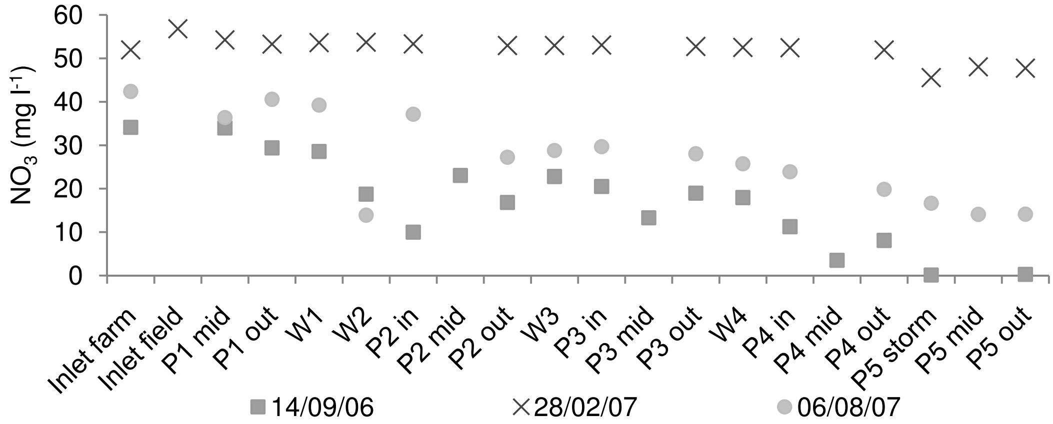


Figure 4.42 Concentration of NO_3 from the inlet to the outlet of CFW2 on different dates.

Seasonal fluctuations in water quality

Table 4.13 summarizes the data obtained for selected water quality parameters, for samples collected in autumn and winter (AUTWIN) and spring and summer (SPRSUM). Ammonium concentration was significantly lower at the farm inlet (H= 24.0, p= 0.043) and P1 out (H= 80, p= 0.027) in autumn/winter (when inflow was diluted by field drainage) than in spring/summer (when septic tanks were used by workers) but no differences existed for the other locations. No differences existed for RP between seasons at any of the locations, but differences existed for NO_3, which was significantly higher in autumn/winter at Inlet farm (H= 80, p= 0.043), P1 out (H= 231, p= 0.003), P4 out (H= 171, p= 0.002) and P5 out (H= 317, p< 0.0002).

Table 4.13 Seasonal variations in water quality along CFW2 (n: number of samples; SE: standard error).

Sampling location	Season		Water quality parameter		
			NH_4	NO_3	RP
Inlet farm	AUTWIN	n	9	9	9
		Mean (mg l^{-1})	0.33	36.4	0.09
		SE	0.06	4.7	0.07
		Median (mg l^{-1})	0.28	42.6	0.02
	SPRSUM	n	13	13	7
		Mean (mg l^{-1})	1.05	28.1	0.07
		SE	0.25	3.5	0.05
		Median (mg l^{-1})	0.58	26.3	0.00
Inlet field	AUTWIN	n	6	6	5
		Mean (mg l^{-1})	0.69	35.4	0.03
		SE	0.37	7.6	0.02
		Median (mg l^{-1})	0.41	41.0	0.00
	SPRSUM	n	3	3	2
		Mean (mg l^{-1})	0.50	41.7	0.08
		SE	0.22	9.4	0.08
		Median (mg l^{-1})	0.50	37.4	0.08
P1 outlet	AUTWIN	n	17	17	13
		Mean (mg l^{-1})	0.28	37.6	0.06
		SE	0.04	2.4	0.04
		Median (mg l^{-1})	0.27	40.3	0.00
	SPRSUM	n	19	19	15
		Mean (mg l^{-1})	0.80	25.9	0.09
		SE	0.19	2.3	0.03
		Median (mg l^{-1})	0.42	26.2	0.04
P3 outlet	AUTWIN	n	10	10	9
		Mean (mg l^{-1})	0.49	39.6	0.03
		SE	0.23	4.6	0.02
		Median (mg l^{-1})	0.33	37.5	0.00
	SPRSUM	n	10	11	7
		Mean (mg l^{-1})	0.38	15.8	0.05
		SE	0.05	3.4	0.04
		Median (mg l^{-1})	0.34	14.1	0.00
P4 outlet	AUTWIN	n	14	14	12
		Mean (mg l^{-1})	0.56	30.8	0.16
		SE	0.30	4.0	0.09
		Median (mg l^{-1})	0.25	29.8	0.00
	SPRSUM	n	18	18	10
		Mean (mg l^{-1})	0.41	12.6	0.03
		SE	0.10	2.3	0.03
		Median (mg l^{-1})	0.28	10.5	0.00
P5 outlet	AUTWIN	n	15	15	12
		Mean (mg l^{-1})	0.32	36.0	0.07
		SE	0.07	2.2	0.04
		Median (mg l^{-1})	0.35	36.6	0.00
	SPRSUM	n	23	23	17
		Mean (mg l^{-1})	0.40	12.3	0.05
		SE	0.06	2.8	0.02
		Median (mg l^{-1})	0.30	9.3	0.01

4.5.3.3 Water quality fluctuations and hysteresis during storm events

Table 4.14 summarizes the hydrological characteristics of the storm events investigated at CFW2.

Table 4.14 Summary of hydrological characteristics of the storm events investigated at CFW2 (NA: not assessed; FSC: field storage capacity).

Storm event No. Date	Total Rainfall Max intensity	Inflow characteristics	Outflow volume (% total volume attenuation)	Notes
Storm event 1 27/02/07	15 mm 0.8 mm 5 min^{-1}	Max flow: 40 l s^{-1} Volume: 3153 m^3 Runoff: 210 m^3 mm^{-1}	2280 m^3 (28%)	FSC = 0 Outflow underestimated due to lateral flows
Storm event 2 12/06/07-13/06/07	20.6 mm 0.6 mm 5 min^{-1}	Max flow: 20 l s^{-1} Volume: 377 m^3 Runoff: 18 m^3 mm^{-1}	324 m^3 (14%)	FSC > 0
Storm event 3 03/08/07-06/08/07	13.6 mm 2.2 mm 5 min^{-1}	Max flow: 12 l s^{-1} Volume: 166 m^3 Runoff: 12 m^3 mm^{-1}	145 m^3 (13%)	FSC > 0 Inflow underestimated (sensor failure)
Storm event 4 13/08/07-21/08/07	9.6 mm 1.2 mm 5 min^{-1}	Max flow: NA Volume: NA Runoff: > 16 m^3 mm^{-1}	180 m^3 (NA)	FSC > 0
Storm event 5 17/11/07-22/11/07	57.6 mm 0.6 mm 5 min^{-1}	Max flow: 28 l^{-1} Volume: 2254 m^3 Runoff: 39 m^3 mm^{-1}	2551 m^3 (20%)	FSC > 0 Inflow underestimated (sensor failure)

Inputs into CFW2 were higher during larger storm events occurring when the fields were saturated, and large peak flows were usually observed with higher rainfall intensity. Flow attenuation by the wetland was between 13% and 28%, low compared to larger ICWs (Rory Harrington, *pers. comm.*, 2005), mainly due to short-circuiting (improper levelling) between ponds. Storm events 2, 4 and 5 are detailed below, to illustrate the main flow and water quality patterns during rainy periods.

Storm event 2: 12th - 13th June 2007

Event 2 (Figure 4.43) was characterised by a large and continuous rainfall (20.6 mm) of low intensity, a maximum flow of 20 l s^{-1}, and generated a runoff (380 m^3) enriched in RP, NH_4, but containing less NO_3 than the inflow previous to the rainfall.

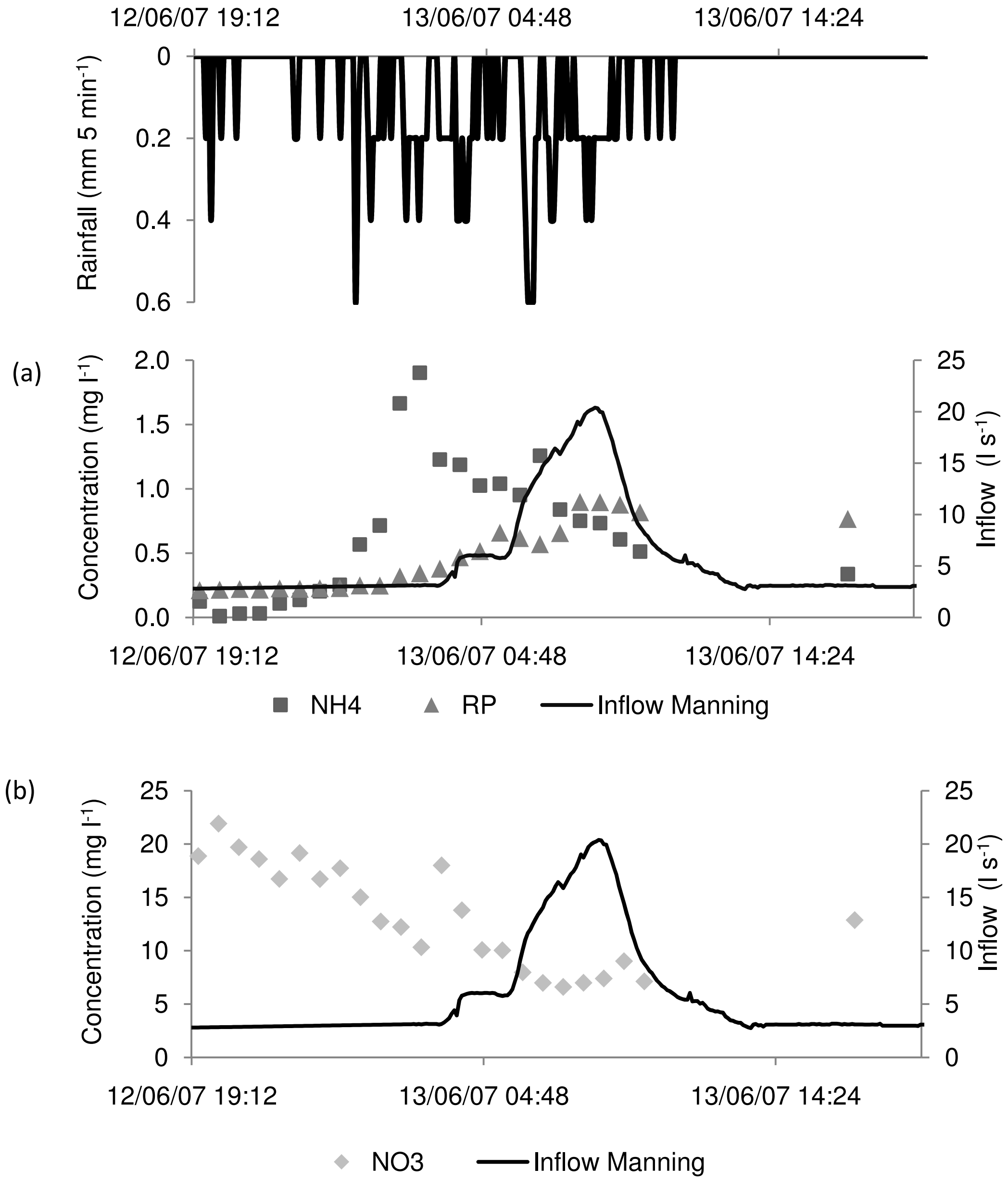


Figure 4.43 Concentration of (a) NH_4 and RP, and (b) NO_3, and flow at the inlet of CFW2 during a storm event in June 2007.

A first flush of NH_4 and RP was observed at P1 out, the earliest being for NH_4. Concentration of NH_4 increased rapidly at the beginning of the event (the sensor did not detect the real inflow), reached 2 mg l^{-1} and then decreased nearly to its original value, while RP showed a later and steady increase, reached 0.9 mg l^{-1}, and decreased only slightly. This trend may be explained by an early input of rapidly mobilised NH_4 from farmyard runoff at the beginning of the rainfall, and later input from field drainage which contains RP, but lower concentrations of NH_4. A different pattern was observed for NO_3, whose concentration decreased, due to dilution of groundwater inputs by farmyard and field drainage, containing less NO_3.

The hysteresis pattern at inlet for RP (Figure 4.44) illustrates its early mobilisation, its concentration increasing together with the flow

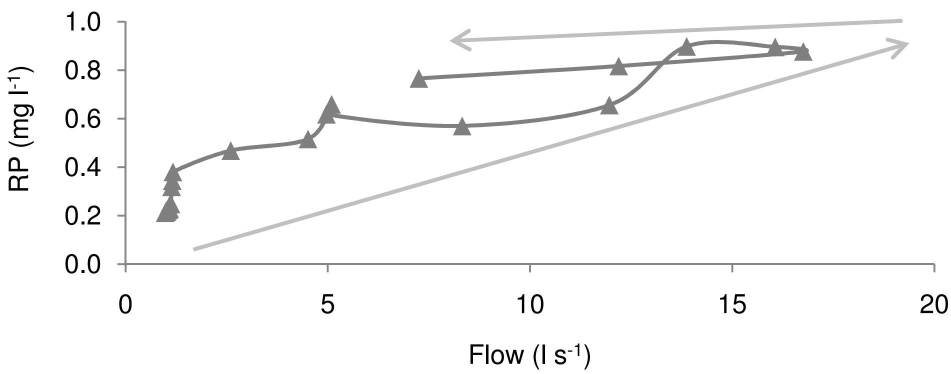


Figure 4.44 Hysteresis pattern observed for RP during storm event 3 (June 2007) at the inlet of CFW2.

At P5 out (Figure 4.45), NH_4 concentration first decreased, due to dilution by runoff and rainfall, and increased later on, which may be explained by fresh inputs or mobilisation within the pond. Reactive phosphorus did not seem to be flushed out and its concentration remained relatively stable during the event, around 0.25 mg l^{-1}. Nitrate concentration remained relatively low (< 3.0 mg l^{-1}), which illustrates the lower concentrations usually observed in summer. It increased early on, coupled with an increase in the outflow, decreased in the later part of the event, perhaps due to the displacement of water with a higher concentration (as shown at inlet) already present in the wetland before the storm, followed by more diluted runoff later on. Finally, NO_3 concentrations returned to those prior to the event, suggesting less dilution.

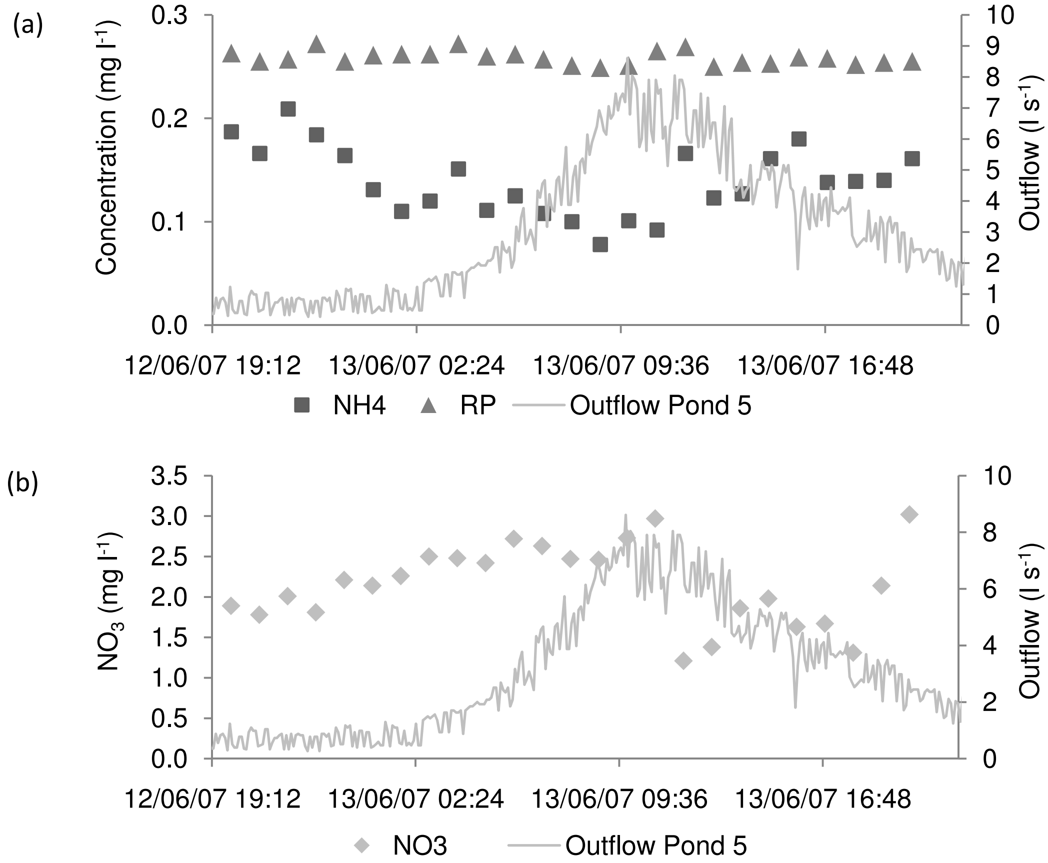


Figure 4.45 Concentration of (a) NH_4, RP and (b) NO_3 and flow at the outlet of CFW2 during a storm event in June 2007.

Mean concentration reduction efficiency (± standard error) for this event between P1 out and P5 out was 99% (± 2%) for NH_4, 84% (± 1%) for NO_3, and 43% (± 6%) for RP, and mean flux reduction efficiency was 92% (± 2%) for NH_4, 90% (± 2%) for NO_3 and 81% (± 6%) for RP.

Storm event 4: 13th - 21st August 2007

Figure 4.46 illustrates changes in NH_4, RP and NO_3 concentrations at CFW2 inlet during a storm event in August 2007. Relatively high concentrations of NO_3 were observed, due to particularly wet conditions.

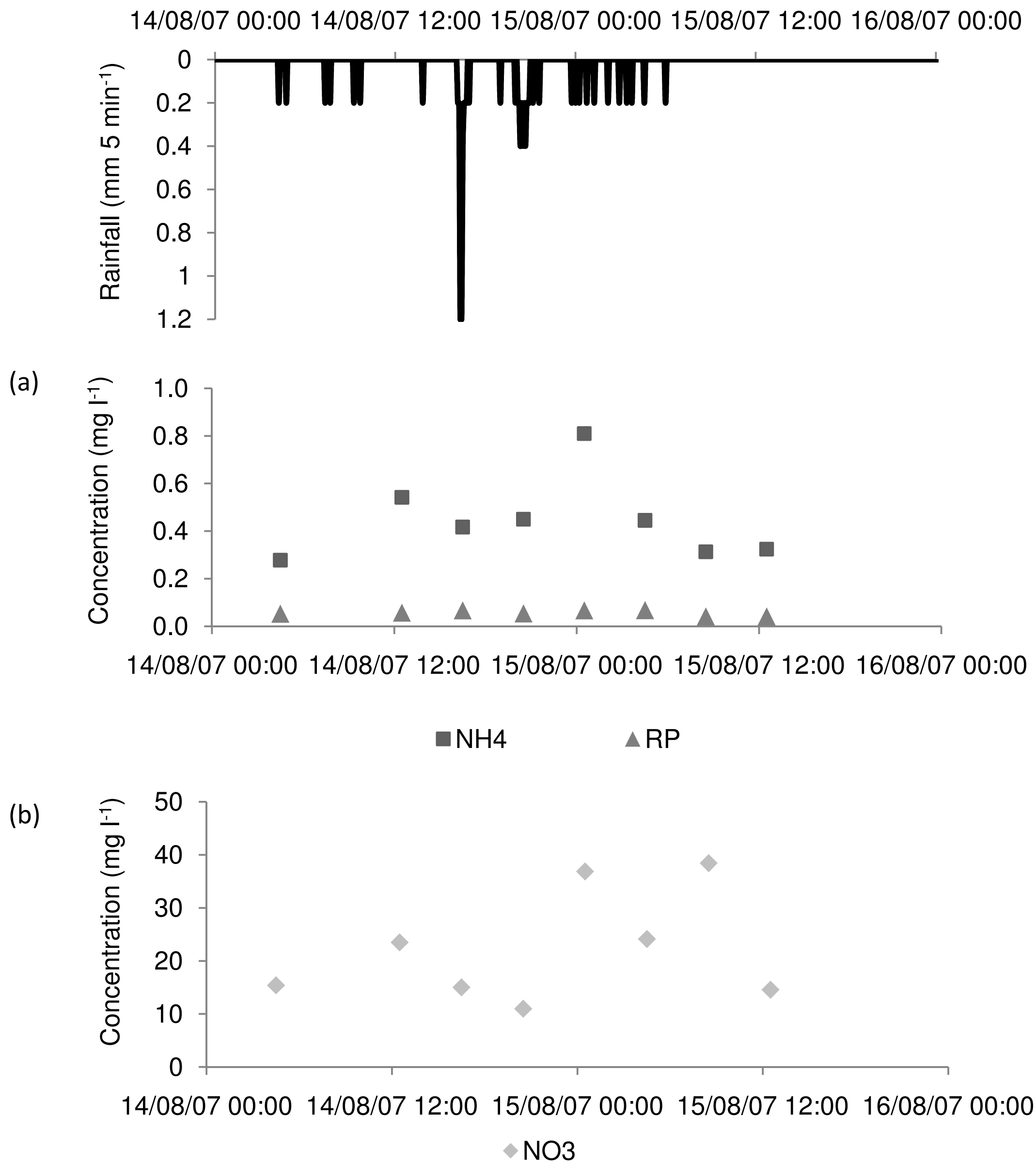


Figure 4.46 Concentration of (a) NH_4, RP and (b) NO_3 at the inlet of CFW2 during a storm event in August 2007.

At the outlet (Figure 4.47), only small concentration fluctuations were observed, NH_4 and NO_3 increased slightly during the rising parts of the flow. Ammonium concentration in the pond decreased between the 15th and 22nd August, while NO_3 concentration remained relatively stable, around 11 mg l^{-1}.

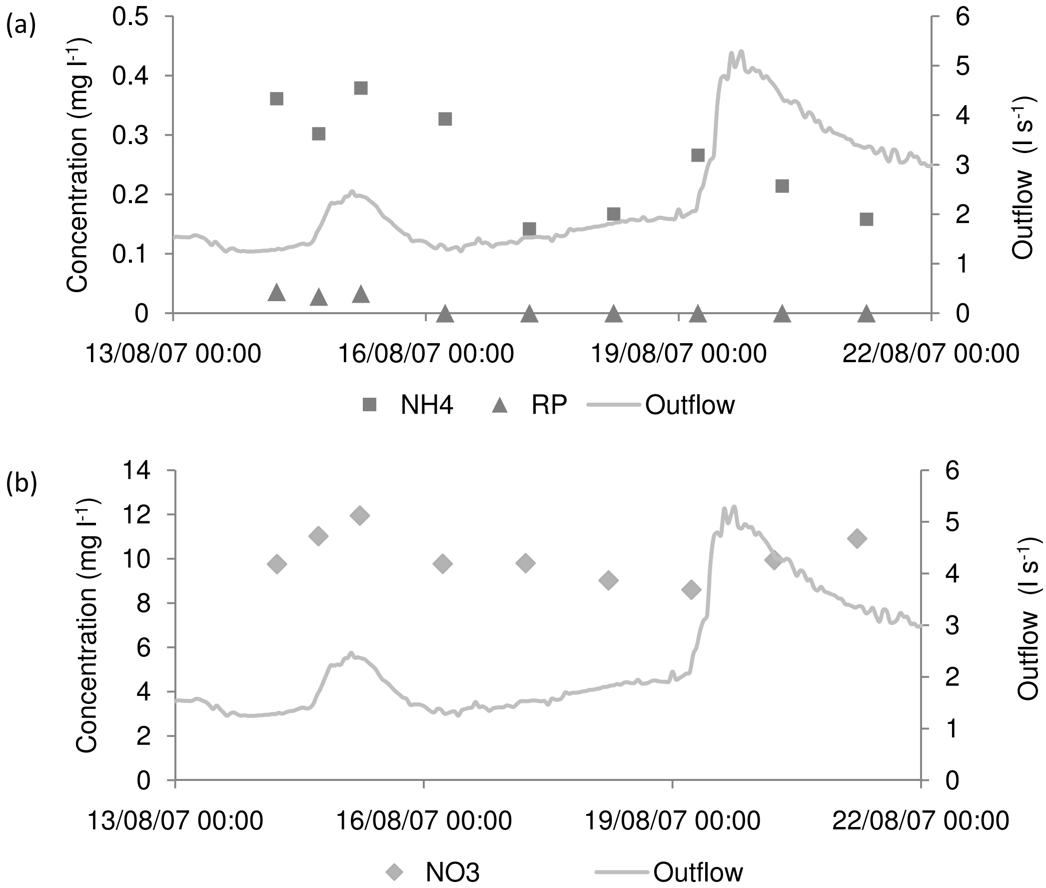


Figure 4.47 Concentration of (a) NH_4, RP and (b) NO_3, and flow at the outlet of CFW2 during a storm event in August 2007.

Mean concentration reduction efficiency (± standard error) between P1 out and P5 out was 42% (± 9%) for NH_4, 54% (± 7%) for NO_3, and 83% (± 17%) for RP.

Storm event 5: 17th - 21st November 2007

Sensor failure led to underestimation of the inflow on the 19th November. On the 19th, a small increase of NH_4 and RP was observed at inlet, corresponding to a period with the onset of the rain. On the 20th and 21st, in contrast to storm event 2, the highest RP concentrations (maximum of only c. 0.3 mg l^{-1}) were observed before flow peak (Figure 4.48). Concentrations of NH_4 and RP appeared lower than in storm event 2, which could be explained by wetter antecedent conditions or smaller inputs from septic tank (no workers in winter). Concentration of NO_3 fluctuated between 25 and 50 mg l^{-1} and decreased at higher flow, due to dilution.

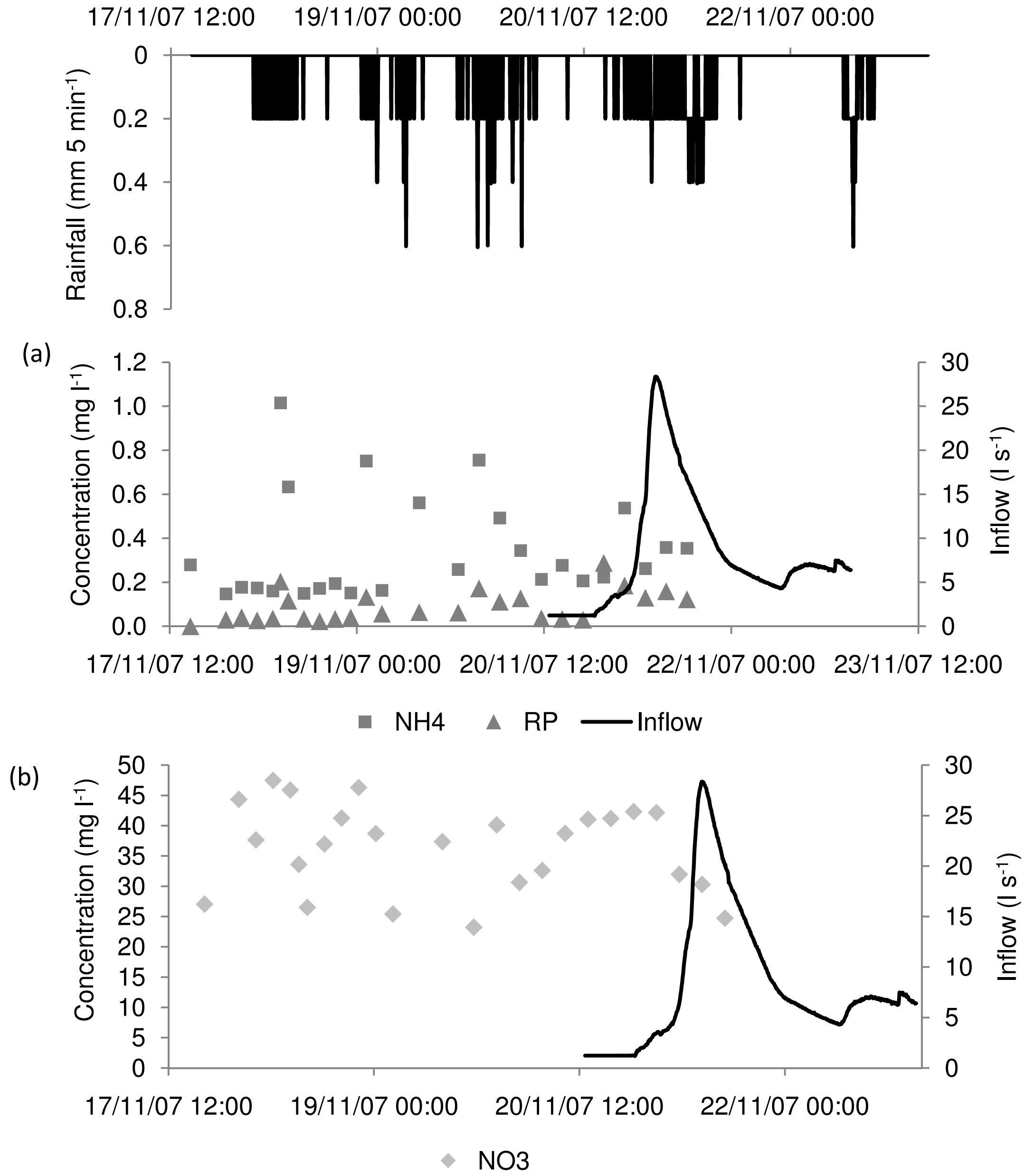


Figure 4.48 Concentration of (a) NH_4, RP and (b) NO_3, and flow at the inlet of CFW2 in November 2007.

At the outlet (Figure 4.49), no significant changes in NH_4, NO_3 or RP were observed during the event. Compared to the first storm event (February 2007), NO_3 concentrations were slightly lower in the last pond, between 30 and 40 mg l^{-1}.

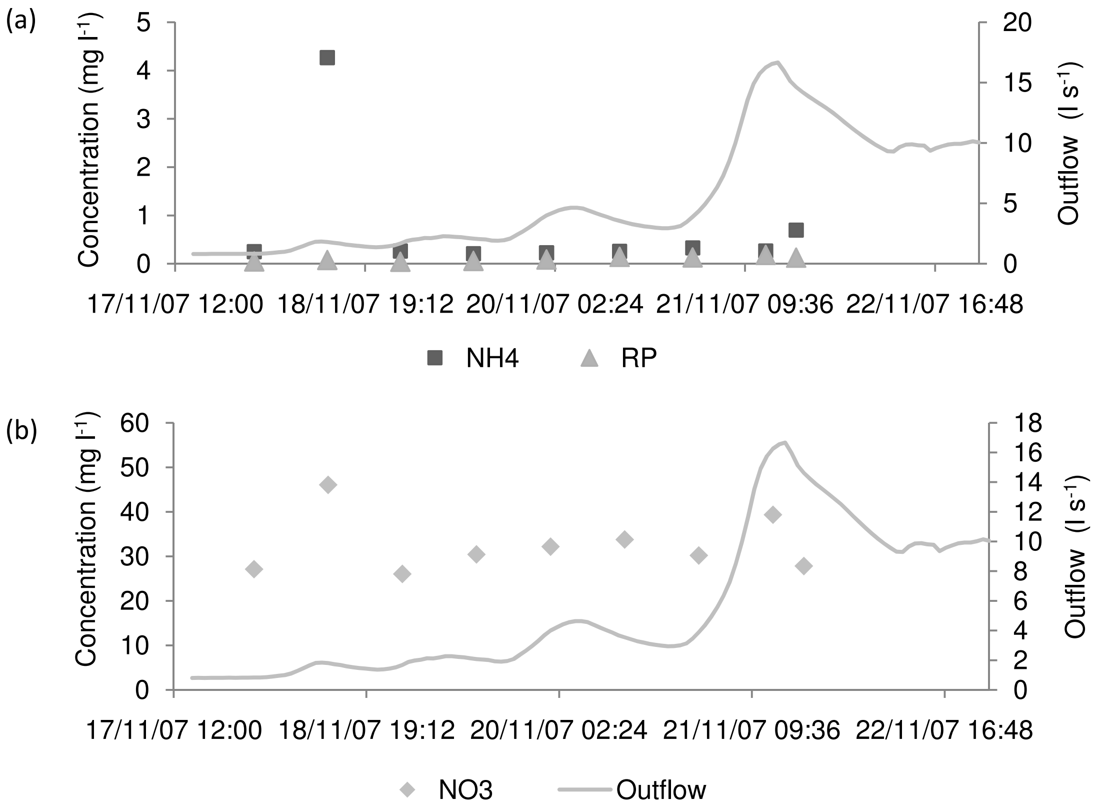


Figure 4.49 Concentration of (a) NH_4, RP and (b) NO_3, and flow at the outlet of CFW2 during a storm event in November 2007.

Mean concentration reduction efficiency (± standard error) between P1 out and P5 out was < 0% for NH_4, 10% (± 7%) for NO_3, and < 0% for RP, and mean flux reduction efficiency was < 0% for NH_4, NO_3 and RP due to inflow underestimation.

4.5.3.4 Influence of antecedent rainfall on water quality

Correlation analyses were carried out to investigate the relationship between concentrations of pollutants and antecedent rainfall (AR, as defined in section 4.4.3.4) at P1 out and P5 out of CFW2. A strong positive correlation existed between RP concentration at P5 out and 5-day AR (r_s= 0.65, p= 0.0018), but no other correlation existed for any of the other pollutants. Figures 4.50 and 4.51 illustrates the relationship between 2-day AR at P1 out and 5-day AR at P5 out for NH_4, NO_3 and RP concentrations measured in grab samples and first samples of storm events.

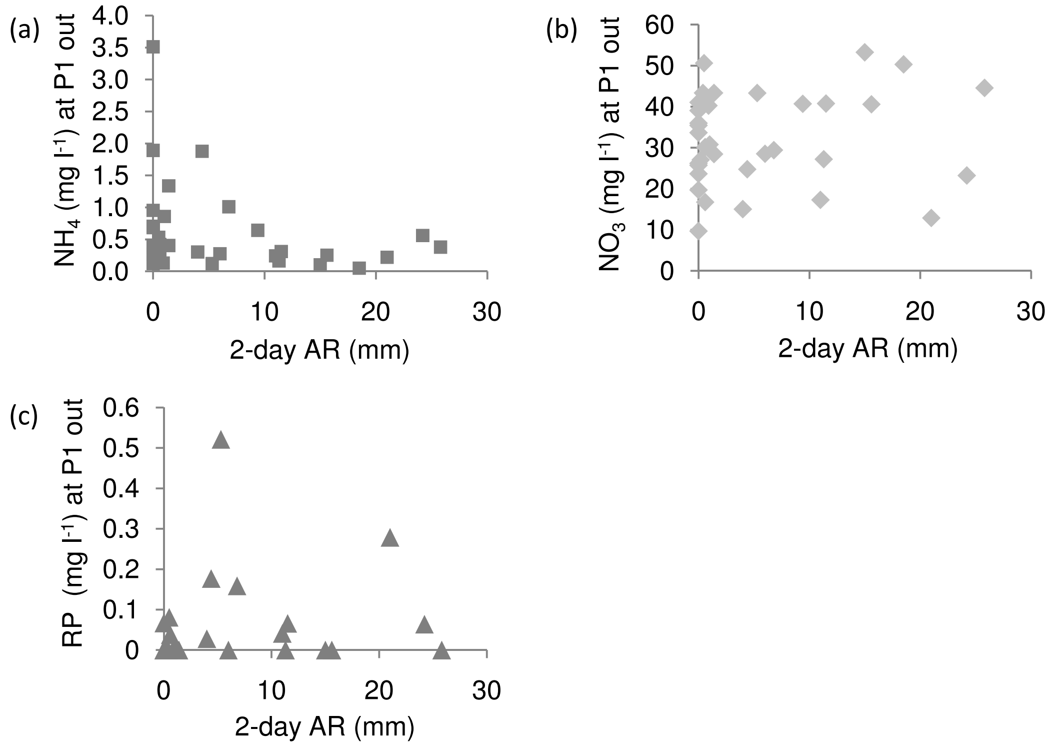


Figure 4.50 Relationships between the concentration of (a) NH_4, (b) NO_3 and (c) RP and 2-day antecedent rainfall (AR) at P1 out, CFW2.

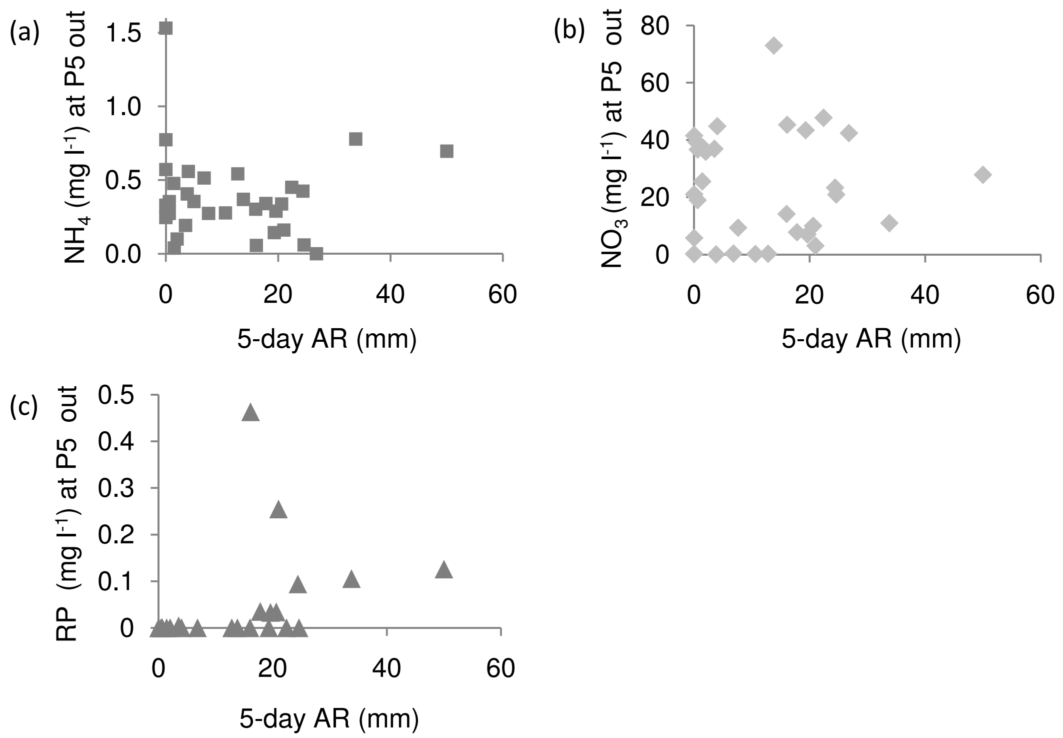


Figure 4.51 Relationships between the concentration of (a) NH_4, (b) NO_3 and (c) RP and 5-day antecedent rainfall (AR) at P5 out, CFW2.

A very weak inverse relationship (R^2= 0.06) between inlet NH_4 concentration and 2-day AR was observed, the highest NH_4 concentrations occurring with the lowest antecedent rainfall, indicating a source of NH_4 that can be exhausted. No relation appeared between 5-day AR and concentration at the outlet. No obvious patterns appeared for NO_3, probably because its main source is shallow groundwater in the field drains, rather than farmyard runoff which would respond more to antecedent rainfall. A very weak positive relationship (R^2= 0.13) between RP at outlet and 5-day AR was observed, which could be explained by higher inputs and maybe release of RP during wetter conditions and higher flows.

4.5.3.5 Treatment efficiency by concentration

Overall treatment efficiency by concentration between P1 out and P4 out or P5 out is illustrated in Table 4.15, which shows that higher efficiencies by mean concentration were for SS, RP, NO_3 and NH_4.

Treatment for mean NH_4 and mean RP concentrations was significantly improved by P5, while NO_3 and SS removal was actually higher between P1 and P4 than between P1 and P5. This increase in NO_3 and SS in P5 compared to P4 could be caused by nitrification, resuspension or release of solids and organic matter from sediment and plants, or by lateral or ground inputs from adjacent fields. A high concentration of faecal coliforms was observed on one sampling occasion, perhaps due to faeces inputs from swans and ducks in P5 or to the resuspension of viable coliforms, explaining the high median concentration at P5 outlet and negative efficiency.

Table 4.15 Overall mean and median concentration reduction efficiency between P1 out and P4 out, and P1 out and P5 out at CFW2 (n: number of samples; SE: standard error).

Water quality parameter	Water sampling location			Concentration Reduction Efficiency (%)	
	P1 out	P4 out	P5 out	Efficiency P1 to P4	Efficiency P1 to P5
NH_4 (mg l^{-1})					
n	36	32	38		
Mean (± SE)	0.554 (± 0.112)	0.476 (± 0.140)	0.366 (± 0.046)	14 (± 31)	34 (± 16)
Median	0.325	0.257	0.316	21	3
NO_3 (mg l^{-1})					
n	36	32	38		
Mean (± SE)	30.960 (± 1.888)	20.477 (± 2.703)	22.811 (± 3.000)	34 (± 10)	26 (± 11)
Median	29.772	20.092	20.950	33	30
RP (mg l^{-1})					
n	28	22	29		
Mean (± SE)	0.078 (± 0.025)	0.103 (± 0.051)	0.054 (± 0.020)	< 0	31 (± 34)
Median	0.033	0.000	0.000	100	100
BOD_5 (mg l^{-1})					
n	12	13	16		
Mean (± SE)	1.3 (± 0.8)	0.6 (± 0.3)	1.9 (± 0.7)	< 0	< 0
Median	0	0	1	0	< 0
SS (mg l^{-1})					
n	15	9	16		
Mean (± SE)	49.6 (± 34.2)	5.6 (± 1.8)	8.4 (± 3.3)	89 (± 10)	83 (± 15)
Median	21.3	4.1	4.4	81	80
FC (cfu 100 ml^{-1})					
n	3	3	3		
Mean (± SE)	1300 (± 1102)	1600 (± 1500)	50 067 (± 49 967)	< 0	< 0
FS (cfu 100 ml^{-1})					
n	3	3	3		
Mean (± SE)	210 (± 110)	300 (± 200)	300 (± 200)	< 0	< 0

Seasonal treatment efficiency by concentration is summarized in Table 4.16. The highest treatment efficiencies by mean concentration occurred in spring/summer for all pollutants, and were similar, in the range of 40 to 50%.

Table 4.16 Mean and median seasonal concentration reduction efficiency (CRE) calculated for NH_4, NO_3 and RP between P1 out and P5 out (n: number of samples; ± standard error in brackets).

Season		Water quality parameter		
		NH_4	NO_3	RP
Autumn Winter	$n_{P1\ out}$ / $n_{P5\ out}$	17 / 15	17 / 15	13 / 12
	Mean CRE (%)	< 0	4 (± 8)	< 0
	Median CRE (%)	< 0	9	0
Spring Summer	$n_{P1\ out}$ / $n_{P5\ out}$	19 / 23	19 / 23	15 / 17
	Mean CRE (%)	50 (± 14)	53 (± 12)	44 (± 29)
	Median CRE (%)	29	64	75

4.5.3.6 Treatment efficiency by mass

To assess mass loadings in the long-term by interpolating concentration when flow was known, correlations and relationships between concentrations of NH_4, NO_3 and RP measured at inlet and outlet during grab and storm sampling, and flow were investigated (Figure 4.52). No significant correlations were obtained for any of the pollutants. However, for RP, the highest concentrations were observed at flows above 3 l s^{-1}, suggesting higher RP leaching from the fields to the drains during rainy periods and no dilution effect for the flows measured.

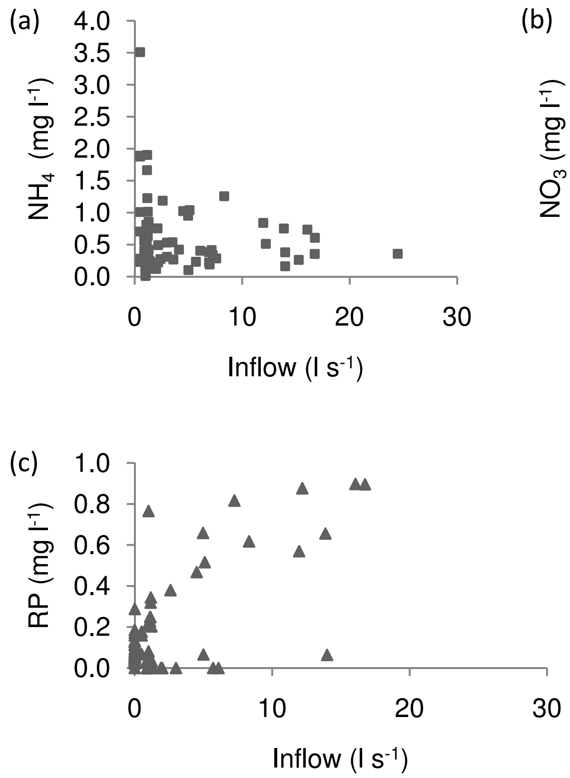


Figure 4.52 Relationships between the concentration of (a) NH_4, (b) NO_3 or (c) RP, and inflow, CFW2.

At P5 out, a strong positive correlation appeared between NO_3 concentration and flow (r_s= 0.41, p< 0.005), suggesting higher concentrations at higher flows during heavy rainfall, and no correlations were found for NH_4 and RP (Figure 4.53).

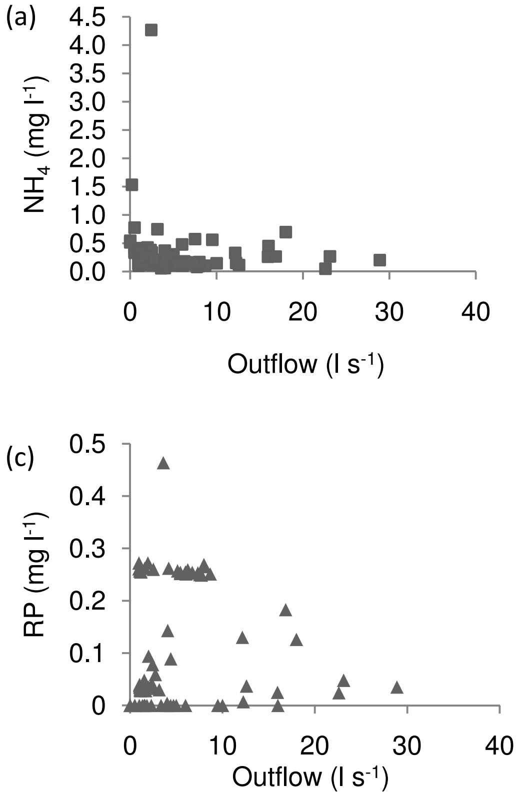


Figure 4.53 Relationships between the concentration of (a) NH_4, (b) NO_3 and (c) RP and outflow, CFW2.

Since there were no straightforward relationships between flow and concentration of pollutants, efficiency was estimated using average concentration data and average inputs/outputs from the wetland, as for CFW 1.

Treatment efficiency by mass using mean concentrations

Treatment efficiencies were calculated for the period April 2006 to June 2008 using the mean measured inputs and outputs to and from the system at P1 out and P5 out of 415 m^3 d^{-1} and 328 m^3 d^{-1}, respectively. Overall, CFW2 intercepted c. 40 kg yr^{-1} NH_4, 1960 kg yr^{-1} NO_3, 5 kg RP and 6510 kg yr^{-1} SS, and higher treatment efficiencies were for SS (87%), NH_4 (48%), NO_3 (45%) and RP (45%) (Table 4.17).

Table 4.17 Mean mass reduction efficiency (MRE) between P1 out and P5 out at CFW2 for selected pollutants using mean concentrations at inlet and outlet (± standard error in brackets).

	Water quality parameter			
	NH_4	NO_3	RP	SS
Mean conc. in (mg l^{-1})	0.554 (± 0.112)	31.4 (± 1.9)	0.078 (± 0.025)	49.6 (± 34.2)
Mass in (g d^{-1})	230 (± 46)	13 030 (± 784)	32.4 (± 10.4)	20 584 (± 14 193)
Mean conc. out (mg l^{-1})	0.366 (± 0.046)	21.7 (± 2.8)	0.054 (± 0.020)	8.40 (± 3.30)
Mass out (g d^{-1})	120 (± 15)	7120 (± 918)	17.7 (± 6.6)	2760 (± 1080)
Mass intercepted (g d^{-1})	110 (± 49)	5910 (± 1210)	14.7 (± 12.3)	17 800 (± 14 200)
Mass reduction efficiency (%)	48 (± 23)	45 (± 8)	45 (± 41)	87 (± 97)
Areal mass removal (g m^{-2} d^{-1}) (area: 5000 m^2)	0.0219 (± 0.0098)	1.18 (± 0.24)	0.003 (± 0.002)	3.57 (± 2.85)

Seasonal treatment efficiency by mass:

Seasonal fluxes and mass reduction efficiencies were estimated (Table 4.18) using mean concentrations of pollutants in samples taken in spring/summer and autumn/winter and assuming a mean inflow of 390 m^3 d^{-1} and mean outflow of 256 m^3 d^{-1} in spring/summer, and a mean inflow of 439 m^3 d^{-1} and mean outflow of 403 m^3 d^{-1} in autumn/winter.

Table 4.18 Seasonal pollutant fluxes and differences in mean concentration and mass reduction efficiency at CFW1 between P1 outlet and P5 outlet (n: number of samples; ± standard error in brackets).

Season		Water quality parameter		
		NH_4	NO_3	RP
Autumn Winter	$n_{P1\ out}$ / $n_{P5\ out}$	17 / 15	17 / 15	13 / 12
	Mass in (g d^{-1})	122 (± 18)	16 070 (± 1070)	26.3 (± 17.6)
	Mass out (g d^{-1})	129 (± 28)	15700 (± 1310)	28.2 (± 16.1)
	Daily mass intercepted (g d^{-1})	< 0	375 (± 1690)	< 0
	Daily mass intercepted per unit area (g m^{-2} d^{-1})	< 0	0.075 (± 0.338)	< 0
	Mass reduction efficiency (%)	< 0	2 (± 10)	< 0
	Concentration reduction efficiency (%)	< 0	4 (± 8)	< 0
Spring Summer	$n_{P1\ out}$ / $n_{P5\ out}$	19 / 23	19 / 23	15 / 17
	Mass in (g d^{-1})	312 (± 74)	10100 (± 905)	35.1 (± 11.7)
	Mass out (g d^{-1})	10.2 (± 15.4)	3140 (± 719)	12.8 (± 5.1)
	Daily mass intercepted (g d^{-1})	301 (± 76)	6960 (± 1160)	22.3 (± 12.8)
	Daily mass intercepted per unit area (g m^{-2} d^{-1})	0.060 (± 0.015)	1.39 (± 0.23)	0.004 (± 0.003)
	Mass reduction efficiency (%)	97 (± 34)	69 (± 13)	64 (± 42)
	Concentration reduction efficiency (%)	50 (± 14)	53 (± 12)	44 (± 29)

CFW2 performed better in spring/summer than in autumn/winter (only c. 2% reduction for NO_3) due to smaller inputs and higher temperature. The highest efficiency in summer was for NH_4 (97%), NO_3 (69%) and RP (64%), aware that NH_4 and RP were around background concentrations. Reduction efficiency by mass was higher than by concentration for all pollutants except for NO_3 in autumn/winter.

4.5.3.7 Correlations between inlet and outlet pollutant concentrations

Significant positive correlations between P1 out and P5 out concentrations existed for NH_4 (r_s= 0.54, p< 0.005), NO_3 (r_s= 0.74, p< 0.001) and RP (r_s= 0.67, p< 0.001) Figure 4.54 illustrates the increase in outlet concentration in response to the increase in inlet concentration (regression is only significant for NO_3), which may suggest a relatively short residence time and limited treatment, despite the size of the system.

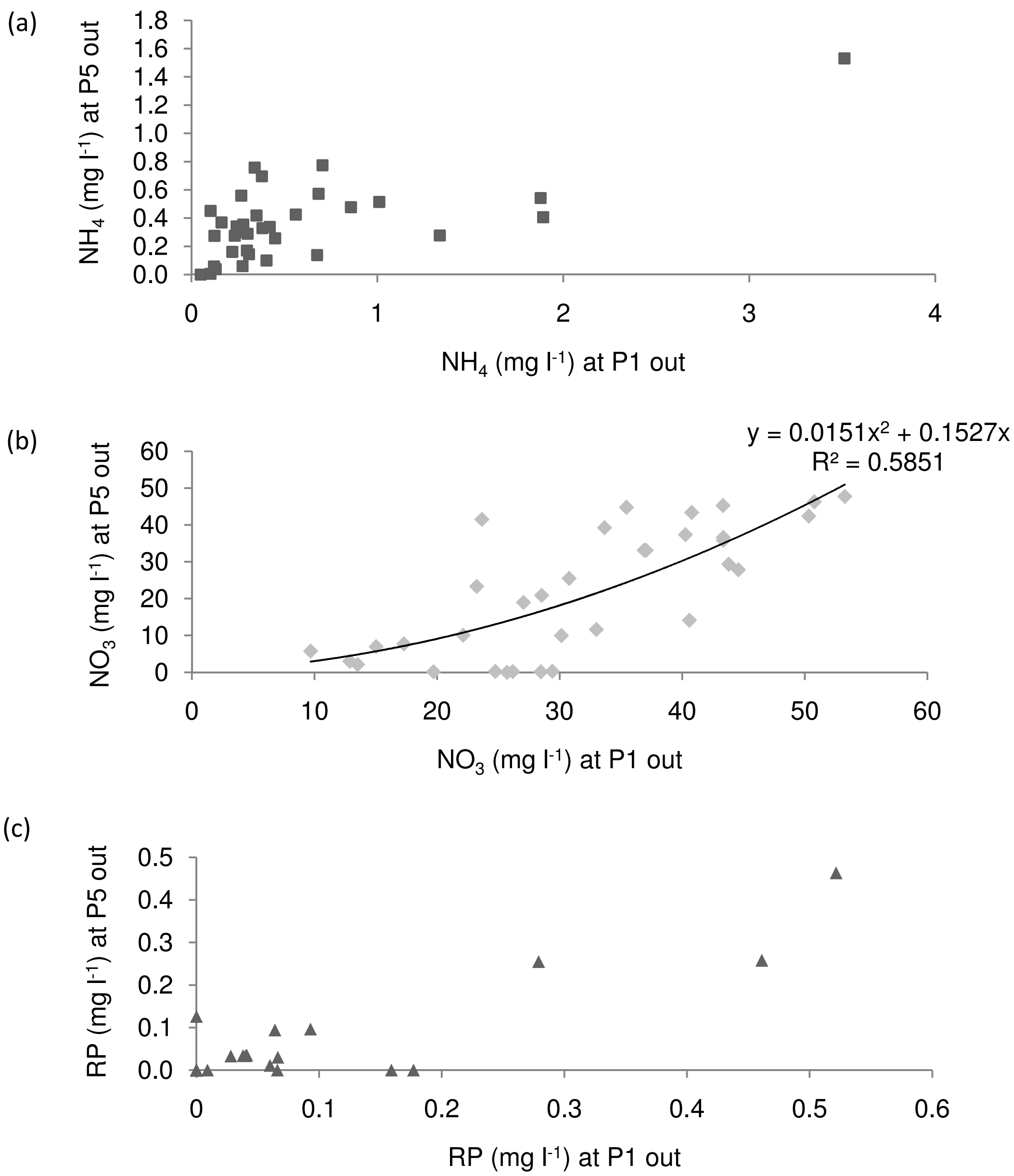


Figure 4.54 Relationships between the concentrations at P1 out and P5 out of (a) NH_4, (b) NO_3 and (c) RP, CFW2.

4.5.3.8 Correlations between water quality parameters at the outlet

As for CFW1, correlations between NH_4, NO_3, RP, BOD_5 and pH at the outlet were investigated in order to identify if some parameters could be predicted using others as surrogates but no significant correlations were found.

4.5.4 Key results for CFW2

- At CFW2, the pollutant of main concern was NO_3 contained in field drainage and shallow groundwater. Farmyard runoff did not seem to contribute significantly to pollution, or was maybe not collected properly, or was diluted during the long distance travelled from the farmyard. Concentrations of NH_4, BOD_5 and RP were low all along the wetland, and close to background concentrations at the outlet.
- Treatment efficiency was relatively good for NO_3 overall (especially in summer), but treatment was very limited in winter when the residence time was smaller and temperature lower, and NO_3 concentration at the outlet of the wetland reached high concentrations close to the drinking water upper limit (c. 50 mg l^{-1}).
- Short-circuiting occurred between ponds (the vegetated area was not fully used for treatment) and the large areas of open water probably did not allow for sufficient denitrification to occur at low temperatures.

4.6 Discussion

4.6.1 Influence of weather and farm practices on runoff characteristics

This study presented flow and water chemistry results from a two year monitoring period at two contrasting CFWs located in the Scottish Borders in the Tweed Catchment, used to treat farm and field drainage. Flow measurements confirmed that there were several different inputs into the systems, some being continuous (e.g. septic tank overflow or groundwater), while others were sporadic and linked directly to precipitation over impervious and non impervious surfaces. At the two farms, the overall quantity and quality of runoff entering the wetlands varied during the year, mainly due to the changes in the contribution of field drainage compared to farmyard runoff or septic tank overflow.

At CFW2, periods of no rainfall were dominated by small but continuous inputs from septic tank overflow (higher inputs in summer) expected to contain significant concentrations of BOD_5, NH_4, and RP, and groundwater inputs, enriched in NO_3. Rainy periods were dominated by nitrate-rich field drainage (c. 50 mg l^{-1}) and farmyard runoff. The combined influent did not appear to contain high concentrations of NH_4 and RP. At CFW1, dry periods were characterized by both septic tank overflow and nitrate rich groundwater (c. 20 mg l^{-1} NO_3) inputs, while rainy periods were associated with large farmyard and field inputs.

Volumes of runoff leaving the wetlands were estimated to be between 10% (CFW1) and 30% higher (CFW2) in autumn/winter than in spring/summer, due to high rainfall, low evapotranspiration rates and small retention capacity of the fields which were saturated over longer periods in autumn/winter. High rainfall inputs in summer 2007 led to field saturation and large inputs during this period. Highest peak flows (up to 60 l s^{-1}) were measured at CFW1 during heavy rainfall, probably explained by the larger impervious areas and shorter distance from the farmyard to the CFW.

In this study, the measured concentrations of pollutants in the runoff were much lower than have been reported in other studies of farmyard runoff, due to the strong dilution by relatively clean areas such as roofs, which represented 40% of the impervious surfaces at CFW1. Others authors have shown that much more concentrated effluents can arise, with concentrations reaching for example 5000 mg l^{-1} for BOD_5 and 500 mg l^{-1} for NH_4 (Brewer *et al.*, 1999; Cumby *et al.*, 1999; Harrington *et al*, 2005; Edwards *et al.*, 2008). In autumn and winter, at CFW1, concentrations of BOD_5, NH_4 and NO_3 in the farm and field drainage were significantly higher than in summer, but no significant difference appeared for RP.

The volume and quality of runoff generated on a farm has been shown to depend on several factors including climate, farm type (e.g. arable, dairy or mixed), impervious area, manure and silage management and farmyard maintenance (Cumby *et al.*, 1999; Edwards *et al.*, 2008). Here, changes in runoff quality were mainly explained by the origin of the runoff (e.g. field drainage enriched in NO_3 and sometimes BOD_5

and NH_4, after slurry application, e.g. in January 2008), timing and intensity of rainfall, surface of impervious area, and by farm type, farming activities and animal numbers (450 dairy cattle in Farm 1, and 130 beef cattle in Farm 2).

The influence of antecedent rainfall was noticeable at the inlet and outlet of CFW1, and at the outlet of CFW2 for RP only. At CFW1, higher concentrations of NO_3 at inlet 1 and of RP at inlet 2 were positively correlated with higher amounts of 2-day antecedent rainfall, which could be associated with wash-off of nutrients in the fields. At the outlet of CFW1, higher concentrations of NH_4 and RP were associated with larger 5-day antecedent rainfall, related to the larger inputs from the farmyard during storm events and lack of time for treatment. At CFW2, RP concentration at outlet appeared positively correlated to 5-day antecedent rainfall.

The effect of farm type and activity was significant, which has been documented previously (Cumby *et al.*, 1999; Edwards *et al.*, 2008). Indeed, in the dairy farm at CFW1, the daily movement of cattle over impervious areas between paddocks and milking parlour, and subsequent deposition of fresh faeces and urine, created a permanent source of easily mobilised contaminants. In contrast, in the mixed farm, cattle were either kept inside or moved to the fields for long periods, and feeding areas were roofed, which limited yard contamination, and wastes consisted of drier materials (e.g. faeces mixed with straw). Farmyard maintenance might have influenced the quality of the runoff. Scraping (twice daily at Farm 1, and occasional at Farm 2) reduced the accumulation of faeces (and urine to a lesser extent) between rainfall events and therefore limited quantities washed out. Other activities involving replacement of the bedding and emptying of slurry stores and middens could also have increased the risk of pollutant runoff. Slurry spreading on its own was also shown to contribute to the diffuse pollution issue on the dairy farm. High concentrations of NH_4 (35 mg l^{-1}), NO_3 (21 mg l^{-1}) and RP (4.8 mg l^{-1}) were found in the ditch upstream from the pond at CFW1 after heavy rainfall following slurry application in January 2008.

Since large quantities of polluted water could bypass CFWs, locating fewer but larger wetlands further downstream from agricultural catchments could be a more effective option for addressing diffuse pollution. Such wetlands could indeed catch larger quantities of polluted runoff but, on the other hand, their efficiency could be lowered, due to dilution of wastewater. Finding enough area of land to build large enough wetlands could also be challenging.

The lack of significant correlations between flow and concentration of pollutants at the inlet of both CFWs could be explained by the mixing of point and diffuse sources, e.g. low flows of septic tank overflow with high concentrations of pollutants and high flows of diluted farmyard runoff. However, a tendency for concentrations to decrease at high flow was observed, suggesting dilution by roof runoff or field drainage and a wash out effect at the beginning of the rainfall. At the outlet of both CFWs, there was also no correlation between flow and concentration, due to the very short-term flow fluctuations in the ponds which have small residence times.

The individuality of each site, the multiple sources of runoff, and discrepancies between intended and actual flows entering CFWs, make it difficult to predict water quality and flow into the systems and contaminant loads, unless considerable investment is made to separate field drainage from farm and roof runoff, and to divert field drains and lateral inputs.

The form in which the different pollutants were transported was shown to be influenced by the origin of the pollutant. Concentrations of OP and RP were higher in the inflow to CFW1 and were attributed to greater presence of cattle dung and slurry on this farm. Septic tank overflow appeared to contain a large fraction of RP, which has been confirmed by more intensive surveys in other studies (Grant and Moodie, 1997). Field drainage at CFW2 was shown to contain lower concentrations of RP, < 1 mg l^{-1}, with the highest concentrations occurring at higher flow.

Elevated NO_3 concentrations in groundwater (although below the drinking water standard) is of concern, especially since both the CFWs are within NVZs. Even if only farmyard runoff was treated, seepage of groundwater into watercourses would still occur, suggesting that source control measures to minimise nitrate leaching are important, e.g. fertilization and slurry spreading management.

At CFW1 OP accounted for a significant fraction of the TP concentration (on average 52% at inlet 1, 67% at inlet 2, 20% at outlet), due to the source of P being predominantly animal wastes. The higher proportion of nitrite compared to nitrate, and higher concentrations of NH_4 in the samples at CFW1 also indicated animal waste contribution. In contrast, at CFW2, OP only accounted for a small fraction of the TP (< 10%) at the inlet, due to large input from field drainage, richer in phosphate, and NH_4 and nitrite concentrations were also quite low in the inflows, reflecting the smaller impact of farm wastes and slower mobilization.

In this study, no distinct pattern was identified regarding the forms of P mobilised during the year. However, many studies have shown that the contribution of different forms of P can vary significantly over time and depends on the catchment under study (Johnes and Hodgkinson, 1998).

4.6.2 Influence of design and climate on water treatment efficiency

Water treatment efficiency by mass at CFW1 was shown to be very good for BOD_5 (92%), although outlet concentration was often > 20 mg l^{-1}, and for SS (91%), relatively good for NO_3 (c. 80%) and NH_4 (c. 65%) but only limited for RP (c. 45%) whose outlet concentration remained between 1 and 2 mg l^{-1} during most of the year, well above the river water quality standard. Regarding FIOs, a significant reduction occurred in coliforms and *streptococci* counts, but more samples are needed to assess the real removal efficiency. At CFW1, no treatment occurred within the 30 m long swale for any of the pollutants due to very limited residence time, but the swale may have helped reduce flow velocity during storm events, attenuating the resuspension of sediment within the pond.

CFW1 retained large quantities of all pollutants, 2340 kg ha yr^{-1} NH_4, 2860 kg ha^{-1} yr^{-1} NO_3, 111 kg ha^{-1} yr^{-1} RP, 23 200 kg ha yr^{-1} BOD_5 and 41 500 kg ha yr^{-1} SS, larger than average quantities mentioned by DeBusk (1999c) for surface and subsurface constructed wetlands (e.g. 2740 kg ha^{-1} yr^{-1} BOD_5, 44 kg ha^{-1} yr^{-1} RP), and only slightly lower than TP removal (128 kg ha^{-1} yr^{-1}) obtained in a 1.2 ha CW in Northern Ireland (Bob Foy, *pers. comm.*). This could suggest an overestimation of the treatment efficiency, explained by the young age of the CFWs (sediment in small quantities, more P adsorption sites available), the dilution of pond water by groundwater, the limited number of samples, especially during storm events (large concentration fluctuations may occur over short time periods at the outlet), limited duration of the study (compared to CFW life span), underestimation of outflow volume and concentrations, and unaccounted losses by infiltration. As underlined by Johnes (2007), limitations in the water sampling methodology and frequency (e.g. monthly sampling) usually result in large uncertainties in nutrient load estimates. Although removal efficiencies appeared good for all pollutants, the outflow from CFW1 still contained significant amounts of pollutants, i.e. c. 217 kg yr^{-1} NH_4, 161 kg yr^{-1} NO_3, 30 kg yr^{-1} RP, 448 kg yr^{-1} BOD_5 and 918 kg yr^{-1} SS.

CFW2, in contrast, retained smaller quantities of NH_4 (80 kg ha^{-1} yr^{-1}) and RP (10.7 kg ha^{-1} yr^{-1}) but larger quantities of NO_3 (4310 kg ha^{-1} yr^{-1}). Nevertheless, significant quantities of NO_3 (2600 kg yr^{-1} NO_3) and less of the other pollutants (43.8 kg yr^{-1} NH_4, 6.5 kg yr^{-1} RP and 1007 kg yr^{-1} SS) remained in the outflow. Nitrate was intercepted efficiently, especially in summer, when reduction by mass reached 69%. Treatment efficiencies for other pollutants were not very meaningful since concentrations were close to background concentrations all along the system. However, in winter, when temperature was lower and residence time smaller, NO_3 removal was low (c. 2%) and NO_3 concentration remained high, although below the drinking water standard (50 mg l^{-1}). This may suggest, in spring and summer, the influence of both a greater microbial activity and a longer residence time in the wetland enhancing treatment, which has been suggested by other studies at the same site (Reay and Paul, 2008; Rao Pangala, 2008).

In summer 2007 and 2008, after prolonged heavy rainfall periods, NO_3 concentration reached up to 40 mg l^{-1} at the outlet, illustrating the importance of residence time as a limiting factor for treatment. A surprisingly high concentration of FC was observed at CFW2 in August 2007, probably explained by faeces introduced by five swans, and several moorhens and ducks in the last pond.

Several authors have shown that low temperatures can impede treatment by slowing biological processes and plant uptake. For example, Newman *et al. (*2000) who assessed the efficiency of a surface flow wetland (400 m^2) used to treat parlour washing found that mean mass removal efficiencies dropped between summer and winter from 31% to 7% for NH_4-N, from 68% to 54% for RP, 79% to 33% for BOD_5. Reddy *et al.* (2001) and Poach *et al. (*2004b) who worked on marsh-pond-marsh systems used to treat swine wastewater found significantly higher treatment during warmer months. However, in this study, although seasonal differences in efficiency were noted, the influence of temperature alone on concentration fluctuations could not be identified, treatment occurring at both low and higher temperatures. The temperature effect may have been masked by the effect of the residence time, which was strongly reduced in winter.

These results are comparable to those obtained in other studies in the UK, Finland, New Zealand and the USA (Crumpton *et al*., 1993; Reed *et al*., 1995; Kadlec and Knight, 1996; Tanner *et al*., 2000; Koskiaho *et al.*, 2003; Dunne and Culleton, 2004; Forbes *et al*., 2004; Poach *et al*., 2004b; Vymazal, 2005; Shilton, 2006), where research has shown that unvegetated ponds and wetlands are most efficient at removing BOD_5, SS, FC (often > 90% removal by mass) and NH_4 (they promote nitrification) from wastewater, and usually less efficient for nitrate (very mobile form), total nitrogen and phosphorus (30% to 60%). For example, Koskiaho *et al.* (2003) found a maximum retention of 40% for TN and 33% for RP in three Finish CWs treating agricultural runoff, and Uusi-Kämpä *et al.* (2000) reported c. 40% TP reduction. However, in this study, NO_3 mass reduction was higher at CFW1 (80%) than at CFW2 (43%), maybe due to anaerobic conditions enhancing denitrification in the deeper parts of the ponds, and to a good carbon supply (Baker, 1998).

Poach *et al.* (2004b) reported treatment efficiencies for a marsh-pond-marsh system treating swine wastewater, of 35–51% for TSS, 30–50% for Chemical Oxygen Demand (COD), 37–51% for TN and 13–26% for TP, finding lower efficiencies in winter for COD and N. However, they could not conclude on the impact of the pond section on the overall nitrogen removal.

Research in Norway and Finland showed good treatment efficiencies for P and solids under cold conditions within small farm wetlands (Braskerud, 2002a and 2002b). A high TP removal of 100 g m^{-2} yr^{-1} was measured, higher than RP removal at CFW1 (11.1 g m^{-2} yr^{-1}) or CFW2 (1.1 g m^{-1} yr^{-1}), the sorption of P being enhanced by high redox potentials and aerobic conditions (Braskerud *et al.*, 2005).

BOD_5 concentration in the outflow at CFW1 was significantly higher in summer than in winter, while inputs were higher in winter (higher average concentration and larger volumes), which resulted in a higher winter treatment efficiency. BOD_5 removal in the pond could be due to dilution, aerobic conditions favouring bacterial activity, and also to the presence of organic matter feeders such as cladocerans (e.g. *Daphnia magnum,* observed in huge quantities in spring and summer). However, warmer summer periods were also characterised by the growth of green algae which might explain higher BOD concentration at this time, since samples were unfiltered, as documented in other studies (e.g. Cathcart *et al.*, 1994).

The mass removal of SS in both wetlands was very good (> 85%) and comparable to literature data (Kadlec and Knight, 1996; Dunne and Culleton, 2004), with only limited concentration increase at the outlet during storm events, and is confirmed by sediment accumulation in both CFWs between 2006 and 2008. Sediment removal is expected to increase with vegetation colonization which is expected to favour filtration, sedimentation, and to impede resuspension (Craft, 1997; Braskerud, 2002c). However, sediment accumulation will reduce residence time and the risk of resuspension could increase, suggesting the need for sediment removal after a few years (Carty et *al.*, 2008b). After being dredged, sediment can be spread on land and be used as a source of nutrients.

Ammonium concentration decreased in both CFWs, mean reduction of 42% at CFW1 and 34% at CFW2 for the monitoring period, but reduction showed a strong seasonal pattern, with efficiencies in summer and winter respectively of 65% and 17% at CFW1 and 50% and < 0% at CFW2. Nitrification of NH_4 depends on the presence of aerobic conditions, which might occur in the shallowest zones of the ponds due to oxygen diffusion, and water mixing. Vegetation, especially *P. australis*, has been shown to contribute to water oxygenation, by diffusion through aerenchymatous tissues or roots (Furniss, 1992) and it also provides a substrate for the growth of microorganisms and enhances their contact with wastewater. Ammonium removal is therefore expected to increase with the increase in vegetation cover. However, the impact of the vegetation alone in the CFWs could not be assessed due to the limited duration of the study, and limited vegetation colonization at CFW1 and low NH_4 concentrations at CFW2. In fact, in the CFWs studied, the colonization of vegetation from the edges could cause short-circuiting between inlet and outlet, i.e. water will flow through the areas of least resistance, and reduction in treatment volume, time and efficiency.

The limited size of CFW1 (volume of 1500 m^3) compared to the actual inputs results in a relatively limited residence time (calculated as the mean outflow divided by the pond volume) of the water within the pond, i.e. c. 24 days on average, but only 1 to 5 days during periods with heavy rain, which did not allow for full treatment. A large part of the treatment efficiency by concentration between inlet and outlet may be due to dilution (daily inflow from milk cooling water and rainfall) and efficiency by mass could be over-estimated by omitting infiltration of contaminated water.

Design issues

This study identified several issues related to the design of the wetlands and to the inadequacies between their intended and actual design, implementation and use, which have been mentioned by several authors (e.g. Nuttall *et al*, 1998; Stewart, 2008). Indeed, both the wetlands investigated were young, and appeared too small for the actual volumes of wastewater they receive during rainy periods.

CFW1 was not planted in contrast to plans and vegetation colonization was very slow and localised, it received additional inputs daily and had a rather large outlet (> 16 cm diameter). Rapid mixing occurred within the single pond and contact with vegetation was limited, resulting in a suboptimal treatment during storm events. Hydraulic efficiency and adequate residence time have indeed been shown to be essential elements of robust and effective systems (Kadlec and Knight, 1996; Dunne *et al.*, 2005; Carty *et al.*, 2008a).

At CFW2, the segmentation of the wetland into several cells, coupled with the combination of dense vegetation stands and open water areas seemed to promote efficient plant uptake, bacterial assimilation and denitrification in spring and summer, when residence time was longer and temperature higher. However, the limited earthworks did not provide adequate levelling, and hence, preferential flow occurred between ponds and untreated water escaped the wetland laterally. This limited the contact between vegetated areas and wastewater, and reduced residence time, resulting in a low treatment efficiency at this time. Experience from Ireland mentions that wetland area should be at least 1.3 to 2 times the interception area (impermeable surface), to allow for sufficient residence time and treatment (> 90 % mass removal) (Dunne *et al.*, 2005), which suggests a size of at least 3.0 ha for CFW1 and 2.3 ha for CFW2, if only runoff from impermeable areas were treated.

In both CFWs, the presence of storm overflows close to the pond inlet at CFW1 and in P5 at CFW2, might have had a negative impact on the water environment, by letting escape contaminated water before treatment. A single large (e.g. 30 cm in diameter) vertical perforated pipe located at the outlet (where water quality is higher) would play better the role of storm overflow.

4.6.3 Compliance with river standards and impacts on receiving water courses

Assessing the real "efficiency" of CWs is often very difficult due to the openness of the systems and inaccuracies in the water balance, and because outputs are delayed compared to inputs. Average outflow concentration when sampling frequency is low might be considered as a poor measure of efficiency, due to the high probability of missing storm events and possible pollutant release. Moreover, water treatment efficiencies do not always indicate that the system complies with environmental targets. Loadings and concentrations can indeed be reduced between inlet and outlet, but outlet concentrations can still exceed environmental targets.

The two constructed wetlands investigated released effluents with concentrations above those found in rivers. The effluent leaving CFW1 frequently contained excessive concentrations of NH_4 (median of 9 mg l^{-1}), RP (median of 1.4 mg l^{-1}, above the 0.01 mg l^{-1} recommended for rivers) and BOD_5 (median of 18 mg l^{-1}, corresponding to poor river water quality), although NO_3 (median 3.5 mg l^{-1}) was always below the drinking water standard (50 mg l^{-1}). At CFW2, NO_3 was the only pollutant of concern, although the outlet concentration (median 21 mg l^{-1}) rarely exceeded 50 mg l^{-1}.

The impacts of farm runoff and CFW effluent on river or lake water quality do not only depend on concentration but also on the receiving waterbody and on its assimilative capacity, which can be defined as the ability of an aquatic ecosystem to assimilate substances (e.g. anthropogenic wastes) at certain concentrations without degrading or damaging its ecological integrity (Cairns, 1977; Cairns, 1998; Richardson and Qian, 1999). Assimilative capacity is higher in autumn and winter when larger volumes are present and when dilution occurs, and therefore a low treatment efficiency at this time, might not be as crucial an issue as in summer, when the impact of even a small flow with high concentrations can have very deleterious effects on the more active aquatic wildlife. The environmental impact of CFW1 is expected to be high in summer, due to daily inputs of nitrate enriched groundwater pushing out pulses of contaminated water.

Due to differences in assimilative capacity of waterbodies, difficulties appear in setting quality norms for the effluent discharged. Questions arise whether to use risk approaches at the farm or catchment level or to propose site-specific targets. One approach that has been developed in the USA to address this issue is the Total

Maximum Daily Load (TMDL) (explained in more detail in Chapter 7), which is the calculation of watershed budgets for pollutant influx to watercourses, based on scientific investigation assessing the amount of pollutant that could be assimilated by a waterbody without deleterious effects (USDA, 2009).

4.7 Conclusions

The hydrological monitoring of the two constructed wetlands (especially since their construction was not intended for research purposes) was challenging, as suggested by the literature (Kadlec and Knight, 1996) and personal experience (David Kay, *pers. comm.*). The main difficulties were linked to their "openness" (diffuse inputs and outputs), variable design, variable flow patterns (mainly driven by storm events), and the interaction of biofilm growth and animals with sensors.

Results showed that the two CFWs studied improve water quality overall and are therefore better than not taking any action. However, their design and/or inappropriate use do not allow sufficient treatment. Water quality at their outlet is still of concern, especially in winter, when pollutants inputs are larger, temperature is lower and residence time smaller. Increasing residence time by the use of baffles, and manipulating water level and flow by changing outlet pipe diameter, could help improve treatment.

The long-term performance of the CFWs could not be assessed due to their young age. However, a long-term monitoring of these systems is needed to better understand the changes in treatment efficiency linked to vegetation growth, substrate and sediment accumulation and changes in hydrological characteristics.

Chapter 5: The Ecological Value of Constructed Farm Wetlands

The chapter presents and discusses the results of the two year ecological monitoring programme of two CFWs receiving farmyard and field drainage, and of one non-polluted amenity pond (selected for comparison with the CFWs), all located in the Scottish Borders. It focuses on their ecological value, i.e. mainly aquatic plants and macroinvertebrates, and on the links between design, water quality and ecology of the ponds. It finally proposes recommendations for the design of CFWs to optimize biodiversity while ensuring efficient water treatment.

5.1 Introduction

Freshwater ponds and wetlands, whether originating from erosion, glaciation, material extraction or constructed for recreation or animal watering, can host a great diversity of plants and animals when not heavily impacted by anthropogenic pollution (O'Connor and Shrubb, 1986; Biggs *et al*., 1994; Froneman *et al*., 2000). Their ecological value depends on a wide range of climatic, physical, chemical and structural factors, such as their location in the landscape and connectivity with other wetlands, size, depth and structural complexity, hydrological characteristics and water chemistry (Oertli *et al*., 2002; Williams *et al*., 2003; Nicolet *et al*., 2004; Batty *et al*., 2005). Their loss during the last century, mainly due to agricultural drainage and water abstraction, is raising great concern with regards to the loss of biodiversity and environmental services (Costanza *et al*., 1997; Mitsch and Gosselink, 2000; Otte, 2003; Schuyt and Brander, 2004). Wetland protection is a priority under the Habitats Directive and WFD (Moser *et al*., 1996; OECD and IUCN, 1996), but the preservation and restoration of small ponds is still overlooked (Davies *et al*., 2008).

During the last decades, wetlands have been created in rural and urban areas to mitigate water pollution and recreate lost habitats (Woods *et al*., 2003; Woods-Ballard *et al*., 2005; Carty *et al*., 2008a). A few surveys conducted in wetlands treating mine water, urban or agricultural runoff indicate that, although animal and plant communities are impoverished in those systems (Surrency, 1993; Brown *et al.*, 1997; Brown and Batzer, 2001; Balcombe *et al.*, 2005; Batty *et al.*, 2005; Petranka *et al.*, 2007), their potential for biodiversity conservation is not negligible, but is strongly constrained by water quality, design and management. Indeed, exposure to nutrients, sediment, metals or pesticides affects the fertility and survival of aquatic invertebrates, amphibians and plants (Clarke and Baldwin, 2002; Alonso and Camargo, 2003; Christin *et al.*, 2004, Camargo *et al.*, 2005).

Constructed farm wetlands are now promoted by the Scottish Government as a BMP for farmers to deal with contaminated farmyard runoff, and their number is expected to increase. Consequently, it is essential to understand better their impact on wildlife and to develop a design limiting those impacts, to achieve the necessary compromise between water treatment and biodiversity conservation. The aims of the chapter are to: 1) assess the ecological value (plant and macroinvertebrate diversity) of the CFWs and amenity pond; 2) study the link between the ecological value and CFW design, habitat quality and water quality; 3) propose recommendations to enhance the ecological value while ensuring efficient water treatment.

5.2 Materials and methods

5.2.1 Study sites

The study involved the ecological, water quality and sediment monitoring of seven ponds between February 2006 and June 2008. One of the ponds was a recreational one (amenity pond or "AP") (Figure 3.1, Chapter 3), and the six others were used for treatment of farmyard runoff, field drainage and septic tank overflow, and are referred to as K (a single pond part of CFW1) (Figure 3.1) and P1 to P5 (five ponds in series at CFW2) (Figure 3.6). A full description of the CFWs and the farms on which they were located is provided in Chapter 3. Table 5.1 summarizes the main physical characteristics of the ponds. Sediment depth was assessed from sediment cores taken with a cylindrical hand corer in the different ponds in April 2008, i.e. between 3 and 6 samples per pond, close to inlet, middle and outlet, to account for spatial heterogeneity, and corresponds to the depth above the basal clay layer.

Table 5.1 Physical characteristics of the ponds studied.

	Amenity Pond (AP)	CFW1	CFW2				
		K	P1	P2	P3	P4	P5
Distance (m) from CFW inlet to pond's centre	-	60	7	77	108	144	220
Main use	Amenity	Water treatment					
Source of water	Rainfall	Farmyard runoff and field drainage, septic tank overflow, groundwater, rainfall					
Age at start of research (y)	10	0.5	1.5	1.5	1.5	1.5	1.5
Altitude (m asl)	102	95	65	65	65	65	65
Land use before construction	Unimproved grassland	Improved grassland	Unimproved wet grassland				
Surrounding land use + boundaries	Amenity area, buildings, roads (no nearby waterbody)	Improved grassland; ditch at 10 m	Unimproved grassland, fallow (shrubs and trees); river c. 50 m away				
Grazing	NO (mowing)	NO (fenced)	NO (fenced but cattle sometimes enter)				
Soil Series	-	Whitsome	Hobkirk / Eckford				
Surface area (m^2)	2500	2200	50	115	105	190	2500
Emergent plant cover (%)	10	2	80	30	30	30	30
Average water depth (m)	1	0.8	0.5	0.7	0.7	0.7	0.8
Max water depth (m)	1.5	1.5	1.5	1.0	0.8	0.8	1.7
Mean sediment depth in 2008 (cm) (± SD)	> 10	8 (± 5)	11 (± 5)	12 (± 8)	12 (± 6)	7 (± 2)	6 (± 4)
Distance (m) to closest wetland	250	250	440	400	380	360	300
Shade by trees	Yes	NO	NO	NO	NO	NO	NO

- Data not available

The amenity pond (AP) was vegetated on its edges by *Phragmites australis* (L.), *Typha latifolia* (L.) and several other macrophytes (e.g. *Eriophorum angustifolium* (Honckeny), *Montia fontana* (L.)). Emergent plants such as *Alisma plantago aquatica* (L.), and *Nymphaea* sp. (L.) (ornamental) and submerged species such as the invasive *Lagarosiphon major* (Ridley) and *Myriophyllum spicatum* (L.) were abundant, covering most of the pond's bottom. A small vegetated island was left in the centre of the pond, and boulders were present on the southern side, increasing habitat structural complexity. Several fish species (e.g. cyprinids) were introduced. Pollutant concentrations measured in September 2006, November 2007 and May 2008, were relatively low compared to the CFWs, i.e. below 0.5 mg l^{-1} for NH_4, 0.04 mg l^{-1} for NO_3, 0.10 mg l^{-1} for RP and 4 mg l^{-1} for BOD_5.

K was partly vegetated on its edges, predominantly by *T. latifolia* and *Juncus effusus* (L.). It was not planted immediately after construction in 2005, but *T. latifolia* from the amenity pond was transplanted in 2006, with limited success. Abundance and richness of submerged aquatic plants were still very low in 2008. The pond was fenced to exclude cattle grazing and was surrounded by improved grassland. It is hypertrophic, with concentrations below 37 mg l^{-1} for NH_4, 47 mg l^{-1} for NO_3, 2.6 mg l^{-1} for RP and 50 mg l^{-1} for BOD_5, and outflow water pH fluctuates between 7.3 and 9.5.

P1, P2, P3, P4 and P5 were linked and part of CFW2, and were eutrophic. Within the wetland overall, the mean concentrations of NH_4 (0.5 mg l^{-1}), BOD_5 (1 mg l^{-1}) and RP and TP (0.1 mg l^{-1}) were rather low, close to background concentrations, but NO_3 concentration was relatively high (20 to 30 mg l^{-1} on average) and has reached up to 70 mg l^{-1} after rainy periods, especially in winter. The wetland was originally planted with *P. australis* and a few other species such as *Nuphar lutea* (L.) and *Iris pseudacorus* (L.), and is now dominated by *P. australis* and *J. effusus*. It is used as a breeding and feeding site by a variety of mammals (e.g. *Mustela nivalis*), insects (e.g. dragonflies, butterflies) and birds, such as mallards (*Anas platyrhynchos*), mute swans (*Cygnus olor*), moorhens (*Gallinula chloropus*), coots (*Fulicula atra*) and grey herons (*Ardea cinerea*).

5.2.2 Water chemistry monitoring

Water chemistry monitoring was conducted in the CFWs from March 2006 to June 2008 (methodology and results in Chapter 4), involving grab and automatic sampling during periods of low and high flow. Water samples were taken in clean single-use vials from the inlet to the outlet, transported in a cool box and stored in the fridge or frozen until analysis for NH_4, NO_3 and RP. On three occasions within the amenity pond, 200 ml water samples were taken in three different places within the pond, mixed together to obtain a composite sample and later analysed for the same parameters as well as for BOD_5, SS, pH, temperature and conductivity.

5.2.3 Vegetation surveys

Vegetation surveys involving walking the edges and wading in the ponds, were carried out once a year between June and August 2006 and 2007 to identify emergent, floating-leaved and submerged plants within the wet perimeter of the ponds, as well as trees and plants around them (referred to as “wetland area”), following the methodology of the National Pond Survey (NPS) proposed by Pond Action (1998). Plants were identified in the field or later in the laboratory using appropriate keys (Fitter *et al.*, 1984 and 1996; Rose and O’Reilly, 2006; Greenhalgh & Ovenden, 2007). The nomenclature follows Rose and O’Reilly (2006). The percentage vegetation cover within the outer margins of the ponds was assessed visually each year using the following cover classes: < 1%, < 5%, 5% to 25%, 25% to 50%, 50 to 75% and > 75%.

5.2.4 Macroinvertebrate and wildlife surveys

Aquatic macroinvertebrates were surveyed in spring, summer and autumn to account for seasonal fluctuations in invertebrate diversity, on three occasions in AP (14/09/06, 10/05/07, 2/08/07) and on four occasions in K (27/06/06, 14/09/06, 18/04/07, 2/08/07) and P1 to P5 (on 03/07/06, 26/09/06, 10/05/07, 2/08/07). Sampling followed the technique used for the NPS (Pond Action, 1998). It involved three minutes “kick sampling” with a standard 1 mm mesh pond net (frame size: 0.26 m x 0.30 m), the three minutes being divided between the main mesohabitats identified in a preliminary survey (Appendix A), and a one minute additional search in and around the pond (under stones, around logs, on the water surface). Samples were transferred into plastic bags, stored in a cool box and sorted live within 24 h of collection. Samples were sieved (1 mm sieves) to remove excess fine sediment and ease the identification process, small amounts were transferred to white trays, and invertebrates were counted and identified alive or stored in ethanol (70%) until identification. Identification to family and species level when possible, involved the use of a stereo-microscope with magnification of up to x100, keys from the Freshwater Biological Association (Hynes, 1977; Macan, 1977; Elliott and Mann, 1979; Elliott and Humpesch, 1983; Savage, 1989; Gledhill *et al.*, 1993; Edington and Hildrew, 1995; Elliot, 1996; Savage, 1999; Wallace *et al.*, 2003) and from Croft (1986), and support from Dr. Rob Briers (Napier University, Edinburgh). Findings were compared with data on species distribution across the UK (e.g. National Biodiversity Network). Abundance, richness, diversity indices, rarity indices and biotic indices (e.g. ASPT and BMWP Score) were calculated. Fish, adult insects, amphibians, mammals and birds were also recorded with notes on their numbers and behaviour, but no formal surveys were carried out. Overall pond conditions were also noted including algal growth, odours, presence of oil, technical problems and invasive species.

5.2.5 Data analysis

5.2.5.1 Pond water chemistry

To identify any differences in water chemistry characteristics between the ponds, non parametric Kruskal-Wallis (95% probability level) tests were conducted (on untransformed data) using Infostat Professional Version 2.0.

5.2.5.2 Vegetation diversity and habitat quality

Plant richness and percentage cover were assessed from the field surveys and compared between the ponds, and changes over time were examined, to assess colonization patterns, structural changes, and influence of the initial design and of surrounding habitats.

Species Rarity Index scores (SRI), derived from the Species Quality Score developed by Foster *et al*. (1990) and commonly used for pond surveys (Oertli, 2002; Nicolet *et al*., 2004), and the Conservation Value (CV) derived from species richness and rarity index, were used to characterize the ponds, the SRI for a given pond being obtained by averaging SRI scores of all species (Table 5.2).

Table 5.2 Great Britain Species Conservation Status and Rarity Scores (Eversham, 1983; Wells *et al*., 1983; Shirt, 1987; Pond Action, 2002; Nicolet *et al*., 2004).

Status	Rarity Score	Meaning
Common	1	Recorded from > 700, 10x10 km grid squares in Britain
Local	2	Invertebrates: either (a) confined to certain limited geographical areas, where populations may be common or (b) of widespread distribution, but with few populations Plants: recorded from between 101 and 700, 10x10 km grid squares in Britain
Notable Species (Nationally Scarce)	4	Recorded from 16–100, 10x10 km grid squares in Britain (Notable A: < 30; Notable B: 31-100)
Near Threatened and RDB CD	8	Recorded from 15 or fewer 10x10 km grid squares and Red Data Book (Conservation Dependent)
RDB VU	16	Red Data Book (Vulnerable)
RDB EN, CR	32	Red Data Book (Endangered and Critically Endangered)

The overall conservation value of the ponds was derived using the classification in Table 5.3 (Pond Action, 2002), always giving the pond the highest conservation category using any of the measures.

Table 5.3 Conservation Value of Ponds according to wetland plant number and rarity (Pond Action, 2002).

Conservation value	Ecological characteristics
Low	Few wetland plants (≤ 8 species) and no local species (i.e. SRI = 1.00)
Moderate	Below average number of wetland plant species (9-22 species) or SRI of 1.01-1.19
High	Above average number of wetland plant species (≥ 23 species) or a SRI of 1.20-1.49. No Nationally Scarce or Red Data Book (RDB)
Very High	Supports one or more Nationally Scarce or RDB species or a SRI of 1.50 or more, or an exceptionally rich plant assemblage (≥ 40 species)

5.2.5.3 Macroinvertebrate diversity and colonization patterns

Macroinvertebrate richness (number of families or species), abundance (number of individuals of a given family or species) and Shannon Diversity Indices, taking into account richness and evenness, were calculated for each pond (Shannon & Weaver, 1949; Krebs, 1989). Since only three samples were taken in AP, while four samples were taken in the others, total macroinvertebrate abundance for the ponds were expressed as abundance per sample. The Biological Monitoring Working Party Score (BMWP) (Maitland, 1977) was calculated as the sum of the tolerance indices (which range from 1 to 10, 10 referring to a very pollution-sensitive family) of all invertebrate families found within each pond. The Average Score Per Taxon (ASPT), independent of the sample size, was then obtained by dividing the BMWP score by the number of scoring families. Additionally, the SRI for invertebrates was calculated for each pond and each pond was given a conservation value according to macroinvertebrates richness and rarity (Table 5.4).

Table 5.4 Conservation value of permanent and semi-permanent lowland ponds (single season 3 minute sample) according to macroinvertebrate richness and rarity (Pond Action, 2002).

Conservation value	Ecological characteristics
Low	Few invertebrate species (0-10 species) and no local species (i.e. SRI = 1.00)
Moderate	Below average number of invertebrate species (11-32 species) or a SRI of 1.01-1.19
High	Above average number of invertebrate species (33-49 species) or a SRI of 1.20-1.49. No Nationally Scarce or Red Data Book (RDB)
Very High	Supports one or more Nationally Scarce or RDB species or a SRI of 1.50 or more, and/or an exceptionally rich invertebrate assemblage (≥ 50 species)

5.2.5.4 Overall ecological value of the ponds

Combining data on plants and aquatic invertebrates, the overall conservation values of the ponds were estimated and compared between the ponds studied and other natural and rural or urban treatment ponds in the UK, including sites monitored by the National Pond Survey (NPS), "Realising Our Potential Award" (ROPA) and Department of the Environment, Transport and the Regions (DETR).

5.2.5.5 Biodiversity and water quality

A cluster analysis was conducted to investigate the similarities between the ponds studied in terms of macroinvertebrate assemblages, using the Jaccard Similarity Coefficient based on species presence-absence and the software BiodiversityPro Version 2 (MacAleece *et al*., 1997). Species assemblages were also studied to relate species composition to water contamination, and to look for key bio-indicators of pollution.

5.3 Results

5.3.1 Chemical characteristics of the ponds

The water monitoring showed that the seven ponds were very different in terms of their physico-chemical characteristics due to their location and purpose (Table 5.5). Kruskal-Wallis tests showed significant differences between the medians for pH (H= 37.95, p< 0.0001), and concentrations of NH_4 (H= 66.36, p< 0.0001), NO_3 (H= 59.94, p< 0.0001), RP (H= 71.58, p< 0.0001) and BOD_5 (H= 26.09, p< 0.0001) between K and P1 to P5.

The amenity pond was the oldest and largest pond in terms of volume, mainly received rain water and had the lowest concentrations of pollutants, corresponding to typical background concentrations reported in ponds and wetlands (IWA, 2000). K was strongly eutrophic, with all pollutants investigated reaching very high concentrations at outlet, of up to 32 mg l^{-1} for NH_4, 45 mg l^{-1} for NO_3, 2.5 mg l^{-1} for RP and 50 mg l^{-1} for BOD_5. K had much higher concentrations of NH_4, RP and BOD_5, but lower concentrations of NO_3 than P1 to P5 throughout the monitoring period. P1 to P5 were strongly impacted by NO_3, whose concentration reached > 60 mg l^{-1} in winter, but there was limited contamination by NH_4 or BOD_5. Nitrate concentration significantly decreased from inlet to outlet, especially in summer when denitrification and vegetation uptake are enhanced.

Table 5.5 Summary of the chemical characteristics of the seven ponds studied.

Water quality parameter		AP	CFW1	CFW2				
			K	P1	P2	P3	P4	P5
Water pH	n	4	29	20	15	14	25	26
	Mean	8.26	7.90	7.82	8.12	8.19	8.20	8.28
	Range	6.4-9.4	7.4-9.5	6.6-9.9	6.8-11	6.9-12	6.8-11	6.9-9.4
Cond. (µs cm^{-1})	n	4	18	26	3	3	3	18
	Mean	264	1180	692	592	595	588	661
	Range	230-299	840-1270	497-735	467-679	484-677	475-673	468-718
NH_4 (mg l^{-1})	n	3	45	36	27	20	32	38
	Mean	0.239	9.58	0.554	0.361	0.435	0.476	0.366
	Range	0.039-0.468	0.011-31.7	0.03-3.51	< 0.01-1.69	< 0.01-2.49	< 0.01-4.37	< 0.01-3.84
NO_3 (mg l^{-1})	n	3	45	36	27	21	32	38
	Mean	0.028	7.13	31.4	29.2	27.1	20.5	21.7
	Range	0.014-0.038	0.090-45.0	9.68-54.2	6.09-52.9	4.53-61.3	0.240-51.9	<0.017-48.1
RP (mg l^{-1})	n	3	36	28	16	16	22	29
	Mean	0.064	1.34	0.078	0.076	0.04	0.10	0.10
	Range	< 0.003-0.102	< 0.003-2.60	< 0.003-1.51	< 0.003-0.513	< 0.003-0.472	< 0.003-1.00	< 0.003-0.987
BOD_5 (mg l^{-1})	n	3	26	12	5	3	13	16
	Mean	2.5	20.2	1.3	1.3	0.0	0.6	2.0
	Range	1-4	10-50	0-10	1-5	1-5	1-5	1-10
SS (mg l^{-1})	n	3	34	20	3	2	9	21
	Mean	2.80	40.5	65.6	2.90	3.00	4.10	7.80
	Range	1.2-4.6	10-160	1-703	2.0-3.6	2.7-3.2	0.3-17	0-55

5.3.2 Vegetation diversity and colonization patterns

The amenity pond hosted the highest number of plants species, including submerged, emergent and floating plants, with more than 40 species recorded in 2006 and 2007 (Table 5.6). Many of these might have been transplanted initially (e.g. *T. latifolia, Nymphaea* sp.*, E. angustifolium*), while others established naturally (e.g. *J. articulatus*). The most dominant were the emergent *T. latifolia, P. australis* and *A. plantago aquatica,* and submerged *L. major*, *Ceratophyllum demersum and M. spicatum.* Species composition appeared stable over the monitoring period.

Table 5.6 Plants found and their percentage cover class within the "wet perimeter" of the amenity pond.

Plant species	% Cover 14/09/06	% Cover 3/08/07
Achillea millefolium (L.)	0	< 1
Alisma lanceolatum (With.)	< 1	< 1
Alisma plantago aquatica (L.)	10	10
Arum maculatum (L.)	0	< 1
Butomus umbellatus (L.)	< 1	< 1
Callitriche brutia (L.)	5	5
Carex flacca (Schreb.)	< 1	< 1
Carex pendula (Huds.)	< 1	< 1
Ceratophyllum demersum (L.)	20	20
Cirsium arvense (L.)	< 1	< 1
Epilobium hirsutum (L.)	< 1	< 1
Epilobium obscurum (Schreb.)	< 1	< 1
Equisetum fluviatile (L.)	< 1	< 1
Eriophorum angustifolium (Honck.)	< 1	< 1
Filipendula ulmaria (L.)	< 1	< 1
Geranium endressii (J. Gay)	< 1	< 1
Iris pseudacorus (L.)	< 1	< 1
Juncus articulatus (L.)	< 1	< 1
Lagarosiphon major (Ridley)	75	75
Lemna minor (L.)	5	5
Lythrum salicaria (L.)	< 1	< 1
Mentha aquatica (L.)	< 1	< 1
Montia fontana (L.)	< 1	< 1
Myosotis scorpioides (L.)	< 1	< 1
Myriophyllum spicatum (L.)	20	20
Nymphaea sp. (L.)	5	5
Persicaria amphibian (L.)	< 1	< 1
Phalaris arundinacea (L.)	< 1	< 1
Phragmites australis (L.)	10	10
Ranunculus lingua (L.)	< 1	< 1

Plant species	% Cover 14/09/06	% Cover 3/08/07
Ranunculus repens (L.)	< 1	< 1
Rumex conglomerates (Murray)	0	< 1
Salix caprea (L.)	< 1	< 1
Salix sp. 2 (L.)	< 1	< 1
Silene dioica (L.)	0	< 1
Sparganium erectum (L.)	< 1	< 1
Sphagnum sp. (L.)	< 1	< 1
Trifolium repens (L.)	< 1	< 1
Typha latifolia (L.)	10	10
Vicia cracca (L.)	< 1	< 1
No. of plant species	38	40
No. of wetland plants species (in 2007)	29	
No. of submerged and floating plants species (in 2007)	9	
Overall rarity score	1.18	
Tree species within 2 m of the pond	*Alnus glutinosa, Betula pendula, Fraxinus excelsior, Ilex aquifolium*, *Rosa canina*, *Salix* spp., *Sambucus nigra*.	

As illustrated in Table 5.7 below, K hosted the lowest diversity and abundance of wetland plants, with 18 species recorded overall, and had the lowest rarity score (1.07). *T. latifolia* and *J. effusus* were the dominant species but covered less than 2%. Although the farmer transplanted *T. latifolia* rhizomes (from AP) all around the pond margins in 2006, it was still very localised, only growing on the east side of the pond as transplants closer to the pond inlet were washed out. The first species to appear in the pond in 2006 were *J. effusus*, *A. plantago aquatica, L. portula*, *M. aquatica*, *M. fontana, Persicaria maculosa, and P. arundinacea*, which were all present in AP located 300 m uphill. A few species appeared in 2007 (e.g. *J. articulatus* and *R. lingua*). Colonization overall was very slow, and most species were clearly outcompeted by *T. latifolia* and *P. australis*, which grew on the pond edges and created shade and an unfavourable habitat for other species, which in 2008, were only represented by a few specimens.

Table 5.7 Plants found and their percentage cover class within the wet perimeter of K (CFW1). The Rarity Score is given for the overall period.

Plant species	% Cover 28/06/06	% Cover 14/09/06	% Cover 03/08/07	% Cover 14/05/08
Agrostis stolonifera (L.)	< 1	< 1	< 1	< 1
Alisma plantago aquatica (L.)	0	< 1	< 1	< 1
Alopecurus geniculatus (L.)	0	< 1	< 1	< 1
Iris pseudacorus (L.)	< 1	< 1	< 1	< 1
Juncus articulatus (L.)	0	0	< 1	< 1
Juncus effusus (L.)	0	< 1	< 1	<2
Lythrum portula (L.)	0	< 1	< 1	< 1
Mentha aquatica (L.)	0	< 1	< 1	< 1
Montia fontana (L.)	0	< 1	< 1	< 1
Myosotis scorpioides (L.)	0	< 1	< 1	< 1
Persicaria maculosa (L.)	0	< 1	< 1	< 1
Phalaris arundinacea (L.)	0	< 1	< 1	< 1
Phragmites australis (L.)	0	< 1	< 1	< 1
Polygonum aviculare (L.)	0	< 1	< 1	< 1
Ranunculus lingua (L.)	0	0	< 1	< 1
Ranunculus repens (L.)	0	< 1	< 1	< 1
Rumex obtusifolius (L.)	0	< 1	< 1	< 1
Salix sp. (L.)	0	< 1	Absent	Absent
Typha latifolia (L.)	< 1	< 1	1	2
No. of plant species	3	17	18	18
No. of wetland plant species	14			
No. of submerged and floating plant species	0			
Rarity score in 2008	1.07			
Plant species within 2 m of the pond	*Agrostis stolonifera, Alopecurus geniculatus, Cirsium arvense, Cirsium vulgare, Galeopsis tetrahit, Galium aparine, Geranium dissectum, Holcus lanatus, Lamium album, Lapsana communis, Lolium perenne, Phalaris arundinacea, Phragmites australis, Poa trivialis, Rumex acetosella, Rumex obtusifolius, Sonchus arvensis, Sorghum alepense, Stellaria media, Taraxacum* spp., *Torilis japonica, Trifolium pratense, Trifolium repens, Urtica dioica.*			

Overall for P1, P2, P3, P4 and P5, 22 wetland species were recorded (Table 5.8). The last and largest pond (P5) hosted the highest richness of macrophytes of all five ponds, with 27 species recorded in 2006 and 2007, of which 18 were considered as wetland plants, and had the highest rarity index (1.12), together with P4 (1.11).

Table 5.8 Plants found and their percentage cover class within the wet perimeter of P1 to P5 (CFW2). Rarity score is given for each pond (06: 2006, 07: 2007).

Plant species	P1		P2		P3		P4		P5		Wetland area
	06	07	06	07	06	07	06	07	06	07	07
Achillea millefolium (L.)	0	0	0	0	0	0	0	0	0	0	< 1
Agrostis stolonifera (L.)	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	1	1	< 5
Alopecurus geniculatus (L.)	0	0	0	0	0	0	0	0	0	0	5
Arctium minus (Hill) Bernh.	0	0	0	0	0	0	0	0	< 1	< 1	0
Butomus umbellatus (L.)	0	0	0	0	0	0	0	0	1	1	0
Callitriche brutia (L.)	0	0	20	20	20	20	30	30	30	30	0
Caltha palustris (L.)	0	0	0	0	0	0	0	0	< 1	< 1	0
Capsella bursa pastoris (L.)	0	0	0	0	0	0	0	0	0	0	< 1
Chamerion angustifolium (L.)	0	0	0	0	0	0	0	0	< 1	< 1	0
Cirsium ravense (L.)	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Cirsium vulgare (Savi) Ten.	0	0	0	0	0	0	< 1	< 1	0	0	< 1
Crataegus monogyna (Jacq.)	0	0	0	0	0	0	0	0	0	0	< 1
Dactylis glomerata (L.)	0	0	0	0	0	0	0	0	0	0	5
Deschampsia cespitosa (L.)	0	0	0	0	0	0	0	0	0	0	< 1
*Epilobium hirsutum (*L.)	< 1	1	1	1	0	0	0	0	< 1	< 1	< 5
Epilobium obscurum (Schreb.)	< 1	< 1	< 1	< 1	0	< 1	< 1	< 1	< 1	< 1	< 5
Glyceria fluitans (L.)	20	30	1	2	< 1	< 1	< 1	< 1	< 1	< 1	5
Holcus lanatus (L.)	0	0	0	0	0	0	0	0	0	0	30
Hyperichum pulchrum (L.)	0	0	0	0	0	0	0	0	< 1	< 1	< 1
Iris pseudacorus (L.)	0	0	0	0	0	0	0	0	< 1	0	0
Juncus effusus (L.)	5	10	5	10	5	10	5	10	5	10	5
Lemna minor (L.)	< 1	< 1	10	20	20	10	20	20	70	50	< 1
Lolium perenne (L.)	0	0	0	0	0	0	0	0	0	0	< 5
Mentha aquatica (L.)	0	0	0	0	0	0	0	0	< 1	< 1	< 1
Myriophyllum spicatum (L.)	0	0	0	0	0	0	0	0	50	30	0
Nasturtium officinale (R. Br.)	20	30	0	0	0	< 1	0	0	5	5	10
Persicaria amphibia (L.) Gray	0	0	0	0	0	0	0	0	< 1	< 1	0
Persicaria maculosa (L.) Gray	0	0	0	0	0	0	< 1	< 1	0	0	0

Plant species	P1		P2		P3		P4		P5		Wetland area
	06	07	06	07	06	07	06	07	06	07	07
Phleum pratense (L.)	0	0	0	0	0	0	0	0	0	0	< 5
Phragmites australis (L.)	20	30	10	20	10	20	10	20	10	20	< 5
Plantago major (L.)	0	0	0	0	0	0	0	0	0	0	< 5
Poa trivialis (L.)	0	0	0	0	0	0	0	0	0	0	10
Potamogeton crispus (L.)	0	0	0	0	0	0	0	0	20	20	0
Prunella vulgaris (L.)	0	0	0	0	0	0	0	0	0	0	< 1
Ranunculus acris (L.)	0	0	0	0	0	0	0	0	< 1	< 1	0
Ranunculus omiophyllus (Ten.)	0	0	0	0	0	0	50	50	0	0	0
Ranunculus repens (L.)	1	1	< 5	< 5	< 5	< 5	< 5	< 5	< 1	< 1	5
Rumex acetosa (L.)	0	0	0	0	0	0	0	0	< 1	< 1	< 1
Rumex obtusifolius (L.)	< 1	< 1	0	0	0	0	< 1	< 1	< 1	< 1	< 1
Salix sp. (L.)	0	0	0	0	0	0	0	0	0	0	< 1
Senecio jacobea (L.)	0	0	0	0	0	0	0	0	0	0	< 1
Senecio vulgaris (L.)	0	0	0	0	0	0	0	0	< 1	< 1	< 1
Sonchus arvensis (L.)	0	0	0	0	0	0	0	0	0	0	< 1
Sparganium erectum (L.)	0	0	0	0	0	0	0	0	< 5	< 5	0
Stachys sylvatica (L.)	< 1	< 1	0	0	0	0	0	0	0	0	< 1
Stellaria media (L.)	0	0	0	0	0	0	0	0	0	0	< 1
Trisetum flavescens (L.) Beauv.	0	0	0	0	0	0	0	0	0	0	< 1
Trifolium repens (L.)	0	0	0	0	0	0	0	0	0	0	5
Urtica dioica (L.)	< 5	0	0	0	0	0	0	0	< 1	< 1	< 1
Veronica beccabunga (L.)	< 1	< 1	< 1	< 1	5	5	5	5	5	5	5
Vicia cracca (L.)	0	0	0	0	0	0	0	0	0	0	< 1
Number of plant species	14	12	11	11	9	11	14	14	28	27	38
Wetland plant species	9		9		9		10		18		11
Submerged and floating species	2		3		3		5		7		2
Rarity score	1.00		1.00		1.00		1.11		1.12		1.00

Changes in species richness between 2006 and 2008 were not significant, illustrating a relative stable composition of plant assemblages. However, a few plants seemed to have disappeared, such as *I. pseudacorus* and *N. alba* (planted initially and recorded in 2005 by Marcello Windsor), outcompeted by *P. australis* or grazed and uprooted by swans. Compared to the amenity pond, species richness and rarity were much lower. Colonization by *P. australis* was fast in and around all the ponds (c. 1 m a year), reducing the area of open water over time. In the wetland area between P3 and P4, *N. officinale* established very well and covered more than 400 m^2 in 2008.

5.3.3 Macroinvertebrate diversity

Table 5.9 presents a list of the aquatic macroinvertebrates found in each pond, with their total abundance (three samples at AP and four samples at the other ponds) or their presence only (indicated by "x") when the specimen was not identified to species level, and the species rarity index (SRI).

Table 5.9 Aquatic macroinvertebrate taxa and abundance in each pond. Corixidae were not systematically identified to species level and diptera were not identified beyond order level (x: indicate the presence of a taxa).

Taxa	Sampling location								SRI
	AP	K	P1	P2	P3	P4	P5	P1- P5	
CRUSTACEA									
Asellus aquaticus (L., 1758)	485	1	0	0	0	0	0	0	1
Daphnia magna (Straus, 1820)	0	10^5	0	0	0	0	0	0	1
Gammarus pulex (L., 1758)	153	0	0	0	0	0	1	x	1
MOLLUSCA									
Planorbis carinatus (M., 1774)	413	0	0	5	0	52	102	x	1
Limnaea peregra (M., 1774)	487	0	127	694	540	480	264	x	1
Potamopyrgus jenkinsi (Sm., 1884)	0	0	35	200	120	95	129	x	1
Pisidium sp. (Pfeiffer, 1821)	33	0	323	12	1	1	0	x	1
Physa acuta (Drap., 1805)	0	53	0	0	0	0	0	0	1

Taxa	Sampling location								SRI
	AP	K	P1	P2	P3	P4	P5	P1- P5	
INSECTA									
COLEOPTERA									
Agabus bipustulatus (L., 1767)	0	1	18	12	9	1	4	x	1
Agabus guttatus (Paykull, 1798)	0	0	1	0	0	0	0	x	1
Agabus nebulosus (Forster, 1771)	0	9	1	0	0	0	4	x	1
Agabus sturmii (Gyllenhal, 1868)	11	0	0	6	0	0	2	x	1
Anacaena lutescens (Steph., 1829)	7	1	0	0	0	0	0	0	1
Colymbetes fuscus (L., 1758)	1	1	0	3	1	0	5	x	1
Dytiscus marginalis (L., 1758)	0	0	1	0	3	0	1	x	1
Gyrinus substriatus (Steph., 1828)	3	3	0	0	1	0	0	x	1
Haliplus confinis (Steph., 1828)	43	0	2	31	20	10	63	x	1
Helophorus aequalis (Th., 1808)	0	0	10	0	1	0	0	x	1
Helophorus brevipalpis (B., 1881)	3	2	6	4	3	3	3	x	1
Helophorus grandis (Illiger, 1798)	0	2	1	2	1	0	0	x	1
Hydrobius fuscipes (L., 1758)	1	0	7	0	4	1	4	x	1
Hydroporus gyllenhalii (Sc., 1841)	4	0	0	0	0	0	0	0	1
Hydroporus memnonius (N., 1822)	0	0	14	0	0	0	1	x	1
Hydroporus palustris (L., 1761)	4	3	40	9	12	11	7	x	1
Hydroporus tesselatus (D., 1819)	0	1	0	0	0	0	0	0	1
Hygrotus inaequalis (Fab., 1776)	157	0	0	5	3	2	63	x	1
Ilybius fuliginosus (Fab., 1792)	1	3	27	0	6	2	3	x	1
Ilybius ater (De Geer, 1774)	1	0	0	0	0	0	0	0	1
Laccobius biguttatus (Gerh., 1877)	21	0	0	0	0	0	0	0	1
Laccobius minutus (L., 1758)	0	0	161	0	4	0	0	x	1
Laccobius striatulus (Fab., 1801)	0	0	0	1	0	0	0	x	1
Laccophilus minutus (L., 1758)	0	0	0	1	0	0	3	0	1
Potamonectes depressus elegans (Panzer, 1794)	1	0	0	0	0	0	0	0	1

Taxa	Sampling location								SRI
	AP	K	P1	P2	P3	P4	P5	P1- P5	
DIPTERA									
Chaoboridae	1	x	x	42	37	151	170	x	1
Chironomidae	44	1536	240	633	555	412	288	x	1
Cyclorrhapha	7	0	15	7	5	11	11	x	1
Dixidae	8	x	x	x	x	x	x	x	1
Simulidae	0	0	1	0	0	0	0	x	
Tipulidae									
*Tipula rufina (*Meigen, 1818)	5	0	0	16	5	4	5	x	1
EPHEMEROPTERA									
Cloeon dipterum (L., 1761)	141	4	2	668	387	558	289	x	1
HEMIPTERA									
Corixidae	249	474	36	297	234	177	172	x	
Corixa punctata (Illiger, 1807)	x	x	x	x	x	x	x	x	1
Callicorixa praeusta (Fieber, 1848)	0	x	0	0	0	0	0	0	1
Gerris lacustris (L., 1758)	20	2	1	0	0	1	1	x	1
Hesperocorixa linnaei (Fieber, 1848)	x	0	0	0	0	0	0	0	1
Hydrometra stagnorum (L., 1758)	4	0	0	1	0	0	5	x	1
Notonecta glauca (L., 1758)	15	14	0	20	89	18	77	x	1
Sigara spp.	x	x	x	x	x	x	x	x	
Sigara concinna (Fieber, 1848)	0	x	0	0	0	0	x	x	1
Sigara distincta (Fieber, 1848)	0	x	0	0	0	0	0	0	1
Sigara dorsalis (Leach, 1817)	x	0	0	0	0	0	x	x	1
Sigara fossarum (Leach, 1817)	0	0	0	0	0	0	x	x	1
Sigara lateralis (Leach, 1817)	0	x	0	0	0	0	0	0	1
Sigara semistriata (Fieber, 1848)	x	0	0	0	0	0	0	0	1
MEGALOPTERA									
Sialis sp. *(Latreille, 1802)*	0	0	0	13	0	2	2	x	1
ODONATA									
Ischnura elegans (Van der L., 1820)	144	0	0	22	26	31	26	x	1
Sympetrum striolatum (Char., 1840)	0	0	1	11	9	8	36	x	1

Taxa	Sampling location								SRI
	AP	K	P1	P2	P3	P4	P5	P1- P5	
PLECOPTERA									
Nemoura cinerea (Retzius, 1783)	0	0	0	9	2	0	0	x	1
TRICHOPTERA									
Anabolia nervosa (Curtis, 1834)	8	0	0	0	0	0	0	0	1
Limnephilus extricatus (Mac Lachlan, 1865)	0	0	4	4	3	2	6	x	1
Limnephilus lunatus (Curtis, 1834)	0	0	14	103	17	7	11	x	1
Oecetis sp. (Mac Lachlan, 1877)	7	0	0	0	0	0	0	0	1
Phryganea bipunctata (Retzius, 1783)	8	0	0	0	0	0	0	0	1
ACARINA									
Hydracarina	40	2	1	16	40	15	96	x	
OLIGOCHAETA	0	201	240	273	347	62	215	x	1
HIRUDINEA									
Erpobdella octoculata (L., 1758)	4	52	0	33	7	61	52	x	1
Helobdella stagnalis (L., 1758)	0	0	100	5	30	27	0	x	1
Theromyzon tessulatum (Müller, 1774)	81	0	0	17	54	55	181	x	1
Glossiphonidae spp.	0	0	0	0	0	11	0	x	1
TRICLADIDA	4	1	101	0	0	0	0	x	
Number of BMWP scoring families	23	13	16	21	19	20	21	27	
Number of BMWP scoring species	36	24	> 28	> 32	> 32	> 28	> 33	> 46	
Total BMWP	113	54	69	92	85	85	93	150	
ASPT	4.78	4.15	4.31	4.38	4.47	4.25	4.42	5.55	
Mean abundance per sample	870	592	383	794	644	568	576	-	
Rarity Score	1	1	1	1	1	1	1	1	
Shannon Index (H’)	2.52	1.44	2.36	2.18	2.27	2.15	2.64	-	
Evenness	0.71	0.45	0.71	0.63	0.66	0.65	0.75	-	

The highest family and species richness were found at AP, which hosted more than 23 BMWP scoring families and 36 scoring species, while the lowest richness was at K, with only 13 scoring families and 24 scoring species. On all sampling occasions at K many dead hemipterans were found in the first segment of the pond, indicating a high mortality rate probably due to high pollutant concentrations and low oxygen levels. Macroinvertebrate richness at CFW2 was only slightly lower than at AP, ranging between 16 (P1) and 21 (P2 and P5) scoring families and 28 (P1) and 33 (P5) scoring species, indicating a favourable environment for aquatic life.

At AP, the most abundant species were the snail *Limnaea peregra* (19%), the water louse *Asellus aquaticus* (19%), the snail *Planorbis carinatus* (16%) and species from the family Corixidae (10%), followed by the Coleoptera *Hygrotus inaequalis* (6%), the freshwater shrimp *Gammarus pulex* (6%) and the mayfly *Cloeon dipterum* (5%). The families Leptoceridae and Phryganeidae present in AP are absent from CFW1 and CFW2, and only one specimen of Asellidae was found at K and one specimen of Gammaridae in P5, indicating less suitable environments for those families.

At K, the most abundant species belonged to Chironomidae (65% of the overall abundance), Corixidae (20%) and Oligochaeta (8%), which are usually pioneer and less pollution-sensitive taxa. At CFW2, the most abundant species belonged to Chironomidae (12%-22%), Limnaeidae (8%-21%), Hygrobiidae (2%-6%), Oligochaeta (3%-15%) and Corixidae (2%-9%). Erpobdellidae and Glossiphoniidae (leeches) were relatively abundant at CFW2 compared to the other ponds.

In terms of diversity, the highest Shannon Index (*H'*) and Evenness (*E*) were for P5 (*H'*= 2.75, *E*= 0.76), AP (*H'*= 2.52; *E*= 0.71) and P1 (*H'*= 2.36, *E*= 0.71), while the lowest were for K (*H'*= 1.44, *E*= 0.45). Differences between the BMWP score of the different ponds were observed (Figure 5.1). AP had the highest score, partly due to the presence of pollution-sensitive species such as *Phryganeidae bipunctata* and *Oecetis* sp. (both scoring 10) and *Anabolia nervosa* (scoring 7), while K had the lowest score, with a maximum score of 5 for any of the families. Interestingly, the overall BMWP score for CFW2 was higher than all individual ponds and than AP's.

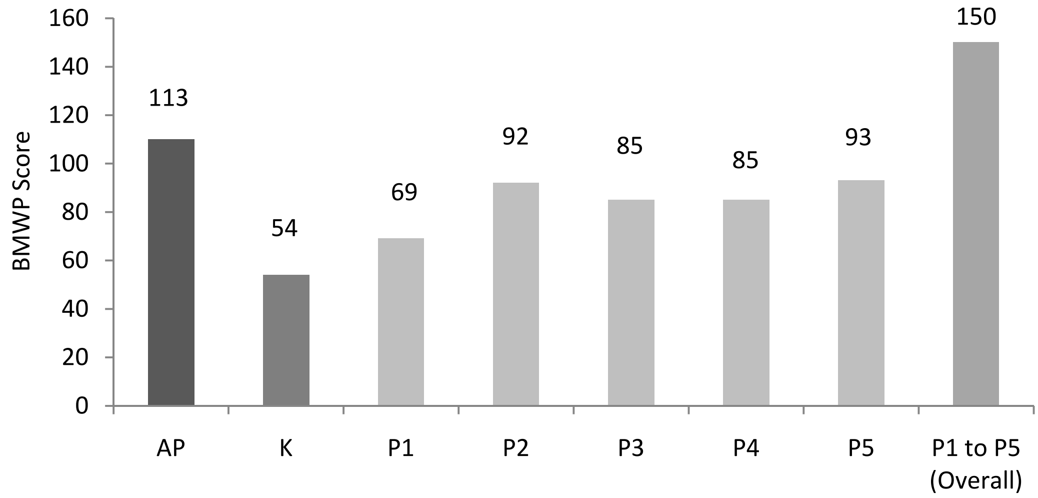


Figure 5.1 Total BMWP scores of the ponds studied.

The cluster analysis based on the Jaccard Index (Figure 5.2) showed differences in species composition between the ponds, and separated them into five groups. K had only 37% similarity with the other ponds. The amenity pond had 50% similarity with P1 to P5, which formed the last three groups. P1 shared 63% of species with the other ponds, P5 shared 67% of the species and P2, P3 and P4 were grouped together, sharing 72% of the species.

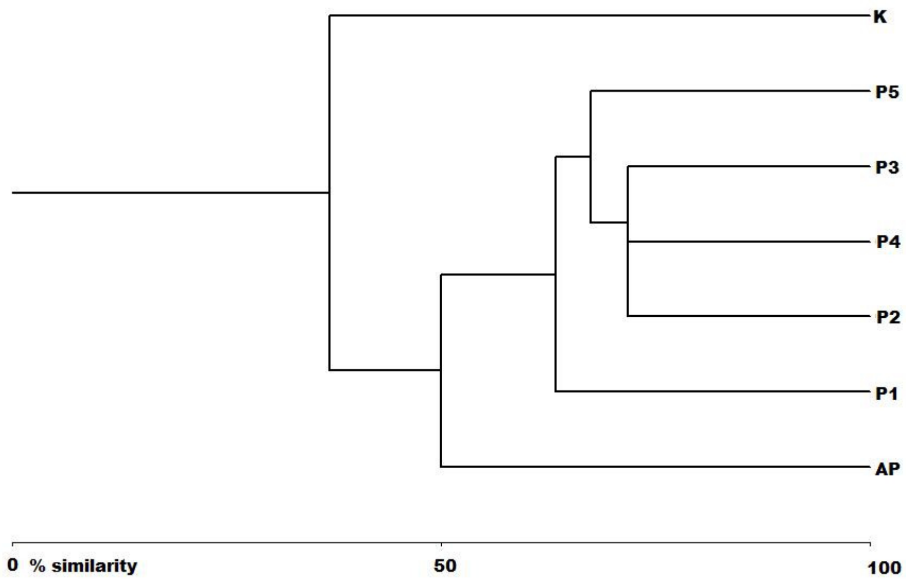


Figure 5.2 Dendrogram illustrating similarities in aquatic macroinvertebrate assemblages between the seven ponds studied, based on the Jaccard coefficient of similarity and single linkage method.

5.3.4 Other wildlife

The most diverse wildlife was observed at CFW2 (Table 5.10), with a large variety of mammals, birds, insects and amphibians using the wetland for feeding, breeding or resting, and benefiting from a better shelter, larger area and more diverse vegetation and floristic composition. Nevertheless, waterfowl appeared to be threatened by dogs or wild animals, which were responsible for the death of all three young swans in 2007. At CFW1, the shelter was less, and therefore, exposure to wind and human disturbance strongly limited its use by wildlife.

Table 5.10 Animals observed at CFW1 and CFW2 between 2006 and 2008.

	CFW1	CFW2
Amphibians	*Bufo bufo* (Toad) *Rana temporaria* (Common frog)	*Bufo bufo* *Rana temporaria* *Triturus vulgaris (*Smooth newt)
Wetland birds/Waders	*Anas platyrhynchos* (> 5 pairs) *Ardea cinerea* *Aythya fuligula* (1 pair) *Carduelis carduelis* *Columba palumbus* *Cygnus olor* (1 pair) *Delichon urbica* *Gallinago gallinago* *Gallinula chloropus* (1 adult) *Haematopus ostralegus* *Hirundo rustica* *Motacilla alba* *Tadorna tadorna* (1 pair)	*Anas platyrhynchos* (> 8 pairs) *Ardea cinerea* *Aythya fuligula* (1 pair) *Carduelis carduelis* *Columba palumbus* *Cygnus olor* (1 pair+ cheeks) *Delichon urbica* *Fulica atra* (1 pair + chicks) *Gallinago gallinago* *Gallinula chloropus* (1 adult + chicks) *Hirundo rustica* *Perdix perdix* *Phasianus colchicus*
Lepidoptera	*Inachis io* *Pieris pieris*	*Aglais urticae* *Anthocaris cardamines* *Aphantopus hyperantus* *Coenonympha pamphilus* *Inachis io* *Pieris pieris* *Vanessa atalanta*
Odonata	*Ischnura elegans* *Sympetrum striolatum*	*Coenagrion puella* *Ischnura elegans* *Sympetrum striolatum*
Mammals	*Mustela erminea* Small rodents (prints and faeces)	*Mustela erminea* Small rodents (prints and faeces)

5.4 Discussion

5.4.1 Habitat quality: influence of design and water quality

The conservation values of the ponds, based on wetland plant and aquatic macroinvertebrate richness and rarity indices, are given in Table 5.11.

Table 5.11 Richness, SRI, BMWP, ASPT and Conservation Value (H: High, M: Moderate) of the Ponds.

Pond	Wetland vegetation			Aquatic invertebrates (Families and species with a BMWP score)					
	Species richness	SRI	CV	Family richness	Species richness	SRI	CV	BMWP	ASPT
AP	29	1.18	H	23	36	1	H	110	4.91
K	15	1.07	M	13	24	1	M	54	4.15
P1	9	1	M	16	28	1	M	69	4.31
P2	9	1	M	21	32	1	M	92	4.38
P3	9	1	M	19	32	1	M	85	4.47
P4	10	1.11	M	20	28	1	M	85	4.25
P5	18	1.12	M	21	33	1	H	93	4.42
CFW2 overall	22	1.12	M	27	46	1	H	150	5.55

In terms of wetland vegetation, AP had a high CV and a higher wetland plant richness than found in the National Pond Survey (Table 5.12), while the other ponds had a moderate value. The high value and relatively stable composition of AP plant community may be explained by several human and natural factors. The older age of the pond, the initial introduction of species collected in natural wetlands, the maintenance work (e.g. control of *T. latifolia*, *P. australis*) which reduced competition by the more invasive species, and the relatively good quality of the water and structural complexity of the habitat (e.g. boulders, variable depth, shaded and open areas) allowed plants and invertebrates with different requirements to thrive (Gee *et al.*, 1997; SEPA and Pond Action, 2000; Alsfeld *et al.*, 2008). The presence of riparian trees and subsequent shade increasing micro-habitat diversity may also have contributed to the higher plant diversity (Biggs *et al.*, 1994).

Table 5.12 Wetland plant species richness recorded from other UK ponds (Pond Action, 2002; Lancaster *et al.*, 2004; Coletto, 2008; Culhane, 2007) compared with the study ponds.

Survey		No. of species of marginal plants	No. of species of aquatic plants[1]	Total No. of plant species
National Pond Survey (high quality ponds in semi-natural areas)	Mean	18	5	23
	Range	1-42	0-14	1-46
Wider countryside ponds (DETR Lowland Pond Survey)	Mean	8	2	10
	Range	0-30	0-10	0-35
Wider countryside ponds (ROPA)	Mean	11	3	14
SUDS (Halbeath, Linburn) (Lancaster *et al.*, 2004; Culhane, 2007)	Mean	2.2	1.2	3.5
	Range	1-4	1-2	1-6
[2]Pond Action (2002) – 13 SUDS ponds (Scotland)	Mean	-	-	13
	Range	-	-	2-25
[2]Behrendt (2004) (25 SUDS ponds, Scotland)	Mean	-	-	11
	Range	-	-	3-20
[2]Pond Action (8 SUDS Ponds, Hopwood, England)	Mean	-	-	9.6
	Range	-	-	6-13
AP	Number	21	9	29
K	Number	14	0	15
P1	Number	8	2	9
P2	Number	7	3	9
P3	Number	7	3	9
P4	Number	7	5	10
P5	Number	13	7	18

- Data not available.
[1]Aquatic plants are both submerged and floating-leaved species.
[2]Only naturally self-colonising species were recorded.

Within K, the limited number of plant species, their very small abundance (except for *T. latifolia*), and the absence of submerged and floating plants indicated very slow colonization of the pond and rather unsuccessful establishment, in contrast to ponds not used for water treatment, where colonization can be very fast (Gee *et al.*, 1997). This may be explained by the young age of the pond, a small soil seed stock (since the land was previously improved pasture), high flow velocity close to the inlet, very limited growth substrate (a clay base was used, without topsoil) and the very poor water quality. Grazing by swans may also have impeded colonisation. The only species colonizing at a fast rate was *T. latifolia*, which outcompeted smaller species growing on the edges.

At CFW2, vegetation was much more diverse, maybe due to a pre-existing soil seed stock, the proximity of natural wetlands and rivers providing seeds and propagules transported by waterfowl, humans and wind (Fenner, 1985; Clausen *et al*., 2002; Soons *et al*., 2008), a relatively good water quality and a permanent groundwater input. The fastest spreading species was *P. australis*, which is expected to colonize the entire area of the ponds within a few years.

In both CFWs the denser plant cover (whether by *T. latifolia* or *P. australis*) expected within the next few years may improve water treatment, enhancing filtration, uptake and providing substrate for bacteria to nitrify and denitrify. However, due to the configuration of the ponds (shallow edges, deeper centre) and planting patterns (only on the edges), colonization could also result in a decrease in the retention volume and time and flow channelization (water will not be forced through vegetation and will find the path of least resistance), subsequently leading to a decrease in treatment efficiency. Colonization by *T. latifolia* may also outcompete other species and could result in a habitat less diverse and therefore less resilient to climate change, water quality fluctuations and pests or invasive species.

5.4.2 Macroinvertebrate diversity: influence of design and water quality

When considering only the aquatic plants present in significant numbers (more than three individuals), there were weak but significant positive correlations in the study ponds between macroinvertebrate richness and both the number of plant species (r_s= 0.80, p= 0.05) and the number of submerged and floating plant species (r_s= 0.85, p= 0.04). This positive correlation has been shown in studies of other ponds (Gee *et al*., 1997; Nicolet *et al*., 2003) and confirms the importance of maintaining a diverse flora in the wetlands, to provide animals with food, shelter and breeding sites.

Table 5.13 compares the ecological value in terms of macroinvertebrates of the ponds studied and other UK ponds, including SUDS and less contaminated non SUDS ponds. Many species found within SUDS were also found at CFW1 and CFW2 suggesting relatively similar assemblages between these systems (Culhane, 2007; Jackson and Boutle, 2008).

Table 5.13 Aquatic macroinvertebrate species richness recorded from other UK ponds and compared with the study ponds.

Survey		Number of aquatic macroinvertebrate species
National Pond Survey (high quality ponds in semi-natural areas)	Mean	32*
	Range	6-98*
Wider countryside ponds (ROPA Survey)	Mean	26*
	Range	2-64*
SUDS (Halbeath, Linburn) (Lancaster *et al*., 2004; Culhane, 2007)	Mean	27
	Range	20-37
Pond Action (2002) - Ponds in Scotland	Mean	39.8
	Range	24-58
Hopwood (Pond Action unpublished)	Mean	36.9
	Range	22-58
AP	Number	36**
K	Number	24***
P1	Number	28***
P2	Number	32***
P3	Number	32***
P4	Number	28***
P5	Number	33***
P1 to P5 (Overall)	Number	46***

*Results are from a single 3-minute hand-net sample.
** Scoring species from three 3-minute samples.
*** Scoring species from four 3-minute samples.

The amenity pond had the highest invertebrate species richness and diversity of the ponds studied, above the average for the ponds monitored during the National Pond Survey and in SUDS, and in the range of the lowly impacted Scottish ponds monitored by Pond Action (2000), and was given a high conservation value. The relatively high aquatic invertebrate diversity in AP can be explained by several factors. The primary factor could be the greater age of the pond (more time for colonization), and the quality of the water (better than in CFWs) which allowed more sensitive species to colonize and survive. The higher structural diversity of the pond was also an important component, as it has been shown to enhance habitat, the diversity of ecological niches and therefore biodiversity (Alsfeld *et al*., 2008).

The large size of AP might only partly explain the high ecological value since studies investigating the influence of pond size on biodiversity have shown mixed results, depending on the taxa considered. Odonata are favoured by larger ponds, while the relationship between size and diversity is weak for other taxa such as Coleoptera, and smaller ponds have often been shown to be equally or more diverse than larger ones (Gee *et al*., 1997; Oertli *et al*., 2001; Ruggiero *et al*., 2008). The apparent absence of dragonflies may suggest a negative impact of the fish, which commonly feed on larger-bodied forms of Odonata (Crowder and Cooper, 1982).

In contrast, the low diversity and abundance characterising K were largely explained by the poor water quality and large fluctuations in pollutants (e.g. NH_3, which may dominate at high pH, is particularly toxic to aquatic wildlife) and oxygen levels (deduced from high BOD_5 concentrations), young age, low structural diversity, and limited vegetation cover. The taxa most affected by water pollution, e.g. insects breathing under water, such as dragonflies, damselflies, mayflies and caddisflies, were absent from CFW1, while air breathing invertebrates such as water beetles, snails and bugs, which are less affected by poor water quality (Williams *et al.*, 2003), seem to have colonized quickly, although they still had a low abundance in 2008.

At CFW2, no significant differences were found between ponds in terms of aquatic invertebrate richness and diversity, probably due to relatively good water quality all across the wetland. Nevertheless, the continued accumulation of sediment in P1 and subsequent drastic reduction in pond's volume, might have hampered seed germination and egg hatching and subsequently affected invertebrate colonization, development and survival (Gleason *et al*., 2003). This could explain the lower abundance and richness found in 2007 (14 and 15 species) compared to 2006 (18 and 14 species), although this could also be due to differences in the timing of surveys (July and September 2006 and May and August 2007). The presence of stoneflies in P2 and P3 was also notable, resulting from the constant flow of relatively clean and oxygenated water (Hynes, 1977).

The relatively high concentration of NO_3 (especially closer to the inlet) did not appear to be a critical factor in determining invertebrate survival and diversity. However, it has been shown that nitrate toxicity can be a threat to invertebrates and amphibians for concentrations above 45 mg l^{-1} and long exposure times, and that a maximum of 9 mg l^{-1} would be appropriate for protecting the most sensitive freshwater species (Camargo *et al*., 2005).

Although no significant influence of pond size, vegetation cover or water quality was observed in this study, surveys of large wetlands treating highly contaminated farmyard runoff in Ireland (Harrington *et al*., 2005) found a higher macroinvertebrate diversity in the wetland cells further away from the inlet, correlated with a significant improvement in water quality. Indeed, most of the invertebrates found close to the inlet were Diptera, while further away, cells were also colonised by Hemiptera (dominant), Crustacea, Molluscs, Ephemeroptera, Coleoptera, Odonata and Hirudinea. This illustrates the potential for increased colonization of CFWs if water quality is to be improved, e.g. by the use of larger, multi-cell configurations.

Different scenarios are possible regarding future changes in macroinvertebrate diversity within the CFWs. As suggested by other studies, species richness could continue to increase during the next few years (Usher & Jefferson, 1989) with the ponds maturing and colonization continuing. Jackson and Boutle (2008) found for example an increase (between 1.2 and 4.5 times) in aquatic invertebrate family numbers between 2006 and 2007 at all SUDS sites studied in Upton, Northampton, UK. Usually, species with high dispersal ability (e.g. winged insects such as Coleoptera and Hemiptera) colonize in the early years while less mobile colonists arrive later (e.g. crustaceans, snails or leeches) (Lancaster *et al.*, 2004).

However, the fast colonization by *P. australis* and increase in plant biomass could cause the disappearance of the other plant species and simplification of the habitat within a few years, which could subsequently lead to a decrease in invertebrate diversity. Additionally, sediment accumulation could also affect plant and invertebrate diversity as discussed earlier.

5.4.3 Wildlife and water quality

While it is known that good water quality benefits wildlife, in this study, a negative impact of wildlife on water quality was found. In summer 2007 higher concentrations of Faecal Coliforms (> 150 000 cfu 100 ml^{-1}) and Faecal *Streptococci* (> 700 cfu 100 ml^{-1}) were measured in the outflow, compared to the inflow (3500 cfu 100 ml^{-1} and 430 cfu 100 ml^{-1}, respectively) corresponding to the presence of two adult swans and three cygnets, and occasionally of ducks and moorhens. This issue has been mentioned previously by several authors and has to be taken into account when the primary objective of a CW is water treatment (Jones and Obiri-Danso, 1999). It could be addressed by increasing vegetation cover and limiting open water surface area close to the outlet.

5.5 Conclusions

This study suggests that CFWs can sustain relatively rich wildlife communities of wetland plants, macroinvertebrates, amphibians and birds at a landscape level, although they generally host fewer and mainly common species compared to natural wetlands or ponds not used for treatment (see National Pond Survey, 2002). However, their ecological value depends on water quality and habitat heterogeneity and therefore on their design, use and management, which confirms hypothesis 2 presented in section 1.6.2. For example, small, single-celled, heavily-polluted CFWs are expected to be less efficient and less ecologically diverse than larger multi-cell systems, in which water quality and subsequently habitat quality will be improved. Also the positive relationship between plant species and macroinvertebrate richness indicates that vegetation establishment is important for development of biodiversity.

The factors influencing habitat quality and animal diversity within wetlands are numerous (Oertli *et al.*, 2002; Nicolet *et al.*, 2004; Batty *et al.*, 2005; Williams *et al.*, 2007; Alsfeld *et al.*, 2008) but the main aspects to focus on when trying to enhance the biodiversity conservation potential of CWs have been summarised by various authors (Williams *et al.*, 1999; SEPA and Pond Action, 2000; Carty *et al.*, 2008a). They address issues of pond construction, i.e. location, size, depth, shape and structure, and maintenance, including vegetation and sediment removal. Since water quality is a crucial parameter influencing wildlife survival, particular emphasis is put on reducing contamination in the first place, e.g. by controlling pollution sources (e.g. scraping the farmyard, ensuring proper functioning of slurry tank valves) or using intermediate treatment options such as buffer strips or swales. From this study and past work, the following recommendations are proposed for the establishment and maintenance of CFWs to maximise biodiversity and also achieve water quality objectives, taking into account socio-economic constraints.

Design and construction:

1) Use a series of linked vegetated ponds with shallow edges to create a water quality gradient, improving progressively away from the inlet. A multi-cell system also restricts to a certain extent the risk of toxicity to wildlife to the initial cells, for example in case of accidental spillages.
2) Sediment capture at the beginning of the constructed wetland should be promoted, using a sediment trap which can be regularly easily emptied.
3) Design ponds with curved edges for a good landscape fit, improved aesthetics and enhanced habitat structure.
4) Ponds should be planted at an early stage (ideally a few months before wastewater application), especially in areas of higher flow and higher pollutant concentrations, i.e. close to the inlet, using a combination of several species coming from approved nurseries or local habitats (if permitted).
5) The presence of a large area of open water in the last wetland cell should be avoided to limit faecal contamination by waterfowl.
6) Whenever possible, relatively clean water (e.g. roof water) should be separated from yard runoff and diverted into ditches or adjacent ponds, to increase residence time and treatment of potentially contaminated water in the CFW and to create more favourable habitats around it. However, a small permanent input of ground water diluting the inflow may be beneficial to wildlife.
7) Structural heterogeneity can be improved by introducing small quantities of materials such as dead wood and gravel (Alsfeld *et al.*, 2008).

CFW management activities:

1) The removal of marginal or aquatic vegetation and trees should be avoided, but when necessary, such activities should be carried out over a limited area at any one time. Management should aim at maintaining variations in plant density and diversity, and adequate timing should minimize animal disturbance.
2) Sediment removal should be limited in space and time, but the recommendation for amenity ponds (< 1 m^3 100 m^{-2}) would be impractical in CFWs. Dredging every 10 years and leaving a small amount of sediment in each cell/pond could help the fauna to re-establish quicker after disturbance.
3) Generally, access by cattle or sheep which could damage banks and contaminate the water should be restricted, and the frequency and extent of mowing of the edges should be limited to the minimum needed for access to the ponds. This allows a more diverse flora to establish and flower, providing nesting, feeding and breeding sites for a variety of animals.

Biodiversity conservation objectives seem compatible with the main water treatment goal of CFWs, if those systems are built large enough and if large waterfowl is kept off the last pond. This could therefore further justify the financial support given to farmers for the construction and maintenance of large, multi-cell and inevitably more expensive systems. However, monitoring of CFWs and further experimentation are needed to assess the long-term impacts of pollutants on wildlife and to help improve CFW design to meet more holistic and multi-objective approaches. Ecological surveys clarifying the link between design, management and ecological value should ultimately help achieve the necessary compromise between water treatment and biodiversity conservation.

Combining multiple objectives, i.e. water treatment, biodiversity conservation and amenity enhancement, within CFWs might be the way forward to increase their cost-effectiveness, broaden their acceptability amongst the farming community and society and persuade policy makers to support them adequately.

Chapter 6: Costs, Benefits and Farmers' Perception of Constructed Farm Wetlands

The chapter discusses the socio-economic aspects associated with the implementation of Constructed Farm Wetlands (CFWs, Scotland), Integrated Constructed Wetlands (ICWs, Ireland) and other systems used in France (e.g. lagoons, vertical wetlands). Results are based on literature review and interviews with farmers and experts (e.g. wetland designers, farm advisers). The chapter details the costs of CFWs and their benefits, assesses the way they are perceived by farmers and the obstacles hindering their implementation. A comparison of the costs with other alternatives for dirty water management in the UK and France is presented, and suggestions are made to encourage the construction of CFWs.

6.1 Introduction

The impacts of farmyard runoff, when it is left to drain freely to waterbodies, have been long overlooked, but evidence suggests that it highly contributes to water pollution (Cumby *et al.*, 1999; Neumann *et al.*, 2000; Edwards *et al.*, 2008), generating market costs (e.g. loss of profit from fisheries, tourism, costs for water treatment, health care) as well as non-market costs, which are often not properly valued and include the loss of biodiversity and amenity, and emissions of greenhouse gases (D'Arcy *et al.*, 2000; Pretty *et al.*, 2003).

To address this issue and comply with the WFD, CFWs, which are considered a low-cost, low-energy, relatively efficient and ecologically valuable option (Mitsch and Gosselink, 2000; Carty *et al.*, 2008a), are promoted to treat farmyard runoff (EA and SEPA, 2009). Until recently in Scotland, no clear design guidance existed and only sporadic financial support was provided for their implementation, which led to the construction of small, simplified, underperforming systems. However, in 2008, a design manual was produced for Scotland and Northern Ireland (Carty *et al.*, 2008b) and financial support is now available to build CFWs, as stated in the Scotland Rural Development Plan 2007-2013 (Scottish Government, 2008a).

Construction costs are site-specific, influenced by wetland size and design, availability of labour and local materials and use of artificial liners (Dunne *et al.*, 2005), and commonly range from £3000 to more than £50 000. However, land and maintenance cost (e.g. for sediment removal and disposal) are often not accounted for, although they are known to represent a significant part of the investment in SUDS (McKissock *et al.*, 2003; Heal *et al.*, 2006b). Hence, further investigation is needed to detail the costs associated with land use, construction and maintenance of CFWs and relate them to treatment efficiency, and to assess the way CFWs are perceived by farmers, identify the obstacles which hinder their implementation and the factors which may lead to their misuse, in order to propose adequate incentives for their adoption and sustainable use. A comparison with other dirty water management alternatives is useful if those systems or others are to be further promoted. Indeed, experience has shown that designing BMPs without accounting for their cost, effectiveness, acceptability and practicality can lead to measures which are cheap but ineffective, efficient but very costly, or efficient and affordable but are not adopted by farmers (Deffontaines *et al.*, 1994; OECD, 2003; Turpin *et al.*, 2005).

The aims of this chapter are to: 1) identify and quantify wherever possible the costs associated with planning, construction and maintenance of CFWs; 2) identify the benefits obtained from CFWs (without quantifying them in monetary terms); 3) compare the costs and benefits of CFWs with other alternatives for dirty farmyard runoff management in the UK and worldwide; 4) assess the way CFWs are perceived by farmers, the major obstacles hindering their implementation and the factors leading to their misuse; and 5) make suggestions with regard to financial support, communication and dissemination of CFWs.

The main hypothesis driving the study is as follows: CFWs, to be effective, need to be large, which involves a significant investment for farmers. Their adoption is therefore conditioned by land availability and suitability and availability of financial and technical support for construction and maintenance.

6.2 Materials and Methods

6.2.1 Farms and constructed wetlands investigated

An assessment of the costs and benefits of CFWs and an assessment of farmer acceptance of this BMP were undertaken based on a sample of 23 farms (15 with an existing CFW designed by Soil and Water Scotland or SAC, and 8 without a CFW) located across Scotland, for which information was publicly available. A simple farm typology was used, based on the dominant type of production: dairy, sheep, pig, arable and mixed (i.e. two or more animal productions or combination of crops and animals).

Nine of these farms (including the farms with CFW1 and CFW2) were personally investigated during the present study (Farms 1 to 9), and at eight of these, CFWs had been built since 2003 instigated by SEPA, SNH or the farmer. The data used for the fourteen other farms (Farms 10 to 23) were collected by Kevin Stewart during a two-month undergraduate Honours Project in 2008, which involved conducting short interviews with farmers and water quality monitoring of CFWs (Stewart, 2008). These 14 additional farms are all located in the River Tweed Catchment and at seven of them CFWs were built between March and July 2007, initiated by and with the financial support of the Tweed Forum and Borders FWAG (Farming and Wildlife Advisory Group).

Table 6.1 summarizes the main characteristics of the farms with constructed wetlands. At Farm 9, the wetland was planned and designed, but could not be built due to financial constraints. The design of all CFWs was based on the Treatment Volume approach, as detailed in Chapter 3, and included elements such as swales (linear vegetated channels), sedimentation or infiltration ponds (open water) or terraces, and shallow vegetated wetland areas.

Table 6.1 Main characteristics of the 15 farms with constructed farm wetlands (Source for Farms 10 to 16: Stewart, 2008).

Farm No. and type	Farm size and livestock[a]	Interception area[b] (ha)	CFW construction date, land use previous to CFW[c], design and dimensions	R[d]	Influent[e]
1 Dairy	320 ha 450 DC	T: 9.0 S: 2.30 F: 6.7 I: 32%	Built in 2005, IG; Planted 2006 (*T. latifolia, I. pseudacorus* from local pond); Swale (60 m^2) + 1 pond (2200 m^2/1500 m^3); Area: 2260 m^2	0.02	Y, F, R, S, G
2 Mixed	550 ha 130 SC 45 S	T: 33.8 S: 1.8 F: 32 I: 5%	Built in 2004, UG; Planted (*P. australis*, *I. pseudacorus* from nursery); 5 ponds (50, 115, 105, 190, 2500 m^2) + grassy areas; Area: 6000 m^2	0.02	Y, F, R, S, G
3 Mixed	450 ha 180 SC 525 S	T: 0.39 S: 0.35 I: 90%	Built in 2006, IG; Not planted; 1 pond (890 m^2); Area: 900 m^2	0.23	Y
4 Pig	162 ha 4000 P	T: 0.85 S: 0.56 I: 66%	Built in 2003, IG; Not planted; Swale (100 m^2) + 2 ponds (750 m^2, 260 m^2); Area: 1110 m^2	0.13	Y, R
5 Mixed	160 ha 180 DC 60 SC 20 S	T: 0.60 S: 0.60 I: 100%	Built in 2004, IG; Planted (*P. australis*, *T. latifolia* from nursery); Swale (200 m^2) + 1 pond (670 m^2) + 1 swale (50 m^2) + wetland (850 m^2/425 m^3); Area: 1770 m^2	0.29	Y, R
6 Dairy	60 ha 120 DC	T: 0.26 S: 0.26 I: 100%	Built in 2005, UG; Planted (*P. australis, T. latifolia* from nursery/pond); 1 pond (100 m^2) + swale (50 m^2) + 4 ponds (300, 360, 225, 200 m^2); Area: 1235 m^2	0.45	Y, R, S
7 Mixed	142 ha 300 S 50 SC	T: 0.28 S: 0.28 I: 100%	Built in 2006, UG; Planted (*P. australis*); 1 pond (376 m^2, 169 m^3) + wetland (564 m^2, 157 m^3) + 1 pond (231 m^2, 73 m^3); Area: 1170 m^2	0.42	Y, R
8 Mixed	250 SC 100 S	T: 0.56 S: 0.56 I: 100%	Built in 2004, UG; Planted (*P. australis, I. pseudacorus, T. latifolia);* 1 pond + infiltration + wetland; Area: 10 000 m^2	1.8	Y, R, M, S
9 Mixed	1007 ha 2500 S 200 SC 50 P	NA I: 100%	Not built, UG; Area: 5000 m^2	NA	Y, R, G
10 Mixed	160 SC	T: 0.74 S: 0.34 I: 46%	Built in 2007, A; Not planted; 1 infiltration pond; Area: 470 m^2	0.06	Y, R
11 Mixed	40 SC	T: 0.31 S: 0.31 I: 100%	Built in 2007, IG; Planted (*P. australis*); 4 narrow terraces; Area: 530 m^2	0.03	Y

Farm No. and type	Farm size and livestock[a]	Interception area[b] (ha)	CFW construction date, land use previous to CFW[c], design and dimensions	R[d]	Influent[e]
12 Arable	0	T: 0.78 S:0.39 I: 50%	Built in 2007, IG; Not planted; 1 deep sedimentation pond + shallow marsh; Area: 1024 m^2	0.13	Y, R
13 Dairy	140 DC 150 S	T: 5.96 S: 1.43 I: 24%	Built in 2007, A; Edge planted (*P. australis*); 1 sediment pond + 1 pond; Area: 1150 m^2	0.02	Y, R
14 Mixed	120 SC	T: 5.53 S: 0.55 I: 10%	Built in 2007, A; Not planted; 1 sediment pond + 1 infiltration pond; Area: 950 m^2	0.08	Y, R, F
15 Dairy	200 DC	T: 1.12 S:0.56 I: 50 %	Built in 2007, A; Edge planted (*P. australis*); 1 sediment pond + 1 pond; Area: 440 m^2	0.04	Y, R
16 Dairy	300 DC	T: 0.55 S: 0.55 I: 100%	Built in 2007, IG; Edge + 5 upper bays planted (*P. australis*); 1 sediment pond (20 m^2) + 16 bays (26 m^2); Area: 450 m^2	0.08	Y, R

[a]Livestock: DC: Dairy Cows, SC: Suckler Cows; S: Sheep, P: Pigs. (Farm size unknown in some cases).
[b]T: total area draining into the CFW (excluding overland runoff); S: Steading area; F: Field area, I: Impervious area (% of T).
[c]Land Use previous to CFW. A: Arable, IG: Improved Grassland, UG: Unimproved Grassland.
[d]R: Ratio of CFW surface area to total interception area.
[e]Y: Yard, F: Field drainage; R: Roof, S: Septic tank overflow, G: Groundwater, M: Midden seepage.

6.2.2 Assessment of CFW costs, benefits and farmers' perception

Cost-benefit analyses help estimate the environmental and economic benefits as well as the costs needed to achieve environmental targets (e.g. threshold concentration for a given pollutant) or a minimum water treatment efficiency (e.g. 80% nitrogen removal) (Pearce *et al*., 2006; Mannino *et al*., 2008).

Whole Life Cycle (WLC) analyses have been carried out on Sustainable Urban Drainage Systems (SUDS) (Lampe *et al*., 2005; Duffy *et al*., 2008) and provide detailed information which can be used to infer costs of CFWs to some extent. Clift and Bourke (1999) defined Whole Life Costs as "the systematic consideration of all relevant costs and revenues associated with the acquisition and ownership of an asset”. WLC analyses improve the understanding of long-term investments, help to choose a cost-effective project at an early planning stage, provide explicit assessment and management of long-term risk and reduce financial uncertainties.

Main life cost stages mentioned for SUDS are (Lampe *et al.*, 2005): 1) Acquisition (feasibility study, commissioning, design, construction); 2) Use and maintenance; 3) Rehabilitation; 4) Disposal/decommissioning. The main costs considered which are also applicable to CFWs include life span, capital costs, operation and maintenance costs, monitoring costs, risk costs (e.g. due to flooding or pollution incidents), environmental costs (e.g. linked to water quality improvement, greenhouse gas emission), disposal costs, residual costs (residual value of the land after decommissioning) and discount rate and discount period.

In this study, most of the costs could be quantified using data provided by farmers and designers. However, these costs were variable due to differences between theoretical “standard contractor costs” and costs competitively tendered or obtained locally. Negative externalities were evaluated as described in detail later based on data for CH_4 and N_20 emissions from a Scottish and a Swedish constructed wetland, using an average cost of £50 kg^{-1} CO_2e (Stern, 2006), i.e. £1.3 kg^{-1} CH_4 and £15 kg^{-1} N_2O. However, some of these emissions would occur without CFWs, which would need to be taken into account when comparing different options.

Many of the social and ecological benefits (e.g. water quality improvement, biodiversity or landscape enhancement), which depend on the efficiency of the systems, on their ecological characteristics, or on personal judgment, could not be assigned a monetary value. In large scale surveys, contingent valuation methods (CVM) or other approaches (e.g. travel cost, hedonic, avoided cost or shadow project approach) are often used to value non-use or non-market use values (Pearce *et al.*, 2006; Costanza *et al.*, 1997; Bateman *et al.*, 1999; Yang *et al.*, 2005) but were not utilised in this study because of the limited number of farmers surveyed.

The core socio-economic information was obtained through semi-structured 2-3 hour face-to-face interviews with nine farmers (Appendix C) and through discussions with several experts (SAC, National Farmers Union Scotland, Ulster Farmers Union, private companies) (Appendix B), and was combined with data from Kevin Stewart (2008) (short interviews with seven farmers with CFWs and seven without) and information from the Scottish Government and other sources, e.g. SRDP 2007-2013 (Scottish Government, 2008a), BMP User Manual (Cuttle *et al*., 2007) and the CFW Design Manual for Scotland and Northern Ireland (Carty *et al*., 2008b).

Interviews allowed collection of information on farm management and practices, reasons for implementation of a CFW (e.g. own willingness or external pressure), willingness to pay for it, time taken to build it, costs involved during construction, maintenance activities, problems with parasites, pests, accidents, sediment removal and disposal, habitat enhancement, water availability in the ponds (potential use for irrigation) and other economic opportunities (e.g. duck shooting, tourism). The case of a large CFW built at Greenmount Campus (Ireland) in 2007 by CAFRE (College of Agriculture, Food and Rural Enterprise), according to current guidelines and for which detailed construction costs were available is presented in Section 6.3.1.3.

6.2.3 Comparison of alternatives for farmyard runoff management

Information on conventional storage and spreading of farmyard runoff and slurry in the UK and Ireland, as well as data from Integrated Constructed Wetlands used in The Republic of Ireland (Rory Harrington, *pers. comm.*; Aila Carty, *pers. comm.*; Carty *et al*., 2008) and lagoons and filters used in France (Cemagref *et al*., 1997; Guillaumin *et al*., 2003; Cemagref *et al*., 2004; CRAPL, 2007) was gathered for comparison with CFWs to give a broader overview of options available to farmers and expected costs and constraints.

6.3 Results

6.3.1 Costs and disbenefits of CFWs

6.3.1.1 Overall cost of CFWs

CFW construction costs (Table 6.2), excluding the cost of land, maintenance and monitoring, appeared comparable and relatively low, i.e. around £5000 for Scottish wetlands built under SEPA and Tweed Forum initiatives for which costs were available (Alan Frost, *pers. comm.*; Stewart, 2008), but were higher for wetlands built in Ireland, c. £20 000 for ICWs (Rory Harrington, *pers. comm.*; Aila Carty, *pers. comm.*) and c. £59 000 for the Glenstal Abbey Wetland (co. Limerick) built by MAXPRO ROI (Costello, 2004) and for the Scottish CFW built independently by a farmer (Farm 9).

In Scotland, the average overall cost for building CFWs (excluding land cost) was around £5000 for a whole system (constrained by financial help available at the time), ranging between £1600 and £7500, for CFWs with effective areas between 900 m^2 and 5000 m^2. Differences in the overall cost were mainly linked to differences in wetland surface area, earth volume excavated (e.g. CFW 15 was built with an excessive depth of up to 3 m), number of cells (e.g. CFW 16 with 16 small bays) or modifications of the steading (e.g. at CFW 3, the steading was modified to create adequate slope for dirty water drainage). On the other hand, the cost of single Irish ICWs is usually higher due to their larger area, and the more substantial earthworks, levelling and planting activities conducted. However, the price per unit area is comparable or lower to CFWs, in the order of £3 m^{-2}, illustrating economies of scale.

Table 6.2 Estimates of overall costs for CFWs and typical cost for ICWs: data obtained from designers and farmers using questionnaires (Stewart, 2008; Carty *et al.*, 2008b; Harrington, *pers. comm.*).

Farm	CFW treatment area[a] (m^2)	Total CFW area[b] (m^2)	Previous land use[c]	CFW construction cost (£)[d]	Overall cost[e] (£)	Cost (£ m^{-2})[f]
1	2260	4000	IG	3000	4100	1.8
2	5000	8000	UG	6500	8000	1.6
3	900	5000	IG	5000	15 000	16.7
4	1110	5000	IG	4400	5200	4.7
5	1770	7500	IG	4400	6000	3.4
6	1235	6000	UG	3000	4100	3.3
7	1170	1600	UG	3000	3500	3.0
8	9000	12 000	UG	NA (> 10 000)	NA (> 15 000)	1.7
9	NA	NA	UG	Provisional: > 10 000	Provisional: > 20 000	> 4
10	450	NA	A	NA	1600	3.6
11	120	NA	IG	NA	NA	NA
12	1024	NA	IG	NA	2500	2.4
13	1150	NA	A	NA	7000	6.1
14	950	NA	A	NA	5000	5.3
15	440	NA	A	NA	6000	13.6
16	450	NA	IG	NA	7500	16.7
					Median cost	***3.5***
ICW[g]	NA	NA	NA	NA	17 000-21 000	2.0
CFW G[h]	6000	12 000	UG	20 000	29 600	4.9
GAWMS[i]		20 000	NA	41 000	59 000	2.9

NA: Not assessed.
[a]Area in contact with waste water actually involved in treatment.
[b]Total fenced area, taken out of production. Estimated by farmer and/or plans.
[c]Land use previous to construction. A: Arable, IG: Improved Grassland, UG: Unimproved Grassland
[d]Costs excluding design, land, fencing, planting, modifications to the farmyard (e.g. concrete work).
[e]Costs including design, fencing, planting and farmyard modification costs but excluding land cost.
[f]Cost per m^2 of treatment area.
[g]Cost per ha for ICWs in Ireland (Harrington, *pers. comm.*; Carty, *pers. comm.*).
[h]Information from CAFRE (Contractor costs in 2007) based on a large CFW in Northern Ireland.
[i]GAWMS (Glenstal Abbey Wastewater Management System, Ireland) (Costello, 2004).

6.3.1.2 Main cost categories

The overall cost for CFW implementation included capital and operating costs and was split into eight main categories (Table 6.3). However, it could also account for negative externalities (not included in Table 6.3 but discussed below as a ninth category) such as greenhouse gas emissions, groundwater contamination or increased risk of transmission of waterfowl or insect-borne diseases (e.g. bird flu and blue-tongue disease). The derivation of individual cost categories is discussed in more detail below.

Cost categories and ranges were based on the interviews with farmers and designers as well as data from the literature on SUDS and CFWs (Steer *et al.*, 2003; Lampe *et al.*, 2005; Duffy *et al.*, 2009; Reay and Paul, 2008). The yearly capital costs were calculated by dividing the total cost by 20, 20 years corresponding to a reasonable lifetime of the CFW without major excavation. Total lifetime of a CFW could be between 50 and 100 years, but would involve sediment removal and replanting in all cells (Harrington, *pers. comm.*; Carty *et al.*, 2008a).

Table 6.3 Cost categories for the implementation of CFWs (Source: interviews with farmers, designers and advisers, and literature).

Cost category	Cost details	Cost range[1]
Capital costs		
1. Land	Cost of land, loss of productivity, loss of Single Farm Payments (SFP)	Land price: 2500-5000 £ ha^{-1} Average GM[2]: UG: 134 £ $ha^{-1}yr^{-1}$; A/IG: 350 £ $ha^{-1}yr^{-1}$; (+ Possible loss of SFP)
2. Site assessment, planning, design, supervision	Preliminary hydrological, soil, ecological, archaeological surveys, design plans, supervision	600-1000 (£ ha) 30 – 50 £ ha^{-1} yr^{-1}
3. Discharge permit	When required (e.g. Ireland): licence fee, water sampling, expert advice	1000-1500 (£ ha^{-1}) 50 – 75 £ ha^{-1} yr^{-1}
4. Construction	Earthworks, piping, lining, fencing, concrete structures, etc.	12 000-27 000 (£ ha^{-1}) 600 – 1350 £ ha^{-1} yr^{-1}
5. Planting	Plants and planting	5000-12 500 (£ ha^{-1}) 250 – 625 £ ha^{-1} yr^{-1}
6. Decommissioning	Earthworks, disposal of pipes and liners, sawing	2000 (£ ha^{-1}) 100 £ ha^{-1} yr^{-1}
Operating costs		
7. Maintenance	Regular inspections, sediment removal and spreading, replanting, vegetation harvesting	985-1516 £ ha^{-1} yr^{-1}
8. Monitoring[3]	Water sampling, ecological surveys	200-1000 (£ ha^{-1} yr^{-1})
Total cost		2350-5070 £ ha^{-1} yr^{-1}

[1]The yearly cost is calculated by dividing total capital cost by 20.
[2]Average Gross Margin as given in the SRDP 2007-2013 (Scottish Government, 2008a). A: Arable, IG: Improved Grassland, UG: Unimproved Grassland, SFP: Single Farm Payment.
[3]Monitoring cost depends on whether it is carried out and paid by local authorities or is a legal requirement for the farmer. It depends on sampling intensity, staff involved, distance to laboratory.

1. Land cost and potential loss of opportunity, productivity and loss of SFP

The value of the land lost depends both on the surface area and land use. The larger the farmyard, the larger the wetland required and the land taken. In addition, a buffer area is recommended between wetland cells (e.g. 5 m wide) and between cells and grazed areas/waterbodies (≥ 10 m) for maintenance and contingency, which considerably increases the overall area needed. If considering a simple configuration of four cells with length to width ratio of 2:1 (Figure 6.1), the larger the effective wetland area, the larger the ratio of effective area to total (fenced) area (Figure 6.2), and the smaller the additional construction cost due to economies of scale.

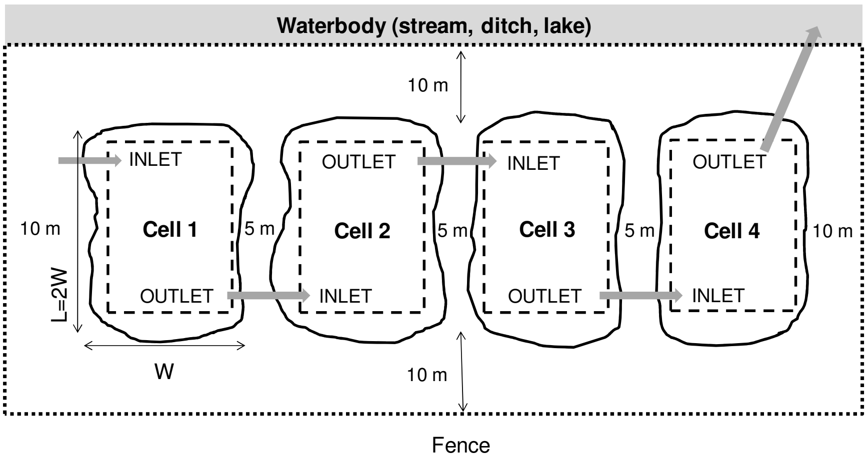


Figure 6.1 Schematic (not to scale) of a possible spatial configuration for a CFW comprising four cells with length (L) to width (W) ratio of 2, with surrounding buffer area (10 m from waterbodies/grazed land). Dashed lines: theoretical design of the cells; Arrows: farmyard runoff flow direction.

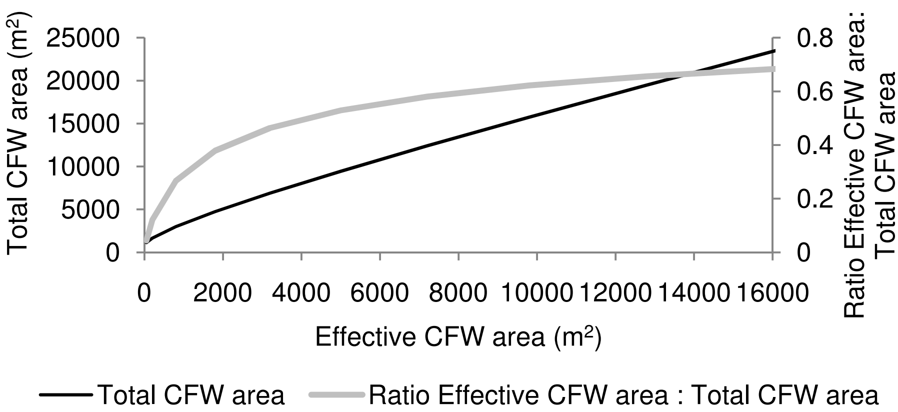


Figure 6.2 Total CFW area and ratio effective area: total area as a function of CFW effective area for the theoretical configuration shown in Figure 6.1.

Wetlands were built over unimproved grassland on five of the farms investigated, and improved grassland and arable land was used in six and four farms respectively, resulting in higher land costs in those cases. The cost of improved grazing land reached around £5000 ha^{-1} in 2009 (Alan Frost, *pers. comm.*), while rough grazing land was considerably cheaper, at around £2500 ha^{-1}. The strategic importance of the land lost also varied between farms. It was, as expected, greater for dairy farms, which rely heavily on improved grassland for cattle feeding, than for larger mixed beef and arable farms whose income comes from both livestock and crops, and for pig farms which rely on concentrated foodstuff.

Loss of opportunity occurred on all farms through loss in silage production or grazing material, loss of crop production or loss of renting area (e.g. Farm 1 rented land for sheep grazing to the neighbouring farmer at £125 ha^{-1} yr^{-1}). Average gross margins of 351.54 £ ha^{-1} for arable and improved grassland and 133.48 £ ha^{-1} for unimproved grassland are mentioned in the SRDP 2007-2013 (Scottish Government, 2008a) and could be used to estimate income foregone by the implementation of the CFWs.

The potential loss of single farm payment was mentioned by only three farmers, and is theoretically proportional to the area of land taken out of production. However, only two farmers mentioned actually losing money (Farm 2 with a loss of £200 yr^{-1} and Farm 8 with a loss of £300 yr^{-1}), the others managing to compensate for it by reorganising eligible land.

2. Cost of site assessment, design, supervision

Site assessment, design and supervision for the CFWs studied were conducted by different companies, involving topographical and hydrological surveys, but no in-depth ecological assessments. The price ranged usually between £400 and £1000, depending mainly on the company involved, size of the farm and wetland and site characteristics (topography, hydrology, soil type) (Alan Frost, *pers. comm.*).

3. Planning and discharge consent

In Scotland, the implementation of CFWs does not require discharge licences in most cases. However, in Ireland, there is a fee of c. £350 (Monaghan County Council, 2009; Aila Carty, *pers. comm.*), for obtaining a discharge application licence (as mentioned in the Local Government Regulations) for which farmers need to demonstrate that the proposed discharge is not going to have a negative impact on the receiving watercourse. Consequently in Ireland, additional costs are incurred for water sampling (minimum of four samples per year needed) and expert advice on the assimilative capacity of the receiving waterbody, the overall cost reaching then about c. £1300 on average (Aila Carty, *pers. comm.*).

4. Cost of construction and ancillary works (e.g. digging, piping, lining, fencing)

Very efficient CFWs have been built in Ireland and Northern Ireland for costs ranging between £17 000 and £39 000 ha^{-1}, including planting but excluding land cost. Construction may involve one or two heavy machines for three days or more (about 1 week per ha). Overall construction costs are site-specific linked to the distance from the farmyard and piping work, excavation of ponds, reworking of topsoil, stock-proof fences if the wetland is next to grazing land, fencing being generally estimated at £3 m^{-1} or more (Cuttle *et al.*, 2007).

Occasionally, other activities such as modification of the steading or drainage system (drain diversion or blockage) were required. CFWs were built over clay soils and consequently no artificial liners were used, to reduce costs. Alternatively, in the case of more permeable soil, a flexible geomembrane liner (e.g. HDPE liner) would be required, with a considerable additional cost of a minimum of about £40 000-50 000 ha^{-1} (depending on the thickness and quality of the liner) (Alan Frost, *pers. comm.*). Infiltration to groundwater is a critical issue and liners (geomembrane or compacted clay) are compulsory in France in the first two cells of lagoon systems (Guillaumin *et al.*, 2003) and is increasingly promoted in Scotland, since most soils do not provide the recommended impermeability (hydraulic conductivity < 10^{-8} m s^{-1}).

5. Cost of plants and planting

The cost for planting depended mainly on the planting density and species used (reeds, bulrushes, grasses), and on plant or seed availability (e.g. if plants can be transplanted from an existing wetland or pond and if the area was already a wet grassland before construction). Guidance on transplanting usually recommends sourcing from accredited nurseries, to avoid damage to existing habitats and inadvertent introduction of exotic species. The recommended planting density is about 2 plants m^{-2}, at least in the first two cells of the CFW, and the price for planting is about £0.9 to £1 per plant (£0.4 to £0.5 for the plant itself and £0.5 per plant for labour), i.e. a minimum of £10 000 ha^{-1} planted, representing a significant part (about 40%) of the overall cost. Planting cost could be cut down by 2 or more if the farmer is allowed to obtained free locally sourced plants and decides to do the planting himself.

6. Costs of decommissioning the CFW

In some cases, for example, if farming activity ceases, land is sold, the CFW reaches the end of its life time, or legislation changes, it might be necessary to decommission the wetland, carrying out simple earthworks to fill the CFW or to transform it. In such a case, sediment can be left *in situ*, cells can be filled using available material, pipes and liners have to be removed and properly disposed of, and vegetation can be harvested to use in other CFWs or left *in situ*. The CFW can also be transformed into an amenity wetland, if a permanent source of water is available. A realistic one-off decommissioning cost of at least £2000 should therefore be included.

7. Cost of maintenance by farmers

Maintenance costs for CFWs are rarely presented in the literature, due to limited experience with those systems, limited maintenance activities carried out by farmers, and no formal recording of the nature and time spent on maintenance activities.

Maintenance activities appeared to be neglected for the CFWs investigated, mainly due to the absence of regulatory requirements, lack of financial incentives and lack of information and understanding of the maintenance needed. Only the farmer who built the CFW independently carried out maintenance activities on a monthly basis (e.g. removal of weeds, planting). Indeed, regular maintenance activities are needed to ensure CFW integrity and function, such as removal of material obstructing pipes, grass cutting on the edges, removal of sediment, replanting areas where establishment failed, and level control. Based on recent recommendations (Carty *et al.*, 2008b) and on the French experience with similar treatment options (CRAPL, 2007), general maintenance time required can be estimated between 30 and 50 hours annually, i.e. c. £235 to £391 yr^{-1} (at £7.82 h^{-1}, Scottish Government, 2008a).

The main maintenance cost results from the need to dredge and replant the first pond every seven to ten years (Scholz *et al.*, 2007b; Carty *et al.*, 2008). In Irish ICWs, sediment has been shown to accumulate at an average rate of 3 cm yr^{-1}, usually with higher accumulation close to the inlet (6 to 7 cm yr^{-1} in the first cell), and lower further away (1 to 2 cm yr^{-1}) (Scholz *et al.*, 2007a and 2007b). A 10 year desludging frequency is recommended in the first cell and 20 to 60 year frequency in the others. Assuming using a contractor twice in 20 years, with a one-off cost of £7500, this amounts to an annual desludging cost to £750 over 20 years. Total maintenance cost may therefore vary between £985 and £1516 ha^{-1} yr^{-1}.

Metals (e.g. Zn, Cu, Fe) are present in yard or roof runoff (from galvanized surfaces, fuel spillages) in variable concentration (Edwards *et al.*, 2008), but their quantity in CFW sediment has not been assessed accurately. If sediment was shown to contain amounts of heavy metals and noxious pollutants under given thresholds and was not classified as a hazardous waste (see Waste Technical Guidance, EA *et al.* (2005)), it would be exempt from waste regulations. Sediment could therefore be piled up close to CFWs (but not directly adjacent to avoid runoff back into the system) for drying and storage before being spread on agricultural land, to make use of its high P content and fertilizing value. Phosphorus accumulation between 15 and 140 kg ha^{-1} yr^{-1} has been reported within six ICWs (Scholz *et al.*, 2007a).

8. Cost of monitoring by farmers and competent authorities

Water and ecological monitoring of CFWs, including visual (e.g. growth of algae, sewage fungus) as well as chemical monitoring (e.g. NH_4-N, TP) should involve farmers as well as the environmental regulatory authority. It appeared neglected in nearly all farms visited, mainly due to lack of financial incentives and staff, and to difficulties in monitoring at relevant times, i.e. during storm events. Most of the CFWs were indeed completely unmonitored after construction, resulting in poor performance being undetected and in the lack of corrective actions. Due to absence of regulation and enforcement in the field, the cost of monitoring which could be borne by farmers or local authorities is currently unknown, and could be significant if the "polluter pays" principle is fully applied and farmers are required to address this issue.

However, water sampling and analysis is necessary to ensure that the effluent is not impacting negatively on the receiving waterbody, especially in sensitive areas, and during rainy periods when peak pollution occurs. Experience shows that monitoring can be done for less than £1000 per year (Aila Carty, *pers. comm.*), depending on the number of water samples, quality parameters measured and distance from the laboratory or office and staff available. Nevertheless, the design and large size recommended by SEPA for CFWs should ensure the release of an effluent of acceptable quality, whatever the climatic conditions, which would reduce the need for costly sampling and chemical analyses.

9. Cost of externalities: greenhouse gases emissions

Natural wetlands release significant amounts of methane (CH_4), and nitrous oxide (N_2O) in smaller quantities, contributing therefore to global warming (IPCC, 2007). At the global scale, they are responsible for 72% (115 Tg yr^{-1}) of the CH_4 produced by natural sources (160 Tg yr^{-1}) and c. 20% of total sources (535 Tg yr^{-1}) (IPCC, 2007). Saarnio *et al.* (2009) estimated that 5.2 Tg yr^{-1} CH_4 was emitted from European wetlands and waterbodies.

Constructed wetlands might release even greater amounts of CH_4 and N_2O than natural ones (Fey *et al*., 1999; Johansson *et al*. 2004). Investigation of CFW2 (Van de Weg, 2006; Paul, 2007; Rao Pangala, 2008) receiving field drainage enriched in NO_3 showed a significant release of CH_4 (especially in summer when bacterial activity is enhanced) estimated at between 450 kg ha^{-1} yr^{-1} (Paul, 2007) and 970 kg ha^{-1} yr^{-1} (Rao Pangala, 2008), valued between £585 ha^{-1} yr^{-1} and £1261 ha^{-1} yr^{-1}, using an average cost of £1.3 kg^{-1} CH_4 (Stern, 2006). Comparable results were found in Sweden, where Johansson *et al*. (2004) reported an average CH_4 flux of 515 kg ha^{-1} yr^{-1} from a CW treating municipal wastewater. Agricultural drainage water is also known to release N_2O (Reay et *al*., 2003), but in the case of the Scottish CFW, the emission of N_2O emissions was small (Rao Pangala, 2008), and its cost was estimated around £59 ha^{-1} yr^{-1}, using an average cost of £15 kg^{-1} N_20 emitted.

Consequently, CFWs, like riparian buffer areas (Hefting *et al*., 2003), may contribute to “pollution swapping” (Reay, 2004; Van de Weg *et al.*, 2008), improving water quality locally but contributing to the global issue of climate change. To reduce those emissions, iron ochre and gypsum may be applied to the CFW surface and have been shown in preliminary field and laboratory experiments to cut CH_4 emissions by 20% to 50% (Rao Pangala, 2008), reducing therefore the cost of this externality.

6.3.1.3 Case study: Greenmount College CFW

This CFW was built in 2007 at a dairy farm on Greenmount Campus (Antrim, Northern Ireland), in a joint project involving CAFRE (College of Agriculture, Food and Rural Enterprise), AFBi (Agrifood and Bioscience Institute), EHS (Environment and Heritage Service, Northern Ireland) and The Queens University Belfast. The CFW received farmyard runoff from an area of 3000 m^2 and parlour washings from a 180 cow dairy unit, estimated at 5 m^3 per day. Clean and dirty water were separated at source. The following calculations and cost benefit analysis were based on a constructed wetland of 6000 m^2 effective area (ratio wetland: yard areas of 2). Details of the system are presented in Table 6.4 (Martin Mulholland, *pers. comm.*).

Table 6.4 Details of the origin of dirty farmyard water and of the design of the CFW at Greenmount College (Source: DARD).

Description	Quantity
Yard area to paddocks	700 m^2
Dirty yards	1200 m^2 (25% of area alongside forage house)
Unroofed silos	1100 m^2 (beef silos and apron)
Total impermeable area	3000 m^2
Total wetland area	6000 m^2
Number of cells	5 (equal size - rectangular)
Pond/cell area	1200 m^2
Depth of excavation	0.4 m
Width of bank at top	5 m
Distance from top of bank to wetland bed	1.3 m
Depth of soil covering clay liner	0.15 m
Earth volume excavated	480 m^3
Earth area shaped into banks	983 m^2
Earth volume returned on top of liner	180 m^3

Table 6.5 presents the construction costs, which are based on FNMS (Farm Nutrient Management Scheme) 5A and 5B (DARD, 2005a, b) and contractor earthmoving costs and actual planting costs have been calculated pro rata.

The total cost for the CFW was about £29 600 (i.e. c. £50 000 ha^{-1}), including land and fencing. Pipework and earthworks were the greatest costs, representing 35% and 34% respectively of the overall construction cost, followed by planting, which represented 26%. The cost per unit area was about £5 m^{-2}, about 35% higher than the median cost of smaller Scottish CFWs (c. £3.5 m^{-2} or £3.3 m^{-2} when removing systems with costly steading modifications). The CFW is very efficient, achieving on average more than 90% mass reduction for BOD_5, N and P (Martin Mulholland, *pers. comm.*).

In practice, the cost for pipework could be much lower, depending on the distance to farmyard and distance between ponds. For example, for a hypothetical minimum distance from the farmyard of 100 m, and a minimum distance between ponds of 5 to 6 m (c. 15 m pipe needed between each pond), and 10 m from a waterbody (15 m pipe), the total pipe length could be less than 175 m, representing a cost of £2975, only 30% of the cost for Greenmount CFW. The total cost of the system would then be £22 371 (£33 682 ha^{-1}, £3.4 m^{-2}), equivalent to Scottish CFW.

Table 6.5 Details of the capital costs associated with the construction of the CFW at Greenmount College (Source: DARD; Costs estimated in February 2007).

Activity	Cost
Bulldozing	2 (£ m^{-3})
Loading, transporting and levelling soil	3.5 (£ m^{-3})
Shaping banks	0.4 (£ m^{-2})
Pipework	17 (£ m^{-1})
Earth movement per pond	1983 (£)
Total earthworks (5 ponds, 50 m x 24 m)	9916 (£) (16 527 £ ha^{-1}) [1]
Connecting pipework between cells: 250 m at £17 m^{-1}	4250 (£)
Connecting dirty water to CFW: 350 m at £17 m^{-1}	5950 (£)
Overall pipework	10 200 (£) (10 200 £ ha^{-1}) [2]
Inspection chambers: 6 chambers at £50 each	300 (£) (300 £ ha^{-1}) [2]
Plants	4500 (£)
Planting labour	3000 (£)
Overall planting	7500 (£) - 12 500 (£ ha^{-1})
Fencing[3]	1380 (£ ha^{-1})
Total estimated cost[4]	29 296 (£) - 39 707 (£ ha^{-1})
Estimated land cost (1.2 ha lost in total, at £250 ha^{-1})	300 £ yr^{-1}

[1]Earthworks costs are assumed to increase linearly with increasing area for simplification (in reality, a large part of the cost is independent of the size).
[2]Pipework and inspection chamber costs depend on the number of cells and distance between them.
[3]Fencing cost is not available (fencing might not have been implemented), but was estimated for 460 m fence (160 m x 70 m; 1.12 ha area) at £3 m^{-1}.
[4]Excluding land cost and maintenance.

6.3.2 Benefits of CFWs

CFWs can bring numerous benefits at the farm, catchment or national scale (Table 6.6) which several authors have tried to value (Costanza *et al*., 2003; Yang *et al*., 2005). For famers, benefits are direct or indirect, mostly related to the fact that CFWs help deal with farmyard runoff and decrease the need for storage and spreading of dirty water, provide contingency options and reduce the risk of being fined and prosecuted, and contribute to improving the image of the business. At catchment and national level, the main benefits arise from the improvement in water quality and subsequently amenity and ecological value of rivers and lakes, helping to ensure compliance with European Directives and to avoid sanctions. The main benefits are discussed below in more detail.

Table 6.6 Main benefits of CFWs for society and farmers.

Main benefits obtained from CFWs	
For society	For farmers
1) CFWs improve water quality, decrease water treatment costs, enhance amenity value and help achieve compliance with European Directives and avoid sanctions.	A) CFWs reduce water pollution, are used for contingency and consequently help avoid fines, prosecution and business closure, and improve the image of the business.
2) CFWs attenuate small-medium size floods at catchment scale.	B) CFWs reduce slurry or dirty water storage need and spreading costs, and increase fertilizing value of slurry; reduce fertiliser costs if P-rich sediment is spread on land; may reduce cost for roofing and separation of dirty and clean water.
3) CFWs enhance biodiversity and habitat on farm and at regional scale.	C) CFWs are partly financed by the Government (Capital Grant).
4) CFWs enhance landscape.	D) CFWs can be used for recreation or as “game reserves”.
5) CFWs can be used for research, education and recreational purposes.	E) CFWs may have secondary uses (e.g. irrigation).

6.3.2.1 Benefits of CFWs for society

1) Water quality improvement

Water quality improvement is one of the major benefits obtained from the implementation of CFWs for society. Indeed, CFWs help control the pollution at source, and also decrease the risk of diffuse pollution associated with slurry being spread during unfavourable conditions (e.g. on saturated soils). The subsequent water quality improvement in rivers, lochs and aquifers brings multiple benefits: it reduces drinking water treatment costs, decreases the risk of eutrophication and sedimentation, providing higher quality habitat for fish and wildlife and higher amenity value which in turns benefit fisheries, tourism and prices of properties adjacent to waterbodies (D'Arcy *et al.*, 2000; Pretty *et al.*, 2003). Nitrate reduction in water has been for example valued by Pretty (2006) between £0.034 and £0.048 kg^{-1} NO_3-N. Using an average value of £0.04 kg^{-1} NO_3-N removed, the saving from NO_3-N reduction by CFW1 and CFW2 was evaluated at £6 yr^{-1} and £20 yr^{-1} respectively.

2) Flood attenuation

Similarly to SUDS, if CFWs are large enough and in high numbers, they can provide additional storage capacity (due to evaporation and infiltration) close to the runoff source and decrease peak flows and velocity. They subsequently contribute to flood attenuation within the catchment, reduce flood damage costs and help ensure compliance with the European Floods Directive (2007/60/EC) and the Flood Risk Management bill in Scotland (2009) which promotes sustainable flood management (Dunne *et al.*, 2005; Lampe *et al.*, 2005; Woods-Ballard *et al.*, 2005; Woods-Ballard *et al.*, 2007; Carty *et al.*, 2008a; Scottish Parliament, 2009).

3) Biodiversity and habitat enhancement

CFWs offer several ecological benefits: they host macroinvertebrates, aquatic plants, provide feeding, resting and breeding sites for birds, mammals, amphibians and reptiles. Their ecological value depends mainly on water quality, structural heterogeneity and proximity to other wetlands influencing colonization (Williams *et al.*, 1999; Williams *et al.*, 2007; Coletto, 2008). Larger vegetated multi-cell systems receiving lightly contaminated water appear more favourable to wildlife than smaller highly contaminated wetlands. Moreover, the recommended absence of trees in the close vicinity of CFWs (to avoid root damage to banks) which limits perching sites for birds of prey, and a mixture of short and tall vegetation close to farmland might benefit ground nesting birds (e.g. snipe, lapwing). Nevertheless, a monetary value for biodiversity conservation cannot easily be given. The ecological aspect valued most by farmers interviewed was colonization by waterfowl, whether due to an interest in duck shooting or simply for aesthetic reasons.

4) Landscape enhancement

The added landscape value of CFWs (even of the simplest systems) was mentioned by all the farmers interviewed, but without relation to the overall price of their property. No contingent valuation was carried out to investigate if farmers would be willing to pay more for a farm with a CFW, but farmers seemed to value the presence of the CFW in terms of aesthetics and improvement of the image of their business. Generally, the landscape value depends on the design and on the way the system fits into the overall environment. Aesthetics are improved by irregular shapes and complex mosaic patterns, a combination of open water with shallow wetland areas, diverse vegetation including colourful flowers (e.g. *I. pseudacorus* or *N. alba)*. The presence of a CFW might affect surrounding land price, especially in or close to urban areas. SUDS have been shown to contribute to increased house prices in their vicinity (Apostolaki and Wallingford, 2004; Woods-Ballard *et al.*, 2007). However, if perceived as a threat for safety and health, such a system could decrease the willingness of people to build in the vicinity and reduce house prices. For example, using a hedonic property price method, Bin and Polasky (2005) showed that wetlands in rural settings in North Carolina (USA) can bring disamenities and can negatively affect house prices, due to issues such as mosquito proliferation or decreased agricultural production. Doss and Taff (1996) found a preference for scrub-shrub and open water wetlands over forested or emergent-vegetation wetlands.

5) Recreational, scientific and educational value

These values depend on design, size, location, safety and access by public and teaching or research staff (Apostolaki and Wallingford, 2004; Woods-Ballard *et al.*, 2007). CFWs can be used for research or education purposes if adequate modifications are made easing access, monitoring and safety and if the owner's permission is granted, to allow investigation of their performance over time or their ecology. They can also be used as recreational sites for bird or wildlife watching, walking or for picnics. The recreational value and perception of farmers of most of the Scottish CFWs visited appeared negatively affected by their small size, lack of vegetation and unfavourable habitat for wildlife, odours or unpleasant appearance, and presence of septic tank solid wastes.

6.3.2.2 Benefits of CFWs for farmers

A) Farmyard runoff management and avoidance of fine and prosecution

CFWs were seen by all farmers as a tool to deal with farmyard dirty water and for seven of them, as a contingency measure to contain accidental pollution accidents (e.g. slurry spillage). This aspect appeared to be a strong incentive to construct CFWs for dairy and pig farmers, for whom the risk of pollution due to tank failure is greater. Amongst the farmers interviewed, the risk of being fined and prosecuted was one of the main incentives for CFW construction, together with grant availability. In theory, farmyard dirty water has to be collected and stored in slurry tanks, and later spread when conditions allow. However, in reality, it appeared to enter drains and to flow directly to ditches, streams or lakes in all farms investigated, due to the difficulties and cost of separating roof and yard runoff and storing it. Nevertheless, there is increasing pressure from SEPA to manage farmyard runoff following SSAFO Regulations, whether by storing and spreading, or by diverting it in a CFW.

All farmers interviewed were aware of the risk of fines, prosecution and business closure associated with pollution incidents (see questionnaire in Appendix B). Fines up to £5000 have been given in the case of accidental spillages or repeated pollution incidents, as was the case for Farm 4, which was fined £5000 for a 380 000 litres slurry spillage in a stream in 2003 following a slurry tank valve failure. A maximum theoretical fine of £20 000 and six months in jail was mentioned by one farmer. CFWs are therefore a practical way to avoid penalties and to improve the environmental image of the business.

B) Farmyard runoff management and reduction of storage and spreading costs

CFWs were mentioned by all farmers as a cost-effective way to deal with dirty farmyard water. Indeed, avoiding mixing large volumes of contaminated water with more concentrated slurry improves the fertilizing value of effluent to be spread, and reduces storage and land spreading costs, especially when existing storage capacity is nearly full. The overall cost for slurry storage can be estimated between £30 m^{-3} and £70 m^{-3}, including excavation and erection of the store itself. The cost depends on store type, e.g. a shuttered slatted tank, 2.4 m deep with piers, heads and slats costs £47-£70 m^{-3}, and an above ground store with reception tank and pump costs £30-£55 m^{-3} (DARD, 2009). The cost can be much higher depending on topography and local conditions. Land spreading by contractor costs in the order of £17 to £22 h^{-1} using a 9000 l slurry tanker (DARD, 2009). Merrilees (2004) mentions an average volume of dirty water of 1400 m^3 yr^{-1} per farm (calculated for 48 farms selected in Cessnock Water (Ayrshire), Ettrick Bay (Bute), Sandyhills (Dumfriesshire) and the River Nairn (Invernesshire) catchments), and an average spreading cost in excess of £0.8 m^{-3}, i.e. an annual cost of at least £1200. Much larger volumes are often generated when roof water is not diverted (Aitken, 2003; DEFRA, 2006a).

The possibility of managing farmyard runoff at a lower cost using a CFW also reduces the need for roofing of extensive areas, with the cost for roofing reaching around £40 to £50 m^{-2} (Steven Andrew, *pers. comm.*). Additionally, the possible use of CFW for septic tank overflow treatment is another incentive, saving money on tank maintenance and emptying, which costs c. £100, around £50 yr^{-1} on average if emptied every two years.

C) Obtaining a Government Grant

A potential incentive for CFW construction is currently in place in Scotland, consisting of a capital grant for wetland construction given through the SRDP 2007-2013 (Scottish Government, 2008a; Francis Brewis, *pers. comm.*). The grant amount is variable, with ceilings of 40% of eligible costs in non Less Favoured Areas (LFA) and 50% in LFA, with a 10% premium on the ceilings for investments undertaken by young farmers. Eligible costs include capital costs only, such as construction, piping and planting. However, no financial support is provided for maintenance and monitoring.

D) Amenity aspects

Some degree of amenity value of the CFWs was mentioned by seven out of the nine farmers personally interviewed, mainly related to the presence of waterfowl, for shooting (e.g. ducks) or visual interest (e.g. swans). The small size of most CFWs and reduced vegetation colonization decreased significantly their amenity value. The efforts put into amenity enhancement of CFWs and the associated costs (e.g. for maintenance, access) were smaller than for SUDS, because CFWs were located on private land, and further away from the public (Apostolaki and Wallingford, 2004; Lampe *et al.*, 2005).

E) Other benefits

None of the CFWs examined here were used for fishing or for irrigation, due to health concerns, lack of need and the small size and depth of the systems. However, it is common practice in France to use reedbed or pond effluent for irrigating pasture or tree plantations (e.g. eucalyptus, willow or orchards). Since climate change is expected to drive changes in terms of water availability (longer periods of drought), building CFWs with larger storage volumes for irrigation could be cost-effective and more attractive to farmers.

The extraction and use of reeds for thatching was sometime mentioned as a potential secondary benefit of constructed wetlands. However, the quality of the reeds has been shown to be affected by rapid growth in constructed wetlands, which strongly limits their actual use (Richard Cooper, *pers. comm.*), and the cost of extraction would be too high compared to the benefits, making this activity not viable economically. This is illustrated by the case of the Tay reedbed, the UK's largest natural reedbed (c. 410 ha), where commercial harvesting ceased in 2005 due to poor production and international competition.

6.3.3 Farmers' perception of CFWs

Most CFWs investigated in this survey were primarily implemented under pressure from the authorities, whether due to chronic pollution or to accidental spillages, and environmental awareness was only a secondary driver. The difficult economic situation for farmers at the time, i.e. low price of milk and meat, was not encouraging and business profitability relatively low, which might have hindered investments in CFWs. Currently, with the higher food prices and higher value of agricultural land, farmers may be even less willing to take land out of production.

Based on the information they received initially and on the type of systems built until now in Scotland (relatively small and cheap), farmers perceived CFWs as practical and cheap tools to deal with farmyard runoff, septic tank overflow and other contaminated effluents, as well as with accidental spillages. This firstly avoids fines and prosecution, and secondly allows them not to invest in additional storage capacity.

Moreover, CFW were wrongly perceived or mentioned as "maintenance free", due to lack of knowledge and information on the maintenance needed, lack of financial incentive, and lack of external control. These misperceptions, together with the lack of information or understanding of CFW limitations often leads to their unsustainable use, due to addition of excessive effluent volumes and no maintenance activities being conducted.

For the eight farmers without CFWs, the four main reasons mentioned for not implementing such a system were: 1) the perceived absence of pollution (or only regarded as minimal) on the farmyard due to limited traffic of animals over impervious surfaces, roofed feeding or transit areas for livestock, no filling or washing of pesticide sprayers on the farm, use of contractors for pesticides or sheep dipping; 2) High cost of CFWs and lack of financial incentives; 3) Lack of information on CFWs and uncertainty regarding their efficiency, and 4) Loss of productive land (especially arable and improved grassland). Another concern mentioned by farmers was the possible increased risk of bird flu propagation by waterfowl attracted by wetlands, and the subsequent risk for poultry farms.

6.3.4 Comparison of CFWs with traditional dirty water management methods

Several methods for dirty water management are used across the UK and in Ireland, including: constructed wetlands; one week storage in slurry store or dirty water tank, and irrigation to grassland with sprinklers or travelling irrigators; two weeks storage and tanker spreading to grassland; four weeks storage in an artificially-lined earthen bank tank and tanker spreading (Brewer *et al.*, 1999; Culleton *et al.*, 2005). Storage is in clay or artificially-lined earthen bank lagoons, concrete-wall lagoons or slurry stores. In Northern Ireland, dirty water tanks have less stringent standards than slurry stores and can therefore be built at a lower cost.

Using the example of the CFW at Greenmount College, a comparison was made between the cost incurred by storage in slurry tank and tanker spreading (a common option, although more expensive than irrigation) and the cost involved in wetland construction, over a 20 year period, which is the minimum recommended life span for slurry stores (SSAFO) and a reasonable life span for CFW without major renovation (Table 6.7). The following assumptions were used (costs based on information from DARD, 2008): yard area: 3000 m^2; rainfall: 4 mm d^{-1}; dairy washings: 5 m^3 d^{-1} (180 cows); slurry storage: £40 m^{-3}; land spreading cost: £20 h^{-1} (9 m^3 tanker); wetland maintenance: 1st pond dredged and replanted every 5 to 7 years at a cost of £7500. Assuming 100% runoff and capture of rainfall, 1910 m^3 of dirty water would be produced over a 16 week period (i.e. 17 m^3 d^{-1} on average) (Culleton *et al.*, 2005). The storage capacity needed therefore reaches £76 400 for an above ground slurry store at £40 m^{-3}, and land spreading of the volume generated annually is estimated at £1415 yr^{-1}. Land taken out of production for the constructed wetland is valued at an annual rental cost of £250 ha^{-1}.

Table 6.7 Simplified cost comparison between management of farmyard dirty water using a CFW, and storage in slurry tank and tanker spreading on grassland.

Constructed Farm Wetland	Storage (slurry tank) / land spreading (tanker)
Annual capital cost (over 20 years): £1485 (Overall cost: £29 700)	Annual cost (over 20 years): £3825 (Overall cost: £76 500)
Annual maintenance cost (checking, dredging, planting): £1125	Annual field spreading cost: £1415
Annual rental of 1.12 ha of land: £280	
Annual cost: £2890	Annual cost: £5240

Using these assumptions, a CFW represents an annual saving of £2350, compared to conventional storage and land spreading. However, in the case of a dry farm or in places where dirty water could be spread every 7-8 weeks (if enough land is available and weather conditions are suitable), the storage cost would only be about half (£38 250) and consequently, the wetland option would be only £437 cheaper than storage (£3327). Nevertheless, the CFW would reduce the reliance on dry weather and the risk of slurry being washed out if spread under unsuitable conditions. It also represents a significantly smaller initial investment.

6.3.5 Comparison of CFWs with treatment systems used in France

Several alternatives for the management of “lightly” contaminated farm water (defined as containing less than 1000 mg l^{-1} total nitrogen) have been validated by the French Government and are being promoted and subsidized (CRAPL, 2007; Dollé *et al.*, 2007).

Effluents treated are farmyard runoff, milking parlour waters, seepage from unroofed middens, cheese factory wastewater (excluding lactoserum) and occasionally domestic effluent (if agreed by departmental administrations). Primary treatment (I) of regular or storm generated wastewater can be done through a straw filter (SF) (a basin made of rectangular straw bales) combined with two water collection basins (e.g. 6 m^3 each), or through a larger single sedimentation basin (SB) made of concrete or using a geomembrane. Secondary (II) and tertiary (III) treatments are mainly by one-stage vertical reed filters with recycling of the effluent or lagoons, followed by irrigation over grassland (including in winter time) or over vegetated areas vegetated with woody or non woody species. Water leaving a secondary treatment is never discharged directly to a waterbody. The use of shallow surface flow vegetated wetlands has not been recommended since 1997 (Cemagref *et al.*, 1997), due to land requirements, maintenance constraints and efficiency uncertainties.

Table 6.8 presents the most common options with their associated costs, excluding cost for land. Lagoons are sized based on annual loads of Chemical Oxygen Demand (COD), Total Nitrogen (TN) and ammoniacal nitrogen (NH_4-N), aiming at a minimum of 80% mass reduction. Surface areas recommended are 0.15 m^{-2} kg^{-1} for COD, 1.7 m^{-2} kg^{-1} for TN and 2.4 m^{-2} kg^{-1} for NH_4-N. Vertical filters are sized according to BOD inputs (0.8 to 1.2 m^2 per Inhabitant Equivalent).

Table 6.8 Options for management of lightly contaminated farmyard runoff with primary (I), secondary (II) and tertiary (III) treatments, and their total costs (2007 estimates for a herd of 40-50 dairy cows) (Cemagref *et al*., 2007; Dollé *et al.*, 2007).

Option (II treatment)	Constraints and work needed	Total cost (in euro, VAT not included)
1. One stage vertical reed filter with recycling (three cells in parallel; 4 plants m^{-2}; 80% of the effluent sent back to I treatment for denitrification) I treatment: SB alone or SF with 2 basins III treatment: grass or woody vegetated area	1.5 m difference between yard and outlet; 3 layers of graded stones and gravels (diam. 20-40 mm, 10-20 mm, 3-6 mm); electric pump; frequent monitoring; maintenance of I: emptying SB twice a year; maintenance of II: weed control, reeds cut each year, pipe clearing; maintenance of III: vegetation; input location changed weekly.	C: 24 000-30 000; F: 15 000-20 000 (Maintenance: € 600-1000 yr^{-1})
2. Lagoons (3 non vegetated basins in series) I treatment: SB or SF III treatment: grass or woody vegetated area	Large area (1500-2000 m^2); only for clay soils (soil survey required); experienced contractors and designers; geomembrane if soil is not suitable; maintenance of II: grass cutting on the edges, pipe clearing.	C: 11 000-20 000; F: 9000-15 000 If geomembrane: C: 21 000-35 000; F: 18 000-29 000
3. Irrigating over grassland (1) perforated pipes; (2) sprinklers in line (3) travelling sprinklers I treatment: SB	(1) for small areas (< 1 ha); need to burry distribution network; move pipes frequently; high-maintenance equipment (calibration, cleaning); pastures need to be close to farm; suitable soils needed; rotating grazing; manual pumps; not possible during freezing periods.	(1) C: 9000-18 000; F: 6000-12 000 (2) C: 12 000-20 000; F: 8000-14 000 (3) C: 15 000-25 000; F: 10 000-17 000
4. Vegetated filters (non woody species) I treatment: SB or SF	Constrained by soil types (permeable surface horizon, impermeable deeper horizon) and effluent quality; maintenance of I; change input location weekly; planting and vegetation maintenance.	C: 8000-10 000; F: 4000-6000
5. Treatment through tree plantation (I treatment: SB or SF)	Constrained by soil types and effluent quality; bury pipes; change input location weekly; vegetation maintenance (35-40 h yr^{-1}).	C: 11 000-13 000; F: 8000-10 000

Notes: SB: Sedimentation Basin; SF: Straw Filter; C: built by contractors; F: built by the farmer.

Self-construction is possible for all options and the costs are c. 30% lower than commercial construction costs, creating a significant incentive for farmers. However, self-construction requires skills and experience, and professional advice is therefore indispensable. As shown by Table 6.8, the cheapest options are those relying on irrigation over areas planted with woody species (options 4 and 5) which does not rely on expensive equipment, followed by irrigation over grassland, lagoons and vertical filters. However, the cheapest options require significantly larger areas of land and their use is therefore restricted. Vertical reed filters are more expensive because they involve concrete work, pumping (electricity dependant) and require regular maintenance (30 to 40 h yr^{-1}, with a maintenance cost of 600 to 1000 £ yr^{-1}), but they can be applied to a greater range of situations, since land requirements are smaller, and they are more controllable too. Their cost is in the range of the cost of CFWs promoted in Ireland, but their efficiency in terms of phosphorus removal is often limited. The cost of non vegetated lagoons is also comparable to CFWs, and maintenance requirements are lower. However, the use of a geomembrane is often needed (at least in the first two cells), which can increase costs significantly.

6.4 Discussion

6.4.1 Factors influencing adoption and sustainable use of CFWs

In this study, adoption of CFWs depended on numerous factors, which have been reported in several studies of BMP uptake by farmers (e.g. Turpin *et al.*, 2005). Factors related to the CFW itself were the most critical, and included the perceived need for the CFW (did the farmer consider that his steading is contaminated and that he is polluting?), direct financial benefits and secondary benefits (fine avoidance, improved image of the business), location (e.g. land use type and strategic importance) and scale of implementation, capital and running costs of the CFW, actual or perceived efficiency (did the farmer believe that a CFW would solve the issue?), maintenance requirements (e.g. sediment removal, vegetation harvesting) and person or organisation promoting the BMP (e.g. regulators such as SEPA or advisers such as SAC, which are perceived differently due to their different goals, missions and behaviour).

Factors related to the farmer himself were also important and explained decisions: background, culture (sensitivity to environmental issues, level of understanding of the processes involved in water treatment), objectives (were environmental aspects a primary or secondary preoccupation for the farmer?), economic resources (influenced by profitability of the farm, income, savings, access to loans), technical resources (access to equipment and technical support, access to cheaper wetland plants). Finally, the socio-economic and political environment influenced farmers' decisions through: legislation (existence of clear norms and expectations from authorities), economic situation (e.g. milk or cereal prices), possibility of external financial support (Government grant, European subsidies).

In this study, the adoption of nearly all CFWs (except two) was due primarily to pressure from SEPA or other organisations involved in environmental protection (SNH, The Tweed Forum), willing to enforce the current legislation. Existing legislation and the risk of fines and prosecution (and impact on the image of the business) in the case of chronic pollution or accidental spillages appeared the strongest incentives. However, as mentioned by farmers, subjectivity in the assessment of farmyard contamination, the absence of clear standards for effluent discharged from CFWs, and the uncertainty regarding efficiency seemed to hinder CFW uptake. The lack of communication between regulators and farmers, and between farmers themselves was also reported as hindering CFW construction.

Additionally, possible savings in collection, storage and spreading costs appeared to be an incentive for dairy and mixed farms. Indeed, most farmers who have the obligation to manage farm dirty water and would need to increase storage capacity for this purpose, mentioned they would indeed build a CFW on their own, even without grant aid, if its cost is low (≤ £5000). Farmers' willingness to pay was influenced by the level of a possible fine but also biased by the information available to them at the time, indicating that £5000 systems can cope with farmyard runoff. At the time, the promotion of relatively cheap and easy to build CFWs, combined with a grant covering most or all construction costs were strong incentives for uptake.

Several factors seemed to hinder a more voluntary uptake and a higher willingness to pay for CFWs by farmers. Due to the uncertainties in efficiency and management (e.g. frequency of sediment removal), dairy and mixed farmers perceived CFWs as a more risky option than other alternatives such as storage and spreading. Two of the three arable farmers and one of the six farmers with sheep did not consider CFWs as useful on their farms, perceiving farmyard contamination as low because pollution control measures had already been implemented, e.g. bunded fuel tanks, roofed areas, use of contractors for pesticide handling, or biobeds to collect water from tank filling and washing.

The relatively large area of land needed and the costs for construction and steading modifications, the absence of clear regulations concerning CFWs (e.g. water quality required, licensing, monitoring) and changes in legislation and water quality requirements also limited the interest of farmers in using CFWs. Security and health hazards did not appear to be an issue for any of the farmers due to the absence of children, use of fencing to restrict the access, and no other uses being made of the water. However, the risk of bird flu transmission by waterfowl was sometimes mentioned by farmers with a poultry enterprise. Finally, the lack of communication between farmers who have or are interested in a CFW, and the absence of a strong network also impeded information and experience sharing and slowed down adoption of this BMP. After implementation, the lack of involvement and monitoring by authorities often led to the absence of control, management and corrective actions, and subsequently to the underperformance of the systems.

6.4.2 Reducing the costs and optimizing the benefits of CFWs

The direct and indirect financial costs or losses associated with the implementation of CFWs were strongly site-specific (see Table 6.2), mainly depending on location (distance from farmyard to CFW), on the design (e.g. area, number of basins), on the soil type and need for a liner, and on planting density and area. This is commonly mentioned worldwide, in both rural (Tanner and Kloosterman, 1997; Culleton *et al*., 2005) and urban settings (Knight Merz, 2000).

Design was determined by a compromise between the quantity of effluent to be treated (independent of effluent strength and water quality targets) and by a farmer's willingness to pay. However, other factors influenced the overall cost, such as the use of farmer's machinery and labour, availability of planting material or existing seed banks in the soil, need to modify the existing water collection or drainage system and steading configuration (e.g. if the gradient for gravity drainage is unsuitable), addition of structures for monitoring purposes (e.g. inspection chambers) and fencing. In France, self construction of treatment systems by farmers and use of on-farm equipment and resources has been shown to achieve a 20% to 30% savings in overall construction costs (Cemagref *et al*., 2007), but requires clear understanding of the system and appropriate skills. Supervision by professionals is therefore needed during construction.

Additionally, it is recommended in Scotland to build CFWs during late spring or summer (to benefit from relatively dry conditions to facilitate earthworks, and to allow vegetation to establish before winter), which is also a period of intense farming activity (e.g. silage making, crop harvesting). Consequently, construction could interfere with farmers' routine, and low labour availability might not allow farmers' involvement. Nevertheless, involving farmers during all stages, from planning to planting, seems crucial to develop a strong feeling of ownership and responsibility and subsequently foster a sustainable use and aftercare of CFWs, and efforts should be made to reconcile both farming and CFW construction activities.

A significant economy of scale may be achieved when building CFWs. Most of the costs are relatively independent of the surface area and increase only slowly with increasing wetland area, e.g. costs for design, survey, supervision, discharge licence, for moving machinery to the site, connecting farmyard to wetland and connecting cells, maintenance and monitoring. Only the costs related to earthworks and planting are nearly linearly increasing with increasing labour time. Therefore, increasing wetland size, when land is available, might be cost-effective, and usually provides additional benefits related to higher efficiency, cleaner water, higher amenity value and enhanced biodiversity (Bin and Polasky, 2005; Carty *et al*., 2008a).

Taking into account costs for building the wetland, fencing and loss of output, Cuttle *et al.* (2007) suggested annual costs for constructed wetlands (the design was not mentioned) of £3980 (arable farm), £1930 (dairy), £1280 (beef), £5780 (broilers) and £940 (indoor pigs). The cost for dairy farms seems low compared to the cost of a CFW sized according to current recommendations, i.e. a multi-cell system with a surface area twice the surface area of the farmyard. The high cost for arable farms is due to a larger loss of output, rather than to a difference in design.

6.4.3 Choosing between dirty water management options

Several well studied and tested alternatives are available to farmers worldwide to deal with dirty yard water, the most common and practical including conventional storage and spreading by slurry tanker or sprinklers, CFWs, ponds or lagoons, treatment by vertical or horizontal reedbeds followed by spreading over pastures or woody areas (USDA *et al.*, 1995b; Tanner and Kloosterman, 1997; Dunne *et al.*, 2005; CRAPL, 2007; Carty *et al.*, 2008b). In practice, the final decision on whether to implement a CFW should be guided by several factors:

1) The need for a CFW: if pollution from the farmyard is not an issue, then the construction might not be justified, except for amenity or biodiversity enhancement, or for irrigation (in this case, deeper lagoons would be more adequate than shallow wetlands). For example, the need will usually be higher for dairy farms with large areas of contaminated unroofed yard. However, judging the degree of risk associated with farmyard contamination is crucial (e.g. is the farmyard polluted, is it connected to a water body?), but to some extent subjective. Moreover, some BMPs might be implemented to further reduce yard contamination to an acceptable level (e.g. roofing), but may be very costly, and in this case, a CFW could be cost-effective.

2) Physical factors/constraints: land availability and suitability: surface area available, slope of the land, soil depth and permeability (to limit groundwater contamination), water table depth, distance from dwellings and wells. In many cases, when land availability is limited or soil is not suitable and would require the use of a synthetic liner, other less costly options are to be preferred.

3) The capacity of existing facilities and equipment for storage (e.g. tanks, lagoons) and spreading (e.g. tanker, sprinklers) and future plans for the farm. If the existing storage volume can cope with additional runoff, or if storage extension is planned and can benefit from a Government grant, a CFW may be less economically advantageous. Hence, the long-term farm plans have to be looked at carefully.

4) The cost of the CFW compared to the cost of alternative options, taking into account capital, running and opportunity costs, as well as grants available for construction or for other options.

5) The personal sensitivity regarding additional benefits brought by the CFW (e.g. landscape, biodiversity, amenity). If land is suitable and even if the cost of a CFW is higher than a conventional option, the CFW might still be preferred, due to the valuable secondary benefits it will bring to the farm and environment.

6.5 Conclusions

This study identified great differences in the costs involved in CFWs and the benefits they provide. For the CFWs investigated, costs were highly site-specific and design-specific and ranged from a few thousand pounds to c. £30 000. However, water monitoring so far in Scotland (presented in Chapter 4 and in Stewart (2008)) and farm visits revealed that some of the CFWs in place did not ensure sufficient and consistent treatment, due to their small size, additional inputs, absence of vegetation or adsorption media, mistakes in construction (e.g. wrong location for the pipes, inappropriate depth, field drains not blocked), and lack of maintenance. The environmental benefits of CFWs were mainly linked to their size, structure, inputs and vegetation and appeared to be reduced for small systems. Farmers' understanding of the systems limitations and their involvement in the maintenance were limited, which resulted in the misuse of the CFW, the absence of maintenance or corrective actions, and in their poor performance.

Experience from Ireland shows that relatively large and vegetated wetlands are required to ensure efficiency and robustness in dirty water treatment (Dunne *et al.*, 2005; Scholz *et al.*, 2007a; Carty *et al.*, 2008a; Mustafa *et al.*, 2009), which inevitably incurs higher costs and larger land uptake. Despite being more expensive, the wetland option has been shown to be economically interesting for many farmers, in comparison with other more conventional alternatives such as storage and land spreading, and also brings valuable additional benefits, mainly in terms of contingency, habitat, biodiversity and landscape enhancement as well as flood control.

From this study, the following recommendations are made to foster the adoption of CFWs and ensure their sustainable design and use:

1) Farmers expressed uncertainty regarding what CFWs should and can achieve, and therefore, clear information should be provided to them on water quality standards for wetland discharge (e.g. pollutant concentration thresholds, daily loads).

2) Adequate information and informed expert advice should be given on advantages, limitations, requirements in terms of land, costs and maintenance activities, and liabilities associated with their implementation, i.e. identification of the responsibilities for monitoring, or in the case of underperformance.

3) Financial support should exist for construction through a capital grant, but also for operating costs through annual payments (comparable to agri-environmental measures) to incentivize good maintenance and extended life-time. Support should primarily be granted in relation to water treatment, but could also be given to support biodiversity conservation, if the ecological value of the CFW is shown to be significant, which should be the case (mainly in the last cells) due to the large size recommended.

4) Adequate water quality monitoring should be conducted at the outlet of CFWs and should include low flow periods (when dilution is less and river sensitivity higher), rainy periods (when volume discharged will be larger), and could focus on one or two water quality parameters of high ecological impact and more difficult to remove from wastewater (e.g. TP, NH_4-N) to reduce time and costs. Farmers could be involved in water monitoring, ensuring visual inspections (e.g. turbidity, odours, sewage fungus) or helping with water sampling during rainy periods (a brief training would then be required).

5) Appropriate ecological monitoring of aquatic macroinvertebrates should be carried out at least annually in the final cell of the CFWs or downstream in the receiving waterbodies.

6) Improved communication, experience and resource sharing amongst farmers and between farmers and advisers and regulators should be encouraged, by the use of demonstration days, leaflets and meetings.

More research is needed to understand better the processes responsible for the interception or removal of nutrients and carbon, and the factors influencing those processes, such as hydrology, source strength, soil type, vegetation cover, etc. Assess the water treatment performance of CFWs over the long-term, their effective lifetime (is P retention the main limiting factor?), and their environmental impact, including emissions of greenhouses gases, infiltration to groundwater, sediment disposal or wildlife exposure to contaminants. Valuing these externalities together with the benefits might guide future policy decisions in this field. The current uncertainties related to efficiency and design seem to justify the need for precaution and the promotion of robust systems.

Chapter 7: Final Discussion

The main aims of this research were to investigate the water treatment efficiency of two Constructed Farm Wetlands, to assess their ecological value, and to evaluate the costs and benefits of CFWs from a larger sample of wetlands of different designs. This chapter discusses the implications of the main findings of this research and re-examines the hypotheses presented in Chapter 1. Based on the research it also makes recommendations for CFW design and maintenance, dirty water management and agri-environmental policy for reducing diffuse water pollution from agriculture.

Are CFWs efficient at treating dirty farmyard and field drainage?

Generally, ponds and surface flow constructed wetlands are considered or perceived as a rather effective, ecologically friendly and low-cost solution for dealing with both point and diffuse sources of pollution. However, wide variations in design, efficiencies (from < 0% reduction when pollutants are released, to nearly 100%), and costs (from a few thousand to > £40 000 ha^{-1}) have been reported, and constructed wetlands do not always allow specific water quality targets to be reached (Tanner and Sukias, 1997; Kadlec and Knight, 1996; Dunne *et al.*, 2005; Carty *et al.*, 2008b).

Data from the current study and from similar work in Scotland (David Kay and Tony Edwards, *pers. comm.*) seem to indicate that CFWs that are small in relation to the surface area they intercept (e.g. < 10 % of the interception area), do provide some degree of treatment, but are not effective enough at treating water, with concentrations of pollutants at the outlet of these systems often exceeding river water quality standards. Small CFWs do not allow sufficient residence time to treat excessive loads which can occur following accidental spillages or extremely heavy rainfall events, especially when field drainage is also collected. In addition, in this research, significant differences in treatment performance were observed between seasons at CFW1 and CFW2 for NO_3, NH_4 and RP, with efficiency at CFW2 of < 5% in autumn/winter for NO_3. Nevertheless, the influence of the temperature alone on performance, which has been documented previously (Kadlec and Knight, 1996;

Kadlec, 2003; Dunne *et al.*, 2005) could not be isolated, and efficiency reduction could be mostly due to subsequently larger volumes of field drainage (fields are frequently saturated in autumn/winter and inflows were at least 10% to 30% higher at CFW1 and CFW2 respectively) and shorter residence time. Although receiving watercourses are less sensitive to high nutrient loads in winter, low performance for four to six months of the year is a strong limitation. In contrast, larger Irish ICWs treat more concentrated dirty water (e.g. mean RP between 20 and 40 mg l^{-1}), usually without field drainage, and ensure treatment efficiencies above 90% for all parameters of concern. Concentrations decrease nearly exponentially from one wetland cell to the other, and at the outlet they are under or close to river water quality targets for BOD_5, faecal indicators, NO_3 and NH_4, but not for RP whose concentration is often above 0.1 mg l^{-1} (Scholz *et al.*, 2007a and 2007b; Bob Foy, *pers. comm.*).

Assessing efficiency

Ambiguity exists around the notion of "treatment efficiency", and difficulties arise when assessing it, and when comparing efficiencies between different CWs. Expressing efficiency by mass, i.e. reduction in pollutant loads, is more complex to measure and subject to greater uncertainties (since both flows and pollutant concentrations require assessment), but still more meaningful than efficiency by concentration, since it is based on the water balance and hydrological characteristics of the wetland. It also allows for comparisons between heterogeneous systems receiving different inputs. At CFW1 and CFW2, treatment efficiency estimated by mass was actually greater than by concentration (differences in concentration between inlet and outlet were relatively small). At CFW1 efficiencies by mass and concentration were 65% and 42% for NH_4, 80% and 68% for NO_3 and 45% and 12% for RP, respectively, and even greater differences were found at CFW2, where flow attenuation was greater due to larger size and losses, with treatment efficiencies by mass and concentration of 48% and 34% for NH_4, 45% and 26% for NO_3 and 45% and 31% for RP respectively.

Assessing efficiency is challenging. Firstly, limitations exist in terms of the collection and analysis of water samples and the timing of sampling. Sampling intensively and especially at high flow or in winter is essential to assess pollutant loadings when peak pollution is expected, but is not always practical and is often costly. Secondly, assessing accurately the water balance of natural and open systems is challenging, due to inputs and outputs being irregular and diffuse (e.g. several inlet pipes, lateral flows).

In CFWs without artificial liners, some outputs, for example infiltration volume and composition, are often unknown. Finally, as wetlands age, vegetation cover, hydrological patterns and bacterial populations change drastically, soil P sorption sites may become saturated, organic matter and sediment accumulate and can release pollutants, which explains why efficiency might decrease in the long-term (Kadlec and Knight, 1996; Reed *et al.*, 1995). This effect could not be investigated, due to the limited duration of the study and young age of the CFWs, but evaluating the sorption capacity of sediment for P could help inform sediment management strategies.

In addition, high water treatment efficiencies do not always imply compliance with water quality concentration targets, when those exist, and the notion of efficiency might therefore not be useful for informing policy decisions. Even relatively small concentrations at the outlet with large or constant outflows can also have significant impacts on receiving waterbodies, depending on their assimilative capacity and sensitivity to pollution. For example, CFW1, the smaller, cheaper and most contaminated CFW had a high treatment efficiency for BOD_5 and SS and medium to high efficiency for NH_4, NO_3 and RP but released continuously an effluent whose concentration was frequently above acceptable targets (e.g. 80% of samples contained > 2 mg l^{-1} NH_4, 90% > 1 mg l^{-1} RP, 40% > 20 mg l^{-1} BOD_5). CFW2 also exhibited medium to high efficiencies, although these efficiencies might not very meaningful, due to the very low concentrations at inlet, inaccuracies in the water balance and unaccounted losses through infiltration. Its effluent complied with standards nearly all the time for all parameters except faecal indicators.

Setting targets for constructed wetland discharge

The absence of targets for CFW effluent has allowed construction of underperforming systems, and a real effort is needed to clarify this aspect. Two options could help when setting targets for CFW effluent.

The target could be a single concentration or loading threshold at a national level, for one or several water quality parameters, preferably those of greater ecological impact or most difficult to remove (e.g. TP, NH_4 or NO_3) or those of public health significance (e.g. faecal pathogens). The threshold could be set by consensus according to what is acceptable or feasible or could be based on river water quality standards and requirements from the WFD, for which a new river classification scheme is being designed. CFW performance could then be assessed using a probabilistic approach introducing a notion of "risk", i.e. the frequency of exceedance of a given water quality threshold at the outlet, which should be assessed during rainy periods. Design could then be driven by an acceptable frequency of exceedance.

The target could also be a site-specific concentration or loading which would depend on the assimilative capacity of the receiving water body (if effluent is discharged into surface water) or on the sensitivity of the catchment. This second approach, the "Total Maximum Daily Load" (TMDL) concept, is being used in the USA, regulated by the Federal Water Pollution Control Act ("Clean Water Act") (US EPA, 2002), which requires States to identify "impaired" water bodies, i.e. those who do not meet defined water quality standards or specific criteria. The TMDL approach specifies the maximum amount of a pollutant that water bodies and downstream waters can receive and tolerate while complying with water quality standards. This implies identifying and quantifying the sources of pollutions, associated fluxes, and assessing potential mitigation measures (Kay *et al.*, 2008). On the one hand, site-specific targets would give more flexibility for the design and would reduce the cost of those wetlands discharging into less-sensitive waterbodies and catchments (they could be built smaller), probably encouraging their construction. However, this could be a very costly approach (to identify point and diffuse sources, catchment sensitivity and monitor TMDLs), could result in large quantities of pollutants being released altering the water environment dramatically if priorities change over time, and large differences in design and costs could be perceived by farmers as being unfair.

In Scotland, no target concentrations or loadings have been set until now, and the choice was recently made to recommend a robust design, i.e. large, multi-cell systems which are expected to perform a high level of treatment and to release effluent with acceptable pollutant concentrations under most conditions (very low exceedance frequency), and will often not produce any discharge in summer (Carty *et al.*, 2008b), when assimilative capacity of receiving waterbodies is lower.

Are CFWs ecologically valuable?

Research has shown that CWs with better water quality, more complex habitat structure and substrate, and surrounded by uncontaminated ponds do have the potential to contribute to biodiversity, habitat and landscape enhancement, although their ecological value is usually constrained compared to unpolluted wetlands (Batty *et al.*, 2005; Culhane, 2007; Alsfeld *et al.*, 2008).

From the two CFWs studied, the largest, cleaner, multi-cell wetland (CFW2) was indeed the richest in terms of both plants (22 wetland plant species) and macroinvertebrates (46 BMWP scoring species), due to low concentrations of BOD_5 and NH_4, well established vegetation, a mixture of shallow and deeper zones, fast flowing and slow flowing water, and permanent groundwater inflow. Biodiversity in CFW2 could increase over time, due to water quality being relatively stable, but could also be affected negatively by changes in vegetation cover and sediment accumulation. CFW2 represented an asset in the landscape and a recreation area for walkers and a potential educational site. In contrast, after two years, aquatic life at CFW1 was still much less diverse and abundant (14 wetland plant species, most of them represented by a few sparse specimens and 24 macroinvertebrate scoring species), and macroinvertebrate mortality rates seemed high, due to the high contamination of the inflow, low dissolved oxygen, simpler structure, lack of organic substrate and less diverse vegetation. The amenity value of the pond was also reduced since it is small and vegetation not well established. Biodiversity is expected to remain relatively low, due to pollutant concentrations being high most of the time. No rare species were found in any of the CFWs studied and macroinvertebrate and plant diversity were lower than in the nearby non-polluted pond investigated suggesting that the ecological potential in these CFWs is strongly constrained.

CFWs can be valuable only if built large enough, and most importantly, if composed of several cells allowing gradual water quality improvement the further from the inlet. In the current study, no significant influence of the distance from CFW inlet to sampling point was found on plant or macroinvertebrate diversity and abundance. Nevertheless, at CFW1, mayflies and beetles were most often captured closer to the outlet, amongst the vegetation, and at CFW2, the number of scoring species and BMWP score was slightly higher in P5 (33 species, BMWP score of 93) than in the other ponds (28 scoring species in P1, 31 in P2, 31 in P3 and 28 in P4). However, relationships between distance from the inlet on plant and animal richness could be obscured by changes in the size, depth, substrate, sedimentation and flow patterns (e.g. fast flowing water over gravelly substrate favoured stoneflies). The edges and non submerged areas of both CFWs also appeared to be a valuable habitat for adult insects, amphibians, reptiles, mammals and birds, which suggests the importance of an extensively managed buffer area between water and grazing or arable land, where mowing should be limited to the minimum required for easy access.

The influence of pond size on biodiversity has been investigated and it was found that small ponds can be more diverse than larger ones, but large ponds can host species absent from smaller waterbodies (Oertli *et al.*, 2002), suggesting the importance of combining ponds of different sizes. Interestingly, conflicts sometimes arise between water treatment and biodiversity conservation objectives. Waterfowl populations can contribute significant inputs of faecal matter which can negatively affect wetland effluent quality, if inputs occur close to the outlet, as observed at CFW2, which suggests that open water areas at the end of CFWs should be avoided.

Are CFWs a costly option?

The capital costs of constructed farm wetlands are variable, ranging from a few thousand pounds for small but less efficient systems, to more than £20 000 for larger more efficient ones such as ICWs or CFWs being recommended currently in Scotland and Northern Ireland. The costs depend mainly on the size (the larger, the more expensive, but economies of scales exist), distance from the farm, earthworks, planting density, fencing requirements, type and value of the land used for construction, use of artificial liners (which is currently economically prohibitive), and self-construction by farmers, which can reduce the cost by 30% (CRAPL, 2007). The implementation of CFWs also requires sometimes costly modifications of the steading, roofs (guttering) or drainage system. Moreover, operating costs linked to regular checking, vegetation maintenance, sediment removal and water monitoring, might be significant, although they are often not properly acknowledged due to lack of experience or information (Lampe *et al.*, 2005). In fact, most CFWs visited during this research have been more or less unmanaged since their construction and no corrective actions were carried out to improve their efficiency.

In spite of the relatively high costs attached to their construction and maintenance, CFWs might be a cost-effective and an attractive option for farmers, if their cost is kept relatively low or if sufficient financial support is provided for their construction and maintenance. CFWs first reduce the risk of fine and prosecution for pollution and help improve the image of the business. Compared to other alternatives, they may also represent a saving in money and time, by reducing the need for collection, storage and spreading of diluted dirty water, especially when slurry and dirty water are stored in the same tanks. Allowing higher strength effluents (e.g. parlour washings or silage effluent) to be treated in CFWs built according to current recommendations (Carty *et al.*, 2008b) might be a way to increase further their cost-effectiveness. Indeed, wetland final effluent quality should not be affected since treatment efficiency is actually more constrained by hydrology (volumes and residence time) than by wastewater quality, as found in ICWs.

Additionally, from a wider societal perspective, CFWs reduce diffuse and point source pollution and hence limit the costs associated with ecosystem degradation, water treatment, loss of amenity value of rivers and beaches, allowing substantial direct and indirect savings. The range of benefits obtained from reducing water pollution at source hence justifies grants and subsidies given to farmers.

How should CFWs be designed and maintained?

Official recommendations now exist in Scotland (SEPA) and Northern Ireland (NIEA) for the design and maintenance of CFWs to treat farmyard or roof runoff (Carty *et al.*, 2008b), based on the experience from Ireland (Dunne *et al.*, 2005; Mustafa *et al.*, 2009) where the monitoring of ICWs has been ongoing for more than 10 years. In order to achieve high pollutant removal (≥ 90%) and relatively low outlet concentrations, e.g. < 1 mg l^{-1} RP, < 10 mg l^{-1} BOD_5, the use of large (ratio wetland surface area: farmyard surface area of 2:1, multi-cell (at least four cells), vegetated (full cover, at least in the initial cells), and shallow wetlands, is advised. Although experience regarding vegetation maintenance and sediment removal is limited, desludging is advised every 5 to 10 years (Scholz *et al*., 2007b; Carty *et al.*, 2008b). Vegetation harvesting is not recommended in CFWs, due to its cost and small impact on treatment efficiency. However, Cemagref *et al. (*2004) suggests that reeds should be harvested and possibly composted once a year in autumn in vertical constructed wetlands to limit the release of pollutants during vegetation die-off.

While in wildlife ponds natural colonization is often preferred, in CFWs vegetation (e.g. *P. australis*, *T. latifolia*, *Carex* spp.) has to be planted initially after the topsoil has been replaced, homogeneously and dense enough (80% to 90% survival rate is expected) across the flow, to reduce velocity and enhance sedimentation and contact between wastewater and biofilms. Natural colonization is not sufficient since it occurs preferentially in areas of low depth, low flow and low pollution. Similarly, planting only the edges of a pond, as widely practised, might result in "short-circuiting" of the flow between inlet and outlet, smaller residence time and lower treatment efficiency.

A robust design for CFWs requires large areas of land to be taken out of production, and is more costly overall, even if monitoring costs for farmers and local authorities will be reduced in the long-term due to smaller uncertainties in performance and variations in effluent quality. Such a design may therefore have both positive and negative impacts.

Indeed, large, multi-cell wetlands will ensure efficient water treatment in the short and long-term, will cope with high volumes during heavy rainfall, with unexpected inputs (e.g. spillages) and with changes in precipitation patterns linked to climate change (increase in intensity, frequency and duration of storm events) (IPCC, 2007). A robust design will also benefit biodiversity and reduce the costs associated with water monitoring. In addition, official guidelines emphasizing and recognizing the need for larger, more efficient and better integrated systems might trigger political and financial support (a grant for construction was indeed introduced in 2009 by the Scottish Government) and should lead to a significant improvement in water quality and ecology in the short-term, as illustrated by the success of restoration schemes in the Anne Valley in Ireland (Carroll *et al.*, 2005). Moreover, recommending a large surface area in relation to the impermeable area might encourage farmers to collect and divert roof runoff (when they do not already do it and when runoff is not contaminated) to decrease the land area needed.

However, such a design is not always feasible, practical and cost-effective. In fact, in many cases, suitable land area and soils might not exist on the farm. In addition, farmers may not be willing to give up land due to its scarcity, high grazing or agronomic value, or future intended use. Although support is now provided, it is only partial (40-50% of capital costs in Scotland), farmers might not be able to afford the "one-off" construction costs and subsequent running costs, which are often overlooked and not supported by grants, depending on the profitability of their farm, their income, savings or access to loans. Consequently, the majority of farmers could refuse the constructed wetland option and choose a more conventional but potentially more costly solution such as storing and spreading, with which they are already familiar and that they perceive as a less risky option. This could put more pressure on storage capacity and might lead to diffuse pollution being exacerbated during wet periods. In addition to financial support, a large effort of communication and transparency is needed between authorities, farm advisers and farmers, to improve knowledge of the potentials, limitations and requirements of CFWs and to incentivize farmers to build their own CFWs. Communication should be based on group meetings, demonstration projects involving farmers during planning, construction and maintenance, and regular updates on research findings should be provided to advisers and farmers to allow for possible corrective actions or innovations to take place.

Are there sustainable alternatives to constructed farm wetlands?

Many options exist for the management of agricultural wastewater, but some are heavily engineered or rely heavily on energy and external inputs and are therefore very costly and more complex to manage (Kadlec and Knight, 1996; Vymazal, 2009).

The most common sustainable options include storage and irrigation (when conditions are suitable). Slurry and yard water may be stored together or separately in above-ground stores (e.g. made of steel, at a cost of £30 to £40 m^{-3}) or below-ground stores (e.g. concrete). Earth-banked tanks may be suitable (Scully *et al.*, 2004) and are usually cheaper (i.e. £10-£25 m^{-3} or up to £50 m^{-3} if a liner is used) (DARD, 2005 a, b), but law requires them to be impermeable, i.e. either lined by artificial liner (Scotland) or built on a soil with less than 10^{-9} m s^{-1} hydraulic conductivity. Storage and treatment can be ensured by lagoons (with or without aeration), surface flow wetlands with or without stabilization ponds as a pretreatment step, and subsurface horizontal or vertical wetlands (with substrates such as sand, clay, limestone enhancing P adsorption). All these options clearly have limitations, and the farmer's choice will be guided by the type and quantity of wastewater to be treated, by land and capital availability and by the local political and regulatory framework.

A CFW should be considered as an “end of treatment-train measures”, and hence, before its construction, efforts should be made to control pollution source on the farmyard using appropriate BMPs, e.g. ensuring regular scraping, separating roof and yard water by guttering and roofing concrete areas, to decrease the volumes of yard runoff produced. Moreover, in a rural context, when considering small to medium farms, sustainable (effective and acceptable by farmers) water treatment options should be robust, but simple, low-technology, easy to manage, should not rely excessively on external inputs or energy, and should be affordable (e.g. financial support should be provided).

Surface flow wetlands are relatively easy to build but require large areas of land and sometimes liners, while subsurface flow wetlands require less space, but are more engineered constructions and do not provide ecological benefits. Interestingly, in France treatment systems involving lagoons with macrophytes have not been recommended since 1997, due to high land requirements and problems raised by vegetation and sediment management. Instead, several options have been officially validated to manage farmyard effluent and dairy parlour washings, which focus mostly on BOD_5, NO_3 and NH_4 removal (aiming for > 80% mass removal), with P removal being a lower priority. They include non vegetated lagoons, multi-stage vertical reedbeds with recycling of the water over grassland or through vegetated areas (reeds, eucalyptus, orchards), and irrigation over grassland (Cemagref *et al.*, 2004; CRAPL, 2007). Discharge from these systems directly into waterbodies is forbidden, and final effluents are always polished by runoff and infiltration over vegetated land, which improves P removal. Figure 7.1 summarizes the key questions and steps guiding the choice of a farmyard runoff management option.

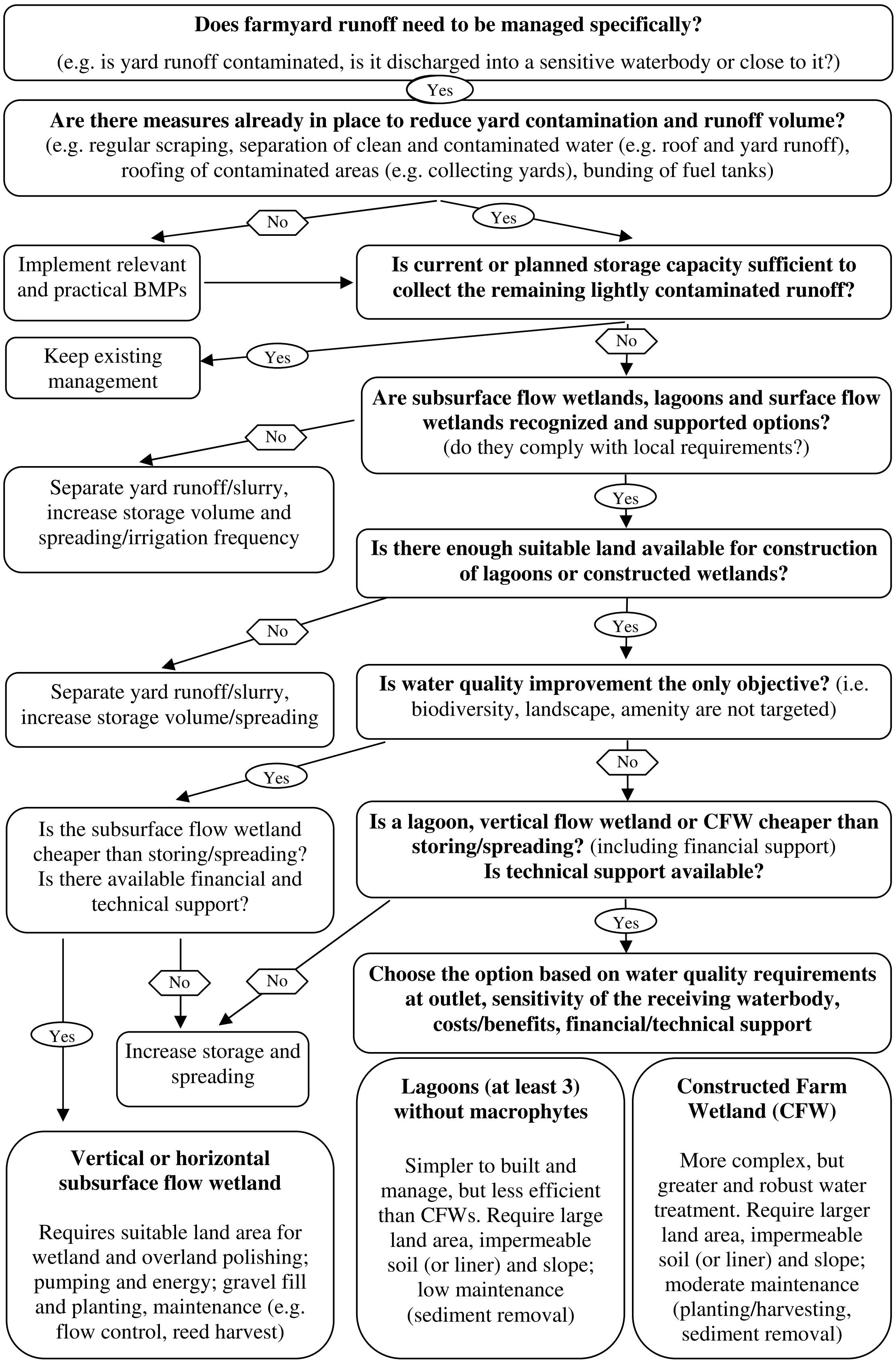


Figure 7.1 Key steps guiding the choice of a farmyard runoff management option.

Chapter 8: Conclusions and Further Work

This research has shown that the two small constructed farm wetlands investigated in the Scottish Borders contributed to some extent to water quality improvement and to flow attenuation, provided some ecological benefits and represented a low initial cost for farmers, compared to the cost for building new storage facilities for dirty water. However, even though these low-cost CFWs are probably better than nothing at all, they clearly did not allow satisfactory nutrient and pathogen removal, and were not robust enough to cope with large volumes generated during rainy periods. Mistakes during construction (e.g. wrong pipe location, excessive pond depth), operation (e.g. additional inputs) and management (e.g. absence of pipe clearing caused clogging in P4 at CFW2 and contributed to flooding of the adjacent track), often aggravated their poor efficiency.

For these reasons, and based on long-term monitoring of integrated constructed wetlands in Ireland, the use of larger, multi-cell wetlands is now recommended in Scotland and Northern Ireland for treatment of lightly contaminated farmyard runoff (Carty *et al.*, 2008b). By integrating water quality improvement with other aspects such as biodiversity and landscape enhancement, CFWs may be the best way forward to ensure compliance with European Directives and achieve substantial environmental improvements.

Field investigation of constructed wetlands is ongoing worldwide, but studies often provide short-term performance data on young or non typical constructed wetlands, and performance is subject to large uncertainties linked to uncertainties in water balance assessments, and variations due to differences in design, climate and loadings. Consequently, further research is needed on older farm wetlands to assess accurately runoff (volume and quality) generated on a variety of farms, volumes discharged by constructed wetlands, and subsequently, to estimate the long-term treatment performance of CFWs (and their longevity) for nutrients but also pathogens, which are of great health concern and often neglected (Edwards *et al.*, 2008). Low-cost monitoring strategies have to be developed, focusing for example on one or two water quality parameters and on biological indicators, which better reflect long-term changes in the water environment. Particular efforts are needed to evaluate P removal performance, since it is a key nutrient triggering eutrophication and the most difficult to remove in the long-term (DeBusk et *al.*, 2005), and to find sustainable options and incentives for its sequestration and reuse within the farm itself. To allow for more accurate assessments of their design and performance, CFWs have to be built anticipating research needs, in order to allow for set up of adequate instrumentation (flow monitoring devices and refrigerated automatic samplers) and easy access.

Estimating infiltration rate through the wetland base and quality of the seepage over the long-term appears crucial to assess the risk of groundwater contamination, and to inform on the use of artificial liners. Where soils are too permeable, to avoid leakage of very labile pollutants (e.g. NO_3), liners are compulsory under lagoons in France and New Zealand. Indeed, while the wetland base may seal itself by accumulation of sediment, vegetation roots or drying of certain areas in summer (crack formation) could also increase infiltration. Research on this aspect is being carried out in Ireland, by sampling groundwater in the vicinity of CFWs or by using lysimeters placed under the wetland base to measure infiltration rates and collect infiltrating water for analysis (Rory Harrington, *pers. comm.*).

Investigation of greenhouse gas emissions such as CH_4 and N_2O by CFWs is also important to assess the extent of the risk of pollution swapping, to evaluate the cost of these externalities and to find suitable mitigation options (e.g. addition of inhibitors in the wetlands, management of water level).

In addition, models of constructed wetlands and of pollutants fluxes at catchment scale, based on field and long-term studies, could enable CFW site-specific design and long-term performance predictions, e.g. due to climate change or changes in farm size, and could help compare the efficiency and cost-effectiveness of different measures and subsequently inform political decisions (Arheimer and Wittgren, 2002; Erik *et al.*, 2002; McGechan *et al.*, 2008).

Finally, international experience suggests that design should not be “frozen” by current national guidelines, and space should be given for experimentation on different systems, including horizontal and vertical flow wetlands, using available materials (e.g. ochre for P removal), and configurations (e.g. combination of shallow and deep water, vegetated and non vegetated ponds, variable inlet and outlet structures to manipulate water level and flows). Indeed, a design reducing the area needed for water treatment would undeniably allow broader adoption of CFWs. However, in their quest for the “optimal design”, researchers and designers should keep in mind the necessity for rural constructed wetlands to be relatively easy to build and manage, and to utilise local materials, plants and renewable energy as much as possible. The cost of CFWs should be kept relatively low and acceptable to farmers, and their further promotion will require appropriate financial support for capital as well as operating costs (e.g. in the form of annual payments to ensure a long-term incentive for maintenance), and detailed information on their potentials and limitations. Water quality targets, as well as responsibilities regarding maintenance, water monitoring, CFW achievement or failure, also have to be clarified rapidly, to decrease uncertainty and foster their adoption.

Finally, the integration of agri-environmental policy and water policy and the improved communication and co-operation between researchers, local actors (private and public) and policymakers are required to address diffuse agricultural water pollution in an efficient, cost-effective, holistic and sustainable way.

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Appendix A: Mesohabitats surveyed for aquatic macroinvertebrates

In the amenity pond (AP), seven mesohabitats were distinguished for the purpose of the macroinvertebrate surveys (26 s in each): 1) Stand of *Nymphaea* sp., 2) Stand of *T. latifolia*, 3) Stand of *P. australis*, 4) Stand of *A. plantago aquatica*, 5) Area with boulders, 6) Shallow edges with *M. fontana* and gravels, 7) Deeper water with *L. major* and *M. spicatum.*

At K, due to the low structural complexity, four mesohabitats were sampled (45 s in each): 1) Area close to inlet between edge and earth bar, supposedly more contaminated and with a faster accumulation of organic matter and sediment; 2) *T. latifolia* stand in the eastern part of the pond; 3) Shallow edges with dominance of *Agrostis stolonifera* and *J. effusus*; 4) Deeper areas closer to the centre of the pond.

For P1 to P5, the following mesohabitats were surveyed: P1: 1) Shallow area close to inlet colonized dominantly by *N. officinale*; 2) Stand of *P. australis* and *J. effusus* on the eastern side; 3) Stand of *G. fluitans*; 4) Deeper area in the south-west corner. P2: 1) Shallow edges colonized by *P. australis* and *J. effusus*; 2) Deeper open water in the centre of the pond with *C. brutia*, S*pirodella* sp. and *L. minor*; 3) Area close to the outlet with *G. fluitans.* P3: 1) Shallow edges colonized by *P. australis* and *J. effusus*; 2) Deeper open water in the centre of the pond with *C. brutia*, S*pirodella* sp. and *L. minor*; 3) Shallower area close to the outlet with *G. fluitans.* P4: 1) Shallow edges colonized by *P. australis* and *J. effusus*; 2) Deeper open water in the centre of the pond with *C. brutia*, *R. omiophyllus*, S*pirodella* sp. and *L. minor*). P5: 1) Shallow edges colonized by *P. australis* and *J. effusus*; 2) Open water close to the inlet of the pond with *P. crispus*; 2) Open water area with dominance of submerged *C. brutia* and *M. spicatum,* and floating *L. minor*; 3) Stand of *B. umbellatus* and *S. erectum,* 8 m from the outlet; 4) Shallow edges (northern area and southern area) with *G. fluitans* and *A. stolonifera.*

Appendix B: Questionnaire to farmers with a constructed farm wetland

(Anonymous; Not a judgement of the farm; Not a control; No report given to SEPA on compliance or pollution)

Date: Place of interview: Time:

1) Personal information

Name: Age: (26-35, 36-45, > 45) Address:

Tel: Fax: Email:

Main Activities (farming and others):

Farm ownership: tenant or own land? Staff employed on farm?

2) Description of the farm (Ask for plan of farm and work on it; See categories in agricultural and horticultural census)

Total Area of the farm?

Land cover / Land use:

Permanent grassland (area, grass species)? Rotational grassland (area, grass species)?

Arable (area, crops)? Fallow (area)? Woodland (area)? Others?

Livestock:

Dairy cows (numbers, breeds, milking time, volume of milk)?

Suckler cows (number, breeds)?

Pigs (number, breeds)?

Sheep (number, breeds)? Poultry (number, breeds)? Other livestock?

Destination of products:

Milk: Meat: Crops: Others:

Changes in numbers of animals over time (cattle, pigs, etc.):

During the year (seasonally)? Over the years?

What are the main factors driving changes in animal numbers in your farm?

Future Prospects (extension, increase in cattle numbers, etc.)?

3) Description of farming practices

Animal Management:

Are animals kept inside/outside? For what period of the year?

Are animals housed on straw or other materials (e.g. woodchip corrals)?

When are cows / pigs / sheep present on the steading (morning, afternoon? Every day?)

How long do they stay on the steading (are there differences between summer and winter)?

Is part of the steading washed manually? How often (time spent/volume of water used)? Where is water going?

When/How often do you wash the parlour (time spent/volume of water)? Where is water going?
When is silage produced? Composition?	Where and how is it stored (roofed, open)?
Route from silage pit to cows?
Roofing of other areas on steading? If roofing, where does roof water drain to?
Sheep dipping (where, when, products used, fate of the water)? Use of antibiotics (amount, frequency)?

Waste Management:
Waste Management Plan?
Slurry volume annually? Slurry tanks (numbers, volume)? Slurry disposal (when, where, difficulties)?

Pesticides / Fertilisers / fuel:
Pesticide sprayers? Where are they filled?
Where is fuel stored and tanks filled? (Any bunding around these areas to contain spillages?)
How often are the tanks filled? (more often in winter? Summer?)
Where is the workshop in the farm?
Where are fertilisers stored and filled in spreaders (any losses on the steading)?
When are fertilisers applied?
Any preparation/processing of agricultural products on farm (e.g. vegetable washing, ice-cream manufacture, etc.)?

General Questions:
Do you keep record of animal numbers over the year (is it possible to access the data)?
In addition to pond/wetland, any other special measure implemented (e.g. Buffer strip, waste management plan, winter cover, animal diet plan, etc.)?

4) Origin of wastewater entering the wetland (steading, silage pit, sprayer filling area, roof, dipping area, tracks, overall catchment, septic tank overflow)

Where is the wastewater entering the wetland coming from (steading, roof, field, tracks, workshop, sprayers, etc.)?
Where is water collected and which route is it taking (field drains, pipes, etc.)? (Use map of the site/farm to locate pipes, manholes, etc.)
Do you think there is a great variability in the composition of this wastewater through the year (when is it expected to be the most polluted)?
Do you thing that you still have the scope to alter the composition of the wastewater going into the pond/wetland (improving cattle diet, covering silage, etc.)?

5) Questions concerning the constructed wetland

When was it built? Why (own willingness/pressure from SEPA/specific pollution issue)?

Who designed it? Who constructed it?

How long did it take between the "idea" and the end of the construction?

Do you have an idea of the area of land used for the whole system?

Was this area productive (grassland/arable, etc.)?

Does it affect the feeding of livestock? (decreases fodder availability?)

Do you think you are losing income because of the construction of the pond? How much?

Do you lose Single Farm Payment?

Could you have used more or less land to build it?

How much did it cost?

Did you receive financial help (Who?, How much?)?

How much did you personally invest?

What is the maximum amount you would have been willing to pay yourself for the pond/wetland (what influences this amount: avoid fines/sanctions, reduce need for wastewater storage)?

Was the pond/wetland planted (when, which plants)?

Was it lined with clay or plastic liner? (Is it over an aquifer? Are there wells or springs close by?)

Where is "clean" water discharged (ditch, stream)?

Any maintenance activities on the pond since construction? Any planned maintenance activities?

Any problems (rapid sedimentation, odours, algal blooms, etc.)?

Do you use or would you use the CFW for other purposes (recreation/fishing/watering cattle/irrigation)? If not, why (uncertainty about water quality/health hazards)?

What is your opinion of the use of CFWs to treat runoff from steading (Do you think this measure is efficient, Cost-effective, Manageable)?

Does the pond reduce the need for slurry storage? Does it make a difference in volume to be stored, spread and on the cost of these activities? (new tank, spreading).

What are the main obstacles to the spread of ponds/wetlands to treat runoff (lack of information on performance, high cost, uncertainty on efficiency, area of land required, safety issues, etc.)?

How can these obstacles be overcome? What can be done to spread the use of CFWs amongst farmers?

Who should help in the implementation of CFWS?

Do you know other farmers who have implemented such a system (name, address)?

What do they think about it (efficient, difficult to manage, etc.)?

Would you like to know later on about the outcomes of my study?

Which kind of information do you think would be most useful for you (e.g. maintenance, problem solving)? In which form (summary, personal contact, group meeting)?

Any comments, questions, problems you would like to mention?

Appendix C: Questionnaire to experts (CFW designers or farm advisers)

1) How is farmyard runoff managed in Scotland? (Is it stored and spread or diverted to drains/ swales/ditches/rivers?
2) Is roof runoff separated from yard runoff? Is it stored or diverted to swales, ditches, rivers?
3) Are parlour washings stored in slurry tanks or considered as farmyard runoff?
4) What is the proportion of farmers actually storing farmyard runoff? Are farmers prosecuted when they don't store/spread yard runoff? How high are the fines?
5) What are the most common practices for storage and spreading of farmyard runoff: underground or aboveground stores, tanker or sprinkler irrigation? Is dirty water stored in the same tanks where slurry is stored? Is it spread all year round? (Including in Nitrate Vulnerable Zones?)
6) Would you be able to share with me the costs associated with these management practices? (costs of the different options for dirty water storage and spreading: cost of a dirty water tank, slurry tank, tanker, sprinkler, labour, etc.)
7) If farmyard runoff was not stored and spread and diverted to a constructed wetland instead, would it bring farmers benefits? Would a constructed wetland make a significant difference in terms of volume of dirty water or slurry to be stored, spread and on the cost of these activities?
8) If parlour washings were not stored and spread but diverted into a constructed wetland instead, would it bring farmers benefits? Would it make a significant difference in terms of volume to be stored, spread and on the cost of these activities?
9) Do farmers rely on farmyard runoff as fertiliser or is farmyard runoff a rather useless and costly element to manage?
10) What is the maximum amount farmers would be willing to pay for a constructed wetland (what would influence this amount: e.g. avoid fines/sanctions, reduce need for wastewater storage)?
11) At which cost does a CFW become interesting for farmers in comparison with other options?
12) What is your opinion on the use of constructed wetlands to deal with farmyard runoff Do you think this measure is efficient, cost-effective, easy to manage?
13) If a farmer chooses to build a large CW, e.g. 1 or 2 ha, does he lose single farm payments?
14) What are the main obstacles to the adoption of constructed wetlands (lack of information on performance, high cost, uncertainty on efficiency, area of land required, safety issues, etc.)?
15) How can these obstacles be overcome?
16) Who should help in the implementation of constructed wetlands?
17) Do you know farmers who have implemented CFWs (How many? In which type of farms?)
18) What do they think about CFWs (efficient, difficult to manage)? Are they satisfied with them?
19) Are regulatory bodies (e.g. SEPA) monitoring these CFW and are they satisfied with their performance?
20) Update on Land Management Contracts, Scotland Rural Development Programme (potential sources of funding for CFWs? Possibility to combine funds? (FWAG, Government grants)
21) Any concerns, questions, suggestions?