# A REVIEW OF ‘SOFT ENGINEERING’ TECHNIQUES FOR ON-FARM BIOREMEDIATION OF DIFFUSE AND POINT SOURCES OF POLLUTION

By


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## Executive Summary

Improvements in water quality over recent years has been due to the reduction in point source pollution, i.e. pollution from an identifiable source. Diffuse pollution from agriculture has now been recognised as the biggest obstacle to the objectives of the Water Framework Directive. Whereas many of the mitigation measures for controlling diffuse pollution focus on crop, fertiliser, manure or animal management, there is scope to supplement these with soft engineering solutions. These include buffer strips and (natural or constructed) wetlands. The aim of this study was therefore to determine whether 'soft engineered' based systems could be established within agricultural surface water catchments to ameliorate water quality, with particular respect to agricultural diffuse pollutants and whether such approaches are likely to be cost effective.

There is a range of different potential pollutants on farms that can be lost to water:

- Concentrated point sources – potent (slurry, silage effluent), and their safe *storage* is controlled by other regulations (e.g. SSAFO regulations). This does not cover their *use/disposal*, where land application is the preferred option.
- Diffuse sources of nutrients, sediment, agrochemicals and pathogens – by their very nature, diffuse sources are difficult to isolate and treat.
- Dilute point sources (e.g. dirty water from farmyards, parlour washings, etc.) – need to be contained. Usually disposed of/used by land application. Pesticide washings from pesticide handling operations on hard surfaces could also be considered a point source.

*Natural wetlands* - Shallow, permanently flooded or wet marshy ground populated with macroyphytic vascular plants (i.e. reeds) are known to trap and hold large amounts of solids, particulates and dissolved constituents of waters that pass through them. Wetlands as a functioning biological system can clean water by a mix of physical and biological mechanisms, which include: plant uptake, adsorption, sediment deposition/retention, microbial degradation, chemical precipitation, natural die-off or predation (pathogens) and gaseous losses. Natural wetlands take many forms – reedbeds, grazing marshes, fens and lowland raised bogs, for example. Although all systems may regulate water quality to some degree, it is the reedbed systems that are seen as having the greatest potential to act as a treatment system. However – and this is important – the primary function of wetland systems in the UK and Europe is now primarily provision of biodiversity. Most wetlands are under conservation designation, which may compromise their function to clean water. In fact, many would benefit from clean water entering the system, because pollutants can compromise the biodiversity of the ecosystem.

Natural wetlands play an important role in the landscape. They are a valued asset, as is noted by the fact that most are designated as protected areas. They are valued and managed mainly for biodiversity benefits. Their role in water management is more likely to be that of regulating flow rather than water quality protection. We noted the suggestion that using for water treatment may conflict with management for biodiversity.

*Constructed wetlands* - man-made wetlands designed to mimic the action of natural wetland systems. Designs of constructed wetlands can vary but basically involve water being channelled into a series of man-made ponds with an impermeable synthetic liner or clay base, filled with either the original soil from the site or with selected substrates (normally sands and gravels) and aquatic plants.

Constructed wetlands are now a very widespread and quite well understood form of soft engineering for pollution mitigation. Most, if not all, water utilities have examples of constructed wetlands in operation to mitigate point-source pollution from a variety of domestic and industrial origins. They are inappropriate for some agricultural effluents – and are better with dilute sources, such as dirty water/parlour washings. There has been very little work, as far as we could see, looking at constructed wetlands to deal with pesticide

washings. Although originally designed to deal with point sources, there is scope for their use to treat diffuse sources of pollution, and most experience of this is in the USA. Construction costs and design to intercept diffuse sources are both issues that need to be discussed, however. As with any mitigation method, no system is foolproof, and the level of risk that can be tolerated will determine if constructed wetlands (or buffer strips) are deemed an acceptable approach to pollution control. There is little information on their longevity, but 15-20 years is suggested. Much will depend on the levels of inputs. Removing enriched sediment at the end of their useful life poses an environmental risk, either by contamination of the receiving waterbody or during disposal of the sediment.

*Buffer strips* - As with natural and constructed wetlands, there are many types of buffer strip/zones that can function in a catchment, varying with width, vegetation cover and management. They protect water from diffuse pollution through a number of mechanisms: acting as a physical barrier to prevent sediment and sediment bound contaminants from entering the stream, increasing the retention time of sediment bound contaminants to allow degradation or utilisation by vegetation to occur, maximising the uptake of nitrate by the vegetation in the buffer strip and maximising the potential for denitrification within the buffer strip. They can also act as a physical barrier, reducing the likelihood of direct spreading of manure, fertiliser or pesticide into a surface watercourse. Their importance and use is likely to increase, as the use of buffer strips is an option in UK's agri-environment schemes.

In summary, these are the potential advantages of constructed wetlands:

- Generally effective in decreasing pollutant loads, depending on operational conditions;
- Low cost when the construction price is spread over the catchment area;
- Generally, low operating costs;
- Little labour required once operating;
- They use natural processes and have a high buffering capacity;
- They 'fit' into the landscape and are perceived as 'environmentally sensitive' and are generally approved of by the general public;
- They may have the potential for the secondary use of products, such as thatching reed or biomass energy crops;
- They usually increase the wildlife biodiversity of the local area providing key niches for several important species.

In summary, these are the potential disadvantages of constructed wetlands:

- Variable performance, depending on many factors;
- Potentially substantial construction costs, though these vary considerably depending on site requirements; Extreme weather events may overload the system catastrophically;
- There are limits to the level of contamination they can cope with, especially with regard to BOD and nitrate concentrations;
- Some point-sources (livestock holdings) they often need pre-treatment measures in addition;
- After a working life of 15 – 20 years the system may be laden with nutrient rich silt and organic sediments that are difficult to dispose of;
- Integrated systems need designing on an individual catchment basis to deal with the local pollution problem.

### *Potential for reducing the concentration and loading of agricultural pollutants in agricultural surface water catchments?*

There is considerable potential for the use of constructed wetlands, and they are being tried or used in several other countries. They occupy a small area of the catchment ( 0.1-0.7% is the quoted range), i.e. a system of 0.1 ha can serve a catchment of 50-100 ha. Costs of construction vary with individual circumstances, but if the cost is spread over the catchment area, this may be of the order of £120-320/ha with £6/ha annual running costs, depending on the complexity of the installation.

In terms of effectiveness of the approach of constructed wetlands, we have already commented on the variable performance of a system because of its dependence on many factors. The literature is full of reports summarising ‘typical’ values of effectiveness. However, behind these averages are wide ranges and the decision on the potential usefulness of the approach comes down to risk. Our arguments for advocating their use include:

- They generally show some reduction in pollution levels though we accept, in some circumstances, they may act as a source. Improved management and design will minimise this risk.
- They are part of the solution, not the entire solution. They are part of a multi-barrier approach as advocated by the WFD. Therefore, we would expect measures that focus on source and transport controls also to be employed.

This review shows, however, that there is further investigative work to be done on now to construct and manage the systems, as well as how to integrate them into the landscape. Questions also arise over their long-term benefit, the argument being that most monitoring projects tend to last 1-3 years. There are also questions about how to effectively monitor such systems, particularly when used for diffuse pollution control.

There are many parallels with the acceptability of biobeds for treating pesticide washings. This too is a biological system. Even though experiments have shown them to be an effective tool, there is some concern from policy makers over their adoption on farm – will they be managed correctly, will they remain effective, could they make the problem worse by concentrating the problem into small areas? Further demonstration, investigation and use in the catchment (with careful monitoring) will help to assess these risks.

Two issues would need to be resolved before they could be used in the UK:

- Ownership – who is liable and who is responsible for the maintenance. Presumably, this would need to be spread across the catchment from where water is draining.
- The need for licensing and discharge consents.

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# 1 CONTEXT AND BACKGROUND

This report was commissioned in March 2005 and completed in September 2005. The report represents a detailed review on the use of 'soft engineering' approaches, e.g. wetlands, reedbeds and buffer strips. The study aims to establish the feasibility of using such approaches to limit the environmental impacts of agriculturally derived pollutants.

## *1.1 Objectives*

The aim of this study is to determine whether "soft engineered" based systems can be established within agricultural surface water catchments to ameliorate water quality, with particular respect to agricultural diffuse pollutants and whether such approaches are likely to be cost-effective.

## *1.2 Specific Scientific objectives*

1. Review available literature relating to wetland and reedbed water treatment systems (natural and artificial) with regards to construction, hydrological characteristics, vegetation, efficacy, feasibility of installation and cost-effectiveness.

2. Provide recommendations on:
    a) the potential for use of these systems in reducing the concentration and loading of agricultural pollutants in agricultural surface water catchments,
    b) how the systems may need to be adapted to meet the requirements of agriculture and,
    c) whether the investment in such approaches will present other diversification opportunities.

3. Consider whether any initial screening work is required, carried out under controlled conditions, to assess the efficacy of the different systems.

4. Identify suitable, existing wetlands / reedbeds where input / output monitoring could be carried out for key indicator pollutants.

# 2 INTRODUCTION

Improvements in water quality over recent years have been due to the reduction in point source pollution, i.e. pollution from an identifiable source. However, although progress has been made in reducing point source pollution, diffuse pollution, is now of principal concern in terms of water quality (Anon., 2002). Diffuse pollution from agriculture has been particularly recognised as the biggest obstacle to the objectives of the Water Framework Directive, which requires controlled waters to have good ecological status by 2015. To help agriculture play its part in ensuring that water in England meets these requirements, the Catchment-Sensitive Farming programme is seeking to reduce the diffuse pollution contribution made from agriculture. Diffuse pollution can result in the release of a variety of substances to water, including nutrients, sediment, pathogens and chemicals.

While all of these substances are fundamental elements of a farmer's business, all can have significant effects on wildlife and water quality. Driven by the need to control diffuse pollution, a considerable amount of research has been performed, primarily focusing on appropriate land management techniques and how they can be implemented (e.g. better use of fertilisers and manure, cover crops, cultivation methods). There has also been some discussion about 'soft engineering' options, particularly buffer strips, to limit diffuse pollution. There are, however, other approaches (e.g. reedbeds, wetlands) which have previously been used in more industrial settings that could be used in agricultural environments, either to control point or diffuse pollution sources. It could, for example, be envisaged that a stream running through a farm could be 'treated' at the outlet to decrease contaminant loads. This may involve an artificial, heavily engineered solution, or a more natural wetland area.

This review will help to inform Defra of potential novel, landscape-based approaches to meeting water quality (and quantity) objectives, and how their introduction may fit with existing policy measures.

## *2.1 Pollution sources*

### *2.1.1 Agricultural point sources*

The sources of diffuse pollution from agricultural activities in part determine what form of soft engineering solution may be appropriate, and secondly how effective they may be. Traditionally constructed wetlands and reedbeds have been used to ameliorate point sources of 'organic' pollution (chiefly sewage treatment) and, as such, may lend themselves to the treatment of agricultural 'point-sources' from dairy parlours and livestock hardstanding and holding areas. They can be considered as point sources because they are from easily defined places on the farm and are generally collected for future disposal/use.

*Table 1. Typical composition of liquid wastes on farms that could be considered point sources of waste when stored. Source: various.*

| Material | BOD (mg $O_2$ $L^{-1}$) | N (mg $L^{-1}$) | P (mg $L^{-1}$) |
|---|---|---|---|
| Milk | >100,000 | | |
| Silage Effluent | 65,000 | 2,500 | 600 |
| Pig Slurry (4% DM) | 25,000 | 4,000 | 800 |
| Dirty Water | 1,500 | 300 | <100 |

Whereas general dirty water, etc., may be sufficiently dilute to be treatable with soft engineering solutions, other agricultural effluents such as silage effluent, slurry and milk will be too 'potent' to warrant treatment by this method. For example, Table 1 gives typical compositions of these 'point sources'.

We might also consider pesticide washings as a point source (Carter, 1999; Bach *et al*., 2003), and 'biobeds' can be considered a soft engineering solution, relying on similar processes of degradation as wetlands (Fogg *et al*., 2003a&b).

### *2.1.2 Agricultural diffuse sources*

More often the term diffuse pollution is used to describe the pollution that arises more generally from fields and other large tracts of land where the original source cannot necessarily be precisely located, but nevertheless contributes to a much larger burden further downstream. This form of pollution arises equally from both livestock and arable farming practices, and in nature is chiefly;

- Nutrients such as nitrogen (N) and phosphorous (P).
- Soil particles.
- Agrochemicals (pesticides, plant protection chemicals, biocides).
- Microbial contaminants (pathogens).

In this case there are many within-field options for amelioration that can be considered before soft engineering solutions are implemented. These were reviewed recently, along with wetlands and reedbeds, by Vinten *et al.* (2005), but included such approaches as; altering the feed composition, stocking rate, and manure treatment, spreading rate, technique and timing for livestock fields; and also cultivation methods, contour management, strip cropping and choice of crops and pesticide/fertiliser planning for arable fields (taken from Dampney *et al.,* 2002).

In addition to the in-field option above there are a second rank of control methods that can be considered at the field margin. They include; barrier ditches, vegetative barrier strips and riparian buffer zones or strips. Collectively they can be considered as 'soft engineering' options, and their design, construction and effectiveness are considered in this review.

Whether, through treatment at the point of discharge, or treatment of diffuse pollution from an area of land, the use of (constructed) wetlands and reedbeds might be effective in mitigating agricultural diffuse pollution in the wider environment.

Whereas treatment of point sources is the more accepted approach, some innovative thinking might allow this approach to provide some protection against diffuse pollution, as part of an integrated approach. If so, the location of wetlands within a catchment is particularly important, so that they collect the discharge from several fields and point sources within a holding, and use is made of natural watercourse features (Braskerud, 2001).

# 3 TREATMENT OPTIONS

The following section reviews the different 'soft' options for the treatment of agricultural diffuse pollution. It is worth noting that these treatment options do not fall into distinct categories and, often, different types will merge as one treatment option, or will be referred to by different names. However, for the purpose of this review, two broad categories are discussed – wetlands and buffer zones – under which are a number of sub-types.

**Wetlands** are generally defined as areas that 'are inundated or saturated by surface or groundwater at a frequency sufficient to support a prevalence of vegetation typically adapted for life in saturated soil conditions' (Mitsch & Gosselink, 2000). A wider definition, agreed in the Ramsar[1] Convention 1971, defined wetlands as 'areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static, flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six meters' (RAMSAR, 2005). Natural wetlands are often ecosystems that form a transitional gradient between open water and dry land, having characteristics of both (Otte, 2004). As well as being valuable for their inherent beauty, they provide a range of wildlife habitats for a large number of species and an often vital wildlife corridor link between other habitats. It is estimated that over 660 species of plants and 7,500 invertebrates live in UK wetlands. Wetlands also provide humans with a large range of wetland products, such as thatching reed. However, the value of wetlands goes far beyond directly tangible "products" that can be harvested. Wetlands provide a range of inter-linked socio-economic, physio-chemical and conservation functions (Water Policy Team, 2005).

To summarise, wetlands provide a number of key ecosystem functions and services (Otte, 2004):
- Production of biomass.
- Cycling of carbon, N and other plant nutrients.
- Hydraulic moderators.
- Improvement of water quality.
- 'Highways' for wildlife.
- Shelter and food for wildlife.
- Hunting and fisheries.
- Aesthetic value and recreation.

In the UK, anthropogenic activity has largely led to the progressive degradation of wetlands (changes in flood regimes due to flood control, afforestation, poor burning practices and extraction of peat), particularly in the SE (Blackwell *et al.,* 2002). A principal impact on wetlands has been from agricultural activity through, in particular, the drainage of land for conversion to arable/cropped and cessation of traditional grazing practices. Furthermore, increasing inputs of nutrients from runoff and direct fertiliser applications have altered the composition of vegetation in many wetlands (English Nature, 1992). The remaining natural wetlands are under threat of continued degradation or complete loss, although their protection and restoration is now a key issue for environmental policy. For example, the response of the RSPB involved the protection and enhancement of existing wetlands by reserve management and the creation of new wetlands (Benstead, 2000). As such, many

[1] Ramsar sites are wetlands of international importance designated under the Ramsar Convention. In selecting sites, the relevant authorities are guided by the Criteria set out in the Convention. The UK also has a national Ramsar Committee composed of experts who provide further advice. Compared to many countries, the UK has a relatively large number of Ramsar sites, but they tend to be smaller in size than many countries. The initial emphasis was on selecting sites of importance to waterbirds within the UK, and consequently many Ramsar sites are also Special Protection Areas (SPAs) classified under the Birds Directive.

are now strictly managed – RSPB manage around 50 significant wetland areas alone - and are usually designated as conservation areas with biodiversity as the main focus.

One important effect of the loss of natural wetlands has been to reduce the capacity of wetlands to buffer the aquatic environment against pollution. It is this precise ability of wetlands to regulate both water quality and quantity that has motivated many investigations into the use of constructed wetlands (Blackwell *et al.,* 2002), which simulate natural wetland processes as a preferable alternative to conventional treatment systems (Nuttall *et al.,* 1997).

The general processes by which wetlands are considered to regulate water quality result from the unique combination of vegetation, soils and the associated microbial assemblages, all of which act upon pollutants through a variety of processes (Table 2).

*Table 2. General wetland removal mechanisms for agriculturally derived pollutants. Sources: Blackwell et al. (2002); Cooper et al. (1996).*

| **Agricultural Pollutant** | **Associated water quality problems** | **Wetland process for amelioration** |
|---|---|---|
| Nutrients<br>(especially N and P) | Eutrophication<br>Toxicity | Ammonification followed by microbial nitrification and Denitrification<br>Chemical precipitation<br>Plant Uptake<br>Adsorption<br>Sediment deposition/retention<br>Ammonia volatilisation |
| Pesticides/herbicides | Toxicity | Adsorption<br>Plant uptake |
| Sediment | Eutrophication<br>Silting of gravels | Sediment deposition/retention<br>Hydrological regulation<br>Filtration |
| Pathogens | Disease | Sediment deposition/retention<br>Adsorption<br>Predation<br>Filtration<br>Natural die-off<br>UV irradiation<br>Excretion of antibiotics from roots of macrophytes |
| Heavy Metals | Toxicity | Plant uptake<br>Sediment deposition/retention<br>Adsorption and cation exchange<br>Chemical precipitation<br>Microbial oxidation/reduction |
| BOD and OCD | De-oxygenation | Adsorption<br>Sediment deposition/retention<br>Oxidation/mineralisation |

The reduction in each pollutant and the success of the processes by which it is treated varies according to wetland type. Hammer (1992) defined four main types of wetland:

1. Natural wetlands – natural.

2. Restored wetlands – on areas that previously supported a natural wetland ecosystem, but have been modified and used for other purposes and then subsequently restored to original status.
3. Created wetlands – formerly had well-drained soils supporting terrestrial flora and fauna but have been deliberately modified to establish wetland conditions.
4. Constructed wetlands – former terrestrial environments that have been modified to create wetland conditions for the primary purpose of contaminant/pollutant removal from wastewater.

This review provides a focus upon natural wetlands and constructed wetlands, including the sub-categories of reedbed and integrated constructed wetlands.

### *3.1 Natural Wetlands*

Natural wetlands take a variety of different forms. Under the UK Biodiversity Action Plan, four particular types have been identified as priority habitats – Reedbeds, Grazing Marsh, Fens and Lowland raised bogs. Other priority habitats have wetland elements including grazed lowland heath and wet woodlands (English Nature, 2005). Of these, the reedbed habitat 'type' is primarily recognised in the literature as having the potential to act as a 'natural' treatment system for agricultural wastewater, although it is likely that all may regulate water quality in some way (e.g. wet woodlands forming riparian buffer zones).

In the UK, reedbeds originate naturally or as an indirect consequence of human activity (e.g. where agricultural practices have ceased, usually where grazing and water control have been abandoned on low-lying land). A reed-swamp, or wetland, represents the early stages of succession from open water to woodland and, without management, a reedbed will gradually dry out, becoming colonised by other grasses and tall herbs, eventually developing into scrub and woodland (Hawke & Jose, 1996). Whilst this is a natural process, it can be both accelerated (by drainage, water abstraction or isolation from water courses) or slowed/reversed (through management and rehabilitation). The maintenance of reed domination is considered as possibly the single most important aspect of management for reedbed birds (Burgess & Evans, 1989), and can be further managed to promote the development of more diverse flora and fauna.

Irrespective of vegetative/habitat type, there is considerable variation in the occurrence of wetland systems. For example, wetlands often form natural riparian zones. Many natural riparian zones exist, and are considered to be particularly effective in buffering streams from potential nutrient pollution (Lowrance *et al.,* 2002). Their physical, chemical and biological processes can function through the assimilation and transformation of contaminants before they can be transported into stream waters. Other systems may form as in-stream wetlands. Sebilo *et al.* (2003) reported the importance of both the riparian zone and in-stream in the cycling of nutrients. Nitrogen budgets established for large river systems reveal that up to 60% of the nitrate exported from agricultural soils is eliminated, either when crossing riparian wetland areas before even reaching surface waters, or within the rivers themselves through benthic denitrification (Sebilo *et al.,* 2003). Where there are heavy applications of N to agricultural land or drainage that bypasses the riparian zone, the N-cycling process of in-stream wetlands can be very important. Hunt *et al.* (1999) found that re-establishing a small 3.3 ha in-stream wetland was very effective in lowering the mass load of N from a 425 ha agricultural watershed. Current practices for using in-stream cycling of N have generally focused on the benthic processes. However there have also been advances in other methods, one being that of floating wetlands, which are effective and aesthetically appealing (Hunt *et al.,* 2004).

The potential for natural wetlands to act as treatment systems for agricultural pollutants is, however, questionable. Most remaining natural wetlands in the UK and Europe are now under some form of conservation designation; therefore, the primary function of these wetlands is that of habitat provision. There is some debate as to the viability of combining habitat conservation and the treatment of diffuse pollution within the same wetland system and this is discussed further in later sections.

However, one benefit of the study of natural wetlands is that they can provide analogies and be used as models for the provision of constructed wetlands as alternative treatment systems (Otte, 2004).

### *3.2 Constructed Wetlands*

Constructed wetlands are engineered systems that have been designed and constructed to utilise the natural processes involving wetland vegetation, soils, and the associated microbial assemblages to assist in treating wastewaters (Vymazal, 2004).

In the UK, virtually all water utility companies have constructed wetlands for treating sewage, designed to deal with point-source pollution such as industrial effluents, domestic sewage, drain water and storm water runoff. Non-point source pollution, however, such as fertiliser run-off from arable land, cannot be provided for so effectively (Hawke & Jose, 1996). However, in the USA, wetlands have long been constructed to deal with agricultural diffuse pollution from pig farms, crop run-off and livestock wastewater treatment. Most involve the use of constructed grasslands, ponds and emergent marshes (Hammer, 1992). In California, combined marsh and forest systems of around 37 ha have been constructed to make use of treated effluents (Hawke & Jose, 1996). Over the past decade or so, the application of constructed wetlands for agricultural treatment purposes has been given more attention in Europe. In the UK, this has primarily been implemented through ongoing experimental studies conducted by the University of Birmingham, whilst in Ireland a great deal of work has been carried out over the past decade.

The basic principle of a constructed wetland is that wastewater enters through an inlet and, as it passes through the system, undergoes biological and physiochemical transformation before release at the outlet. Constructed wetland have been found to be effective in reducing suspended solids, oxygen depleting substances, organic particulates, nutrients, and most other chemical and biological pollutants including hydrocarbons, de-icing agents, colour and bacteria (Nuttall *et al.,* 1997). If constructed and managed properly, constructed wetlands offer real advantages over conventional treatment systems (McGarrigle, 2004).

Akin to natural wetland systems, the most popularly cited design for constructed wetlands is that of constructed reedbeds. In addition to their extensive use for wildlife conservation, concern over the decline in quality and quantity of reedbeds and their wildlife has increased interest in creating them as way of spreading the risks. Historically, in the UK, the majority of constructed wetlands designed to treat point source pollution have utilised pure reedbed treatment systems (Hawke & Jose, 1996). The popularity of the reedbed for constructed wetlands has extended to systems designed for treatment of agricultural diffuse pollution.

Constructed wetlands can also be located in the upper reaches of water courses themselves, where the level of pollution risk does not warrant their installation at field margins. This may be prove particularly effective at or after the confluence of several small streams that may collect low levels of pollution together until it poses a threat at medium scale stream size. Wetlands can be adapted to include natural features at stream headwaters (Hill, 1990) or artificial constructions at drain outflows (Petersen *et al.*, 1992), and operate by creating shallow (<0.5 m according to Uusi-Kämppä *et al.*, 2000) vegetated areas with slow moving water, which impede sediment flow and provide a sink for soluble nutrients. They are best used in conjunction with other methods, such as buffer strips (Kovacic *et a*l., 2000) and/or settling ponds (Uusi-Kämppä *et al.,* 2000) and, when compared with settling ponds on their own, constructed wetlands retained 41% of the incident total P compared to 17% in the ponds, though these values were often much lower than buffer strip mitigation (27–97% reduction depending on width)( Uusi-Kämppä *et al*., 2000).

### *3.2.1 Constructed Wetland Design*

Essentially an engineered system designed to simulate a natural wetland, the constructed wetland has two fundamental design types:

- Surface flow – water flows along surface channels through the lower stems of wetland plants rooted in flooded soil (i.e. wastewater flows above the support medium).
- Subsurface flow water percolates through the root zone of wetland plants growing hydroponically in flooded gravel-filled channels.

Systems are further subdivided into categories that reflect how the waste enters the system:

- Horizontal flow – water flows in level with, or slightly above, the surface.
- Vertical flow – water falls or cascades onto the wetland.

Numerous studies of constructed wetlands have illustrated the importance of these design types in the effective treatment of the various agricultural pollutants. It is often appropriate to employ a specific design for particular purpose (Table 3).

*Table 3. Characteristics of constructed wetland types. Source: Blackwell et al. (2002).*

| Wetland Type | Sub-type | Size | Hydrology | Vegetation | Efficiency |
|---|---|---|---|---|---|
| Surface flow wetlands (SFW) | N/A | Usually large due to the restricted interaction of water with support medium. Typical range between 100-10,000 m$^2$. | Water flows over the surface of the support medium, which restricts interaction with microbes. Flow is usually continuous. | Can support emergent, submergent and floating vegetation. | Highly efficient for lowering BOD, COD and particulate pollutants. Less efficient for dissolved pollutants, particularly N and P. |
| Subsurface flow wetlands (SSFW) | Horizontal flow | Usually much smaller than SFWs, typical range between 10-3500 m$^2$. | Water flows through support medium predominantly in a horizontal direction. Flow is usually continuous. | Support emergent vegetation only. | Highly efficient for reducing BOD and COD, even at low temperatures, and more efficient than SFWs. |
| | Vertical flow | Typically the smallest of all constructed wetlands, ranging from a few square metres and not normally exceeding 100 m$^2$. | Water flows through the support medium in a vertical direction. Flow is usually periodic. | Supports emergent vegetation only. | Highly efficient for removing dissolved N and P, but low efficiency for reducing BOD and particulate pollution. |

Constructed wetlands have been used for many types of wastewater including industrial, agricultural, landfill leachate and storm water runoff. As many of these wastewaters are difficult to treat in a single stage wetland system, hybrid wetland systems that consist of various design types of constructed wetlands staged in series have been increasingly used.

It has been recognised that this is also an issue for the treatment of the various different agricultural pollutants. Many pilot and experimental studies demonstrate that constructed wetland systems are increasingly unlikely to be a single unit, but rather an integration of units that may incorporate different design types, and also different vegetation and habitats (e.g. reedbeds, marshes, ponds, grasslands and even forest/shrub areas) (Wood, 1995).

#### 3.2.1.1 Design Criteria

Aside from the specific design type, the design criteria for constructed wetlands is also of paramount importance. Key criteria for constructed wetlands encompasses pre-treatment, hydrology, vegetation, support medium and ambient weather conditions. These key components of constructed wetland systems have considerable influence on the treatment effectiveness (Tables 4-7).

*Table 4. Pre-treatment criteria. Source: Nuttall et al. (1997).*

| **Pre-treatment** | | | |
|---|---|---|---|
| **Purpose** | Primary treatment for solids separation | Agricultural wastewater will require some form of pre-treatment prior to wetland treatment in order to lower BOD | Where wastewater is intended for crop irrigation re-use, pre-treatment of P-rich wastewater should limit N removal to maintain the N:P ratio |

*Table 5. Support medium criteria. Source: Nuttall et al. (1997).*

| **Support Medium** | | | | | |
|---|---|---|---|---|---|
| **Importance** | Influences retention time of wastewater, where retention time is a function of flow rate and hydraulic conductivity of medium | | | | |
| **Role** | | | | | |
| Surface Flow Wetlands | Sediment sink for deposited suspended solids and particulate pollutants, including a proportion of BOD | Bedding medium for aquatic vegetation | Habitat for microscopic and macroscopic animals and plants | Chemical ionic interchange between wastewater and ion-rich mineral components at the sediment-water interface | |
| Subsurface Flow Wetlands | Physical separation of particulate together with particulate-bound pollutants | Attachment sites for microbial growth (aerobic and anaerobic) | Support surfaces for rooted emergent macrophytes | Chemical ionic interchange between wastewater and ion-rich mineral components in the support medium | Oxygen-rich or oxygen-limited microzones |
| **Characteristics** | Sand has lower hydraulic conductivity than gravel or rock | Over time, as wastewater is introduced medium undergoes chemical and physical changes | Nature of support medium may also change to due to accretion of micro-organisms and organic matter, accumulation of debris, reduction in pore size, slime growth increasing stickiness | | |

The changes in the characteristics of the support medium can lead to changes in pollutant removal rates. For example, in coarse media as pore size reduces over time the filtering action improves. However, without maintenance, the support medium could become sticky via slime growth and eventually blocked with consequences for flow rates and flow patterns (i.e. a gradual change from subsurface flow to surface flow).

*Table 6. Hydraulic design criteria. Source: Nuttall et al. (1997).*

| **Hydrology** | |
|---|---|
| **Removal rate** | Impacted by precipitation, infiltration, evapotranspiration, hydraulic loading rate, support medium, water depth |

The hydrology characteristics are probably the most important feature of constructed wetlands, as they influence the integrity, characteristics, biology and productivity of the system. The factors that impact on the removal rate of pollutants do so through the alteration of retention time and by concentrating or diluting the wastewater. Surface flow wetlands, in particular, respond to rain and evapotranspiration, therefore these should not be ignored in design (Kadlec, 1995).

Further hydrological characteristics may have direct effects on the response and performance of the aquatic vegetation in a constructed wetland. Small differences in depth and water regime can significantly effect accumulation and allocation of nutrients and biomass, which, in turn, can influence vegetative response.

*Table 7. Aquatic vegetation criteria. Source: Nuttall et al. (1997).*

| **Aquatic Vegetation** | | | | |
|---|---|---|---|---|
| **Properties** | Provision of support for microbial attachment | Stabilise bed surface and reduce scouring through reduction in wind induced turbulence | Shade and attenuate light to provide a constraint on algae growth in water column | Influence soil hydraulic conductivity |
| **Species** | Species-specific response to water depth and regime | Species-specific pollutant removal effectiveness | . | |
| **Benefits** | Provision of habitat for wildlife | Provision of thermal insulation in cold weather | Reduction in water velocity for flood defences or enhanced treatment potential | . |

Tanner *et al.* (1995a & b) demonstrated the key role that vegetation plays in the removal of some pollutants. In experimental trials comparing planted and unplanted gravel bed constructed wetlands, they showed that, whilst the removal rates for suspended solids and faecal coliforms showed little difference between planted and unplanted, there were significant differences in the removal rates of N and P, with the planted wetlands demonstrating higher efficiency. One of the main reasons for this is that plants most likely enhance microbial transformations, and appear to 'be the only sustainable means of nutrient removal' (Tanner *et al.,* 1995a & b).

### *3.2.2 Design Choice*

The choice of constructed wetland design must be made on the basis of the geology, hydrology, topography of the site in question, as well as the type of pollutants to be treated.

#### *3.2.2.1 Surface flow*

In surface flow systems, air-water interactions are very important, as oxygen transfer to wastewater is dependent upon atmospheric input. Such wetlands have a greater capacity for solids storage than subsurface flow, because they are not filled with support medium, whilst vegetation is physically important by helping to distribute flow, dampen incoming flow velocity, filter wastewater solids and function as support structures for microbiological growth (Brown 1994).

There are various different design types for surface flow constructed wetlands, creating a range of differing ecosystem habitats (Table 8), with some proving more effective for particular pollutants than others. For example, Rafted Lagoons have been suggested to be more effective in the treatment of nitrified effluents than the more conventional vegetation lagoons (Nuttall *et al.,* 1997), whilst Polyculture systems have been used more extensively in North America and Australia to provide ecological benefits in addition to wastewater treatment. Similarly, the different designs suffer both type-specific and generic surface flow problems. For example, three types of hydraulic inefficiencies may occur in surface flow treatment wetlands caused by internal islands and other topographical features; preferential flow channels at a large distance scale; mixing effects, such as water delays in litter layers and transverse mixing – and all three influence the ability of the wetland to improve water quality (Kadlec, 1995).

*Table 8. Surface flow design types. Source: Nuttall et al. (1997).*

| Surface Flow Design | Method | Requirements |
|---|---|---|
| **Overland Flow systems** | Wastewater applied in controlled, quantitative amounts | No more than slightly graded topography of less than 1% slope |
| **Constructed Marshes and Vegetated Lagoons** | Typically a sequence of sealed shallow basins containing 20-30 cm of rooting soil with a water depth of 20-40 cm. Dense emergent vegetation covers usually more than 50% of surface area. | Dense emergent vegetation (e.g. *Phragmites australis*; *typha spp.*); no constraints on land availability |
| **Rafted Lagoons** | Constructed rafts of emergent wetland plants used to increase performance of shallow, narrow lagoon systems where depth does not exceed 1 m. | Emergent vegetation |
| **Polyculture Systems** | Uses symbiotic relationship between aquatic plants and animals to improve water quality in impounded waters. | Aquatic macrophytes |

#### *3.2.2.2 Subsurface Flow*

Whereas wastewater flows across the wetland within surface flow wetlands, in subsurface flow wetlands, flow travels through a support medium, being treated as it moves through it. It is, therefore, essential that the support medium should be of high hydraulic conductivity.

These wetlands may have an impermeable liner and appropriate vegetation is rooted in the support medium and grows hydroponically in the wastewater as it flows past the roots. Unlike surface flow wetlands, however, air-water interactions are not so dynamic, as wastewater is designed to stay below the surface of the support medium. However, there is greater potential for interactions between support medium and water.

Once again, the different methods prove successful in different ways (e.g. Gravel Bed Hydroponics prove successful in the extensive control over hydraulic pathways). They also have individual operational issues, which can impede treatment efficiency. For example, whilst the harvesting of plants is not recommended for the RZM design, the MPIP method recommends that harvesting is done at least once a year in order to avoid operational difficulties associated with clogging.

The methods outlined in Table 9 are essentially horizontal flow systems. Subsurface vertical flow systems are essentially vegetated gravel beds planted with aquatic reeds. Typically consisting of channels filled with a graded gravel support medium, the controlled flooding of wastewater is undertaken across the surface of the bed in vertical down-flow systems to promote aerobic conditions within the support medium. Such systems have been recommended for the oxidation of ammonium to nitrate by nitrification, thus providing a fully nitrified effluent low in both BOD and suspended solids (Nuttall *et al.,* 1997).

*Table 9. Subsurface flow design types. Source: Nuttall et al. (1997).*

| Surface Flow Design | Method | Requirements |
|---|---|---|
| **Planted Soil Filters** | Artificial wetlands with subsurface horizontal flow in a soil support medium. | 1-5% slope to promote gravitational flow. |
| **Root Zone Method (RZM)** | Horizontal, one-stage, soil-based system aimed to enhance treatment potential aerobic bacterial activity in root zone and anaerobic bacterial activity in the surrounding soil. | Aquatic vegetation - *Phragmites australis*; surface slope of 1-5%. |
| **Gravel Bed Hydroponics** | Differs from other SSF horizontal systems by having shallow depths of 0.25-0.4 m and high aspect ratios as a result of the creation of channels 1-3m wide, 50-100 m long. | 1-5% slope, gravel support medium. |
| **Max Planck Institute Process (MPIP)** | Consists of lined or concrete trenches 2-4 m wide, up to 100m long with washed gravel or sand. Influent wastewater is intermittently dosed into system. | 1-5% slope; pre-treatment for algae control. |

#### *3.2.2.3 Settling ponds*

One effective mechanism for progressively reducing sedimentation risk, often as part of a constructed wetland system, is a series of small ponds along the flow path, such that particles in the in-flowing water can settle out in the still water. The depth of these ponds should be >1 m compared to the shallower vegetated areas (Uusi-Kämppä *et al*., 2000). The outflow from each pond is thereby progressively clearer, and vegetation in the ponds assists the removal of nutrients and creation of still areas by providing a physical barrier. Tanner & Sukias (2003) in New Zealand, also looked at linking ponds and wetlands to treat the effluent from sewage dairy and piggery wastewaters, finding that bacterial indicators were regularly reduced by one log-unit, but consistent achievement of coliform counts below 500 $100ml^{-1}$ was difficult.

### *3.3 Reedbeds*

Although reedbeds are a design criteria for many of the types listed above, it is worth nothing the particular design requirements, because reedbeds are so popularly used. In particular, land characteristics are very important for reedbed creation. A s outlined by Hawke & Jose (1996), the requirements are:

- a reliable, adequate water supply (sufficient to maintain a flow and up to 30cm surface depth in summer);
- some control of water levels (e.g. existing ditches, sluices etc);

- no potential to flood neighbouring land;
- free from saline intrusion;
- a level or very shallow gradient;
- an existing vigorous reed source to facilitate establishing reed across whole site;
- access for management.

### *3.3.1 Integrated systems*

As illustrated, the numerous designs for constructed wetlands represent a strong potential to treat a wide range of agricultural pollutants. However, it is evident that no single design type alone would be able to comprehensively treat the variety of nutrients, particulates, pesticides and pathogens that are contained in agricultural wastewater. The constructed wetland system is increasingly unlikely to be a single unit but rather an integration of units that may include reedbeds, marshes, ponds, grasslands and even forested or shrubbed areas. The units may also operate as hybrid designs, incorporating both surface or subsurface systems, where appropriate, to optimise physiochemical pollution removal mechanisms, biological degradation, evaporation and infiltration (Wood 1994). Many systems now incorporate combinations of individual units or link with other conventional treatment systems so that the removal of a whole suite of pollutants can be optimised.

The integration of different units to treat a range of pollutants can be broadened into a full-scale watershed-based approach. This has been the focus of much work in Ireland where Integrated Constructed Wetlands (ICWs) are an holistic design specific approach to the use of constructed wetlands to improve water quality as part of wider ecological/environmental schemes, rather than serving a single purpose of improving water quality. The ICW approach and their use within a watershed have its origins in the ‘small watershed technique’ and associated ecosystem studies developed by Bormann & Likens (1981) and Siccama *et al*. (1970). The explicit integration of water quality improvement, along with biodiversity enhancement, provides synergies that facilitate wetland system robustness and sustainability.

The use of local soil material and a wide variety of wetland plant species in ICWs are some of their features that distinguish them from a conventionally engineered reedbed system that typically uses a single species. Furthermore, the explicit inclusion of ‘landscape fit’, ‘biodiversity’ and ‘habitat enhancement’ into ICW design is fundamentally focused on providing additional values to the site, as well as water treatment. The larger land areas used in ICW design, compared with those used in other constructed wetland designs, facilitated the incorporation of these environmental services. In addition the larger area also provides for greater system robustness and sustainability (Harrington *et al*., 2004).

The ICW concept also adopts an ecosystem-based approach to the management of farmyard waters, which is a more holistic approach to their management, rather than simple storage and disposal. Ecosystems such as constructed wetlands are self-regulating systems, which negate the requirement for intense management, although this does not remove the need for management entirely. The requirement of satisfying planning and regulatory authorities is achieved by demonstrating that the performance of an ICW does not impact negatively on the environment (Harrington *et al.,* 2004).

## ***3.4 Buffer strips***

A buffer zone is a generic term, defined as ‘a vegetated area lying between agricultural land and a surface water body, and acting to protect the water body from harmful impacts such as high nutrient, pesticide or sediment loadings that might otherwise result from land use practices’ (Blackwell *et al.,* 1999).

Such zones can take the form of wetland buffer zones (whether or not a wetland is acting as a buffer zone depends on the functions of the wetland), buffer strips (usually a narrow linear feature often on a field boundary or following a contour), or riparian buffer zones (situated adjacent to a river). From these definitions it is possible to disaggregate wetlands from buffer zones and hence this section shall focus on buffer strips and riparian buffer zones (Blackwell *et al.,* 2002).

Buffer strips are uncultivated zones at the edges of fields, adjacent to watercourses and ditches. They function as pollutant control agents through a number of mechanisms, including sedimentation, true filtration, infiltration of water, adsorption and pollutant uptake by vegetation. They also serve as a potential source of biodiversity in the landscape, and as corridors for wildlife. Often operated in conjunction with other in-field measures, and work by creating distance and a physical barrier between the source of pollution and watercourse. The maintenance of a buffer zone will allow sediment to settle, nutrients to be absorbed by bank-side vegetation and aerosols to be intercepted in the air by trees and tall vegetation.

Their development can be traced to studies of the natural ecology of riparian zones (Naiman & Decamps, 1989), and they are adopted in afforested areas (Olson *et al*., 2002; Weston 1995) as well as agricultural areas (Ducros & Joyce, 2003). They can range from a few metres of grass or natural vegetation up to complex strips > 50 m in width running the length of waterways.

One such complex system in the USA, is described by Hubbard & Lowrance (1994) as comprising three sections; 1) a narrow band (5-10 m) of permanent indigenous trees immediately adjacent to the stream; 2) a forest management zone where biomass production is maximised; and 3) a grass buffer strip <10 m wide located between agricultural areas and the forested zone. However, any one of these zones, or similar combinations, can be considered and maintained as buffer zones and will be effective to some degree at reducing diffuse pollution. Which is more appropriate will also depend upon the predominant form of pollution to be dealt with as they are more effective at different functions, and their application to UK conditions has been reviewed by Muscutt *et al.* (1993).

Buffer zones have been shown to be an effective means of reducing the inputs of sediment to aquatic ecosystems (e.g. Schellinger & Clausen, 1992). They are thought to achieve this via two main processes: firstly reducing the velocity of the runoff containing sediment, thus providing increased opportunity for sediment deposition, and; encouraging infiltration within buffer strip and therefore reducing the water depth flowing over the surface and hence the distance that particles have to fall. This is further explained by Leeds-Harrison *et al.* (1996), who listed the buffer zone methods of reducing diffuse pollution as:

- Acting as a physical barrier to prevent sediment and sediment bound constraints from entering the stream.
- Increasing the retention time of sediment bound contaminants to allow degradation or utilisation by vegetation to occur.
- Maximising the uptake of nitrate by the vegetation in the buffer strip.
- Maximising the potential for denitrification within the buffer strip.

However, all of these depend upon the hydrology of the catchment and the buffer strip, and this is considered to be the critical factor in determining the effectiveness of buffer zones. Buffer zones will only be capable of improving water quality if they are located in a position to intercept and process the hydrological pathways transporting agricultural pollutants (Blackwell *et al.,* 2002). Nevertheless, it is argued that much agricultural drainage is through subsurface pipes or ditches which results in large quantities of nutrient-rich water discharging directly into main watercourses (Whipple, 1991), thus rendering buffer zones ineffective. Hence, this adds to the overall conclusion that whilst riparian buffer zones do offer some level of protection, they can only be a tertiary measure to support other measures. In the absence of a landowner diverting drainage to wetlands, buffer strips located on contours, field boundaries or ditch systems have been suggested to optimise pollutant removal as their location will address specific problems (Mockler *et al.,* 1994).

The proportions of a buffer zone are also important for effective pollutant removal. Cooper *et al.,* (1987) suggest that the width of the buffer strips should be proportional to the contributing area, slope and the agricultural practices in the fields above and that consideration should be given to the nature of the drainage (for example, thin riparian strips are effective in trapping sand, but less effective for trapping clay particles).

Palone & Todd (1997) described buffers as 'one of the most effective tools for coping with [non-point source] pollution'. When employed effectively, this may be the case, but they need to be more complex than a simple strip of vegetation. Simpkins *et al.,* (2002) reviewed the implementation of Riparian Management Systems (RiMS) in Iowa, US, where multi-species riparian buffer have been shown to decrease nutrients, pesticides and sediment concentrations from runoff. The multi-species buffer typically consisted of: a zone of trees nearest to the stream to stabilise the bank, sequester chemicals and improve aquatic habitat; a zone of shrubs to provide woody roots, multiple stems and biodiversity, and; native prairie grasses to intercept runoff from the adjacent cropped field and provide rapidly cycling organic matter for microbial processes. Simpkins *et al.* (2002) further demonstrated that these buffers might also be optimised to treat nutrients and pesticides in the groundwater beneath them, through the careful choice of location based on hydrogeologic characteristics. The review suggested a shallow groundwater flow system which channels water directly through the riparian buffer at velocities that allow for processes such as denitrification to occur.

As with constructed wetlands, an integrated holistic watershed level approach is a particularly pragmatic way in which to optimise large scale diffuse pollution control. Meals & Hopkins (2002) report on P reductions following riparian restoration in two US agricultural watersheds as part of a wider cross border (Canada) National Watershed Monitoring Program. Voluntary landowner participation and 100% cost-sharing by the program resulted in a paired watershed design, effective in controlling the influence of extreme variations in precipitation and streamflow over six years of monitoring (Meals & Hopkins, 2002). Reductions of ~20% in mean total P concentration and ~20-50% in mean total P load were observed.

# 4 PERFORMANCE

## *4.1 General performance*

### *4.1.1 Wetlands*

Research does show that wetlands can provide a low maintenance method for reducing the input of nutrients and other agriculturally derived pollutants to surface and shallow groundwaters (Blackwell *et al.,* 2002).

However, their performance is variable depending upon a whole range of factors, including:
- the nature, quantity and timing (seasonality) of pollution inputs;
- wetland size;
- wetland ability to interact with pollutants;
- the appropriateness and accuracy of design;
- management practices and;
- the availability of site-specific data for wetland design.

A survey of the effectiveness of constructed wetlands in New Zealand showed that 75% were perceived to be meeting or exceeding discharge requirements (Tanner *et al.,* 2000). Major failures often reflected the application of increasingly stringent discharge quality requirements since wetland commissioning, the inability of the wetland to compensate for upstream treatment problems, or construction and/or persistent management problems (Tanner & Sukias, 2003).

With respect to constructed wetlands, although many have been successful, some systems have performed poorly. This failure has often resulted from over-optimistic design or unsuitable configurations, poor management or the rigid application of a particular design to a wastewater problem (Nuttall *et al.,* 1997). One example of design issues relating to performance was provided by Mandi *et al.* (1996), who demonstrated that the designed length of a reedbed was influential on the pollutant levels removed.

Research further suggests that the different design types may also be limited in their treatment performance for particular pollutants. For example, vertical systems have been promoted for the ammonification of N and oxidation of ammonium to nitrite and nitrate, thus providing a fully nitrified effluent, low in both biochemical oxygen demand and suspended solids. Horizontal systems, however, are suggested to be better suited for BOD removal, providing higher efficiencies than surface flow wetlands (Blackwell *et al.,* 2002). Combined systems which integrate surface flow, horizontal flow and vertical flow with each other or which are used in combination with conventional treatment systems are seen to be the most effective way of removing a whole suite of pollutants.

The effectiveness of water treatment in such systems can also be highly variable between sites. A constructed wetland seldom functions as a true sink for nutrients and other contaminants and is more likely to have multiple role as source, sink and transformer depending on location, season and environmental factors (Nutall *et al.,* 1997). Hydraulic features, support medium, vegetation and microbial activity all influence the constructed wetlands ability to retain or metabolise and degrade constituents contained in the influent while simultaneously releasing organic matter and other substances into the outflow.

### *4.1.2 Reedbeds*

A 1990 survey of reed growth in UK reedbed treatment systems found that the establishment of reeds was a key issue (Parr, 1990) for performance. Despite being theoretically possible to gain 100% reed cover within 2 years, only 35% of the 52 beds surveyed supported satisfactory reed growth (10% empty quadrats), whilst a further 35% had more than 50%

empty quadrats. None of the beds had established a mature rhizome system, which is necessary to increase soil hydraulic conductivity. Parr suggested that a number of factors affected reed growth, and overall performance, such as the support medium and weed growth and control. Soil water status was found to be the most crucial issue, whilst different management techniques were required for different support mediums (i.e. flooding and constant top ups for soil; raising and lowering of levels for gravel to promote aeration).

One particular performance issue is that most agricultural effluents are much too strong to be economically treated using reedbed treatment systems alone. Gray *et al.* (1990) noted that reedbeds are not always able to provide reliable treatments unless part of a complex, tiered treatment system. Based on initial results from two horizontal flow reedbeds treating farmyard runoff, which indicated their inability to reduce high levels of BOD due to insufficient oxygen transfer in the beds. However, with subsequent additions of two more downflow beds, significantly higher oxygen fluxes were achieved. As part of a complex tiered system, it is also necessary for higher strength agricultural effluents to be pre-treated prior to reedbed treatment. The principle reason for this is that the maximum strength that can be effectively treated with a conventional combination reedbed treatment system has been shown to be approximately 2000 mg $L^{-1}$ BOD and 650 mg $L^{-1}$ suspended solids (Job *et al.*, 1991). However, a further reason for pre-treatment is that, whilst reed grows well in eutrophic water, high levels of nutrients have been implicated in some cases of ‘reedswamp’ decline, most notably in the Norfolk Broads (Crook *et al.,* 1983). Thus, a combination of pre-treated vertical and horizontal flow reedbed treatment systems is viewed as the typical and most effective method to treat agricultural wastewater (Cooper *et al.,* 1996).

### *4.1.3 Buffer Zones*

The performance of buffer zones has been suggested to be dependent upon a number of criteria concerning the size, design and management of the buffer (Storm Water Center, 2005). This includes the minimum buffer width, which Cooper *et al.* (1987) suggested should be proportional to both the physical characteristics of the contributing area and its land use. Further criteria for effective performance is not only implementing a tiered buffer zone system rather than relying upon individual buffers (Simpkins *et al.,* 2002; Storm Water Center, 2005), but also the effective subsequent management.

*Table 10. Factors affecting buffer pollutant removal performance. Source: Storm Water Center (2005).*

| Factors that Enhance Performance | Factors that Reduce Performance |
|---|---|
| Slopes less than 5% | Slopes greater than 5% |
| Contributing flow lengths < 150 ft. | Overland flow paths over 300 feet |
| Water table close to surface | Groundwater far below surface |
| Permeable, but not sandy soils | Compacted soils |
| Growing season | Non-growing season |
| Long length of buffer | Buffers less than 10 feet |
| Organic matter, humus, or mulch layer | Snowmelt conditions, ice cover |
| Small runoff events | Runoff events > 2 year event. |
| Entry runoff velocity less than 1.5 ft/sec | Entry runoff velocity more than 5 ft/sec |
| Buffers that are routinely mowed | Sediment buildup at top of buffer |
| Poorly drained soils, deep roots | Trees with shallow root systems |
| Dense grass cover, six inches tall | Tall grass, sparse vegetative cover |

Pollutant removal effectiveness of buffers depends on the design of the buffer, including the use of vegetation types and the consideration given to localised topography and weather variables. Table 10 indicates some of the factors which affect buffer pollutant removal performance, whilst Table 11 provides some examples of quantitative performance results.

*Table 11. Quantitative performance results of buffer zones (% removal rate). Source: Storm Water Center (2005).*

| Source | Vegetation | Width (m) | Pollutant | | |
|---|---|---|---|---|---|
| | | | TSS | TP | TN |
| Dillaha *et al*.1989 | Grass | 4.6 | 63 | 57 | 50 |
| | | 9.1 | 78 | 74 | 67 |
| Magette *et al*. 1987 | Grass | 4.6 | 72 | 41 | 17 |
| | | 9.2 | 86 | 53 | 51 |
| Schwer & Clausen 1989 | Grass | 26 | 89 | 78 | 76 |
| Lowrance *et al.* 1983 | Native hdwd forest | 20 - 40 | - | 23 | - |
| Doyle *et al*. 1977 | Grass | 1.5 | - | 8 | 57 |
| Barker & Young 1984 | Grass | 79 | - | - | 99 |
| Lowrance *et al*. 1984 | Forested | - | - | 30-42 | 85 |
| Overman & Schanze 1985 | Grass | - | 81 | 39 | 67 |

The efficiency of diffuse pollution control depends on a number of factors, which link with the mechanisms. The most important factors affecting performance are:

- Pollutant type
- Buffer zone width
- Slope
- Vegetation type
- Source area
- Subsurface processes

#### *4.1.3.1 Type of pollutant*

Vegetated buffer strips can be over 90% effective at removing sediment bound P (Withers *et al*., 1998) and it is grass or dense herbaceous vegetation that is most effective for this. However, some studies have found them to be only moderately effective in the longer-term (Dillaha *et al*., 1989). Buffer strips can prove equally effective for sediment bound pesticides, such as chlorpyrifos, 80% of which was retained in grassed buffer strips in studies by Arora *et al.* (2003) compared with 49.7 and 51.2% of the less strongly bound pesticides atrazine and metolachlor, respectively. Their experiment also showed that a field to buffer strip area ratio of 30:1 was all that was necessary to achieve this level of removal. There was not a significant increase in removal when the ratio was 15:1. Boyd *et al.* (2003) showed that herbicides such as acetochlor and atrazine, which are mainly transported in solution in surface and subsurface flow, are controlled more by infiltration rate than deposition. It is, therefore, advisable that buffer strips are not located over permanent under-drainage systems, which may collect such infiltration (Haycock & Burt, 1991).

Soluble P and nitrate pollution are also not contained very well by narrow vegetated buffer strips (Withers *et al*., 1998; Cuttle *et al*., 2004). A paired study of headwater catchments in the UK showed grassed buffer strips to be ineffective at reducing nitrate outflow to streams (Leeds-Harrison *et al*., 1999), though they can be effective at removing N in storm run-off from grassland (Heathwaite *et al*., 1998) where more of the N is in organic and particulate form, and because buffer strips will impede surface runoff more than throughflow. Schmitt *et al.* (1999) also found that removal of particulate materials by grass buffers was much better than removal of soluble pollutants. Little extra pollutant removal was obtained by extending buffer strip width from 7.5 to 15 m. Young trees and shrubs planted in the lower half of the 15 m buffer had no effect on buffer performance.

Bingham *et al*. (1980) found considerable variation in the efficiency of removal of the various pollutants, which can be attributed mainly to their degree of association with sediments. Syversen (2001) found that in simulated runoff experiments at 3 sites, with 5 m and 10 m buffers, grassed or forested, the retention of suspended solids (SS) (particle size not given) was >80% in most cases, intermediate for total P (40-60% in most cases) and low for total N (40% in most cases). However, the total N and total P were as soluble forms in this experiment, and flows were quite low (shallow sheet flow, not concentrated runoff). Retention efficiencies for total N and P were better in natural runoff conditions, and retention of SS was similar. Retention efficiency of the finest clay fraction (0.06-0.2 um) was greater than larger fractions. Syversen (2001) interprets this as transport as aggregates. If this is the case, the extent of P transport will depend on whether the soil aggregates deflocculate during transport but it could also be because the finest clay particles will be more readily filtered, as their diffusion coefficient is significant (Ives, 1975).

Coyne *et al.* (1995) compared the transport and deposition of faecal coliforms and soil from 22 m long poultry manure treated plots through grass filter strips 9 m wide, during simulated rain of high intensity (64 mm/h for 90 minutes – an approximately 10 year return period event). Whereas sediment trapping accounted for 99% of input on both plots, 24% and 57% of faecal coliform inputs were transported across the two plots into runoff. Note that there were no concentrating effects in these field plots, so where runoff becomes concentrated, or catchment areas relative to buffers are larger, this event would be a lot more frequent than a 10 year return.

#### *4.1.3.2 Buffer zone width.*

Buffer strip establishment leads to reallocation of agricultural land to non-productive land. Recommendations vary from 10-60 m for sediment removal and 5-90 m for nutrient removal depending on slope, vegetation, soil, etc. (Castelle *et al*., 1994). Bingham *et al.* (1980) investigated the effect of grass buffer zone length in reducing the pollution from land application areas in grassed clay loam soils, with ratio of length of buffer zone to waste application zone varying from 0 to 2.6. Syversen (2001) found that increase in width led to higher retention but only by a small amount (7.4, 1.9 and 4.8% average increase across the 3 sites for total P, SS and total N (10 m instead of 5 m). The larger volumes led to poorer retention. Forested buffer strip showed higher retention of SS but not of nutrients. Syversen (2001) studied the retention efficiency of buffer strips as a function of (i) Buffer zone width, (ii) amount of surface runoff water, (iii) seasonal variation and (iv) vegetation type at 4 sites in SE Norway. Relative retention increased with width, but specific retention decreased, because retention efficiency was highest in the upper part of the buffer. Winter retention of sediment was 15-35 times higher than summer retention because more sediment transport occurred in winter.

#### *4.1.3.3 Slope.*

Buffer strips are less effective on steeper gradients, or where outflow is concentrated into channels which may carry run-off across narrow (<10 m) strips (Withers *et al*., 1998). Rose *et al.* (2003) showed that the deceleration of flow velocity which occurs upslope of a buffer strip at modest land slopes plays a crucial role in the net deposition of sediment-laden water. These authors described how re-entrainment of particles, and the carriage of larger particles further into the vegetated zone, increase progressively with gradient and sediment flow rate.

Slopes >0% may not be effective when establishing buffer strips as these will tend to erode, rather than acting as sinks for sediment, especially if the slope of the adjacent farming area is less severe. Decreased efficiency with slope increase from 11 to 16% (Dillaha *et al.,* 1989) and 7 to 12% (Robinson *et al.,* 1996) have been reported. This effect may, however, be mitigated to some extent by vegetation. Sediment deposition in buffer strips occurs at the leading edge of the vegetated area where an ‘hydraulic jump’ occurs (Rose *et al*., 2003).

This deposition on the leading, field-side, edge of vegetated buffer strips leads to a wedge shaped barrier or “berm” forming, which may subsequently cause ponding at the field edge. This may at first be beneficial by increasing infiltration rates over run-off, but will eventually cause transverse flow along the edge of the strip and lead to channelling and concentrated flow outbreaks across the strip. For that reason, buffer strips require periodic maintenance to remove these “berms” (Withers *et al.*, 1998). Slopes of <1% may also be unsuitable because of limited hydraulic gradients for infiltration and lateral movement of water (Haynes & Dillaha, 1992).

#### *4.1.3.4 Vegetation.*

Permanent pollutant removal in forest is generally rather higher than in grass. However, tree density is important as this will affect ground vegetation. If it is too dense then ground vegetation may be shaded out. Rapid vegetation growth leads to high permanent removal. Amongst tree species, aspen, poplar, and willow all have the potential for have high nutrient uptake. It should, however be remembered that uptake will only occur in the growing season. Density, height and stiffness of vegetation also affect efficiency of retention. Uusi-Kamppa (2002) found negative retention efficiency of dissolved P during snowmelt from a forest buffer strip compared with a reference plot. There was a positive retention from a grass buffer strip (cut, yield removed in summer). Leakage of dissolved P from frozen vegetation occurred in winter. Dillaha & Inamdar (1997) showed that particle retention efficiency decreases with time, due to sediment accumulation. However, Syversen (2001) showed no significant reduction in retention efficiency with time over 1992-1999. Uusi-Kamppa (2002) also showed increased efficiency with time in Finland (1992-2000).

The field margin may be different for grass fields compared with arable. Water may run along ploughed margins in the last furrow in arable fields, but connection in grass fields is direct and, therefore, there is less likelihood of flow concentration in grass. On arable fields, a ‘jagged edge’ to the ploughed field margin may be beneficial in providing extra sediment storage and less flow concentration.

Grassed buffers are best harvested to remove nutrients and to ensure vegetation does not get too long, when particularly for concentrated flows, the vegetation will collapse and submerged flow (less effective sediment removal) will occur. Short vegetation is often denser than tall vegetation, and tall vegetation will tend to lie flat during high runoff periods. Stiffer vegetation is more likely to provide protection than vegetation that falls over during flows over the buffer. For example, false oat grass (*Arrhenatherum elatius*) or tussock grass (*Deschampsia cespitosa*) are better than cocksfoot (*Dactylis glomerata*), which is better than ryegrass (*Lolium spp.*) or meadow grass (*Poa spp.*).

#### *4.1.3.5 Contributing area.*

Variable source area control is common in temperate catchments, where the close proximity of the water table to the land surface leads to overland saturated flow. Nearly all biologically active P export in the Brown Catchment, Pennsylvania, was attributed to less than 10% of the area (Pionke *et al.,* 1997). The source area relative to buffer area is an important design feature and Haynes & Dillaha (1992) recommend a ratio of <50:1.

#### *4.1.3.6 Subsurface soil processes – plant uptake and denitrification.*

The interception of diffuse nutrient pollution by riparian zones is recognition of the natural function of nutrient uptake and removal by riparian forest and vegetation along waterway corridors and river terraces (Naiman & Decamps, 1989). Soluble nitrate and P enters these zones by both surface run-off and subsurface flow from higher ground, and they also move parallel to the watercourse within the riparian zone (Naiman & Decamps, 1989). Removal before it has a chance to enter the waterway itself occurs by plant uptake into the bankside and terrace vegetation, and also by microbial denitrification in the case of nitrates. The review by Cuttle *et al.* (2004) cites many cases where nitrate concentrations of 5-20 mg $L^{-1}$ in the water entering a buffer strip are reduced to <2 mg $L^{-1}$ in that leaving the zone, and the optimum width of the zone seems to be governed mainly by the soil's water holding characteristics. Although Jacobs & Gilliam (1985) state that buffer strips are most effective when the form of nitrate was primarily as run-off, Heftig & Klein (1998) also found loadings could be reduced in subsurface groundwater flow by 95 % (40 mg $L^{-1}$ reducing to <2 mg $L^{-1}$).

Riparian zones need not be particularly wide for subsurface nitrate removal to occur. Haycock & Burt (1991) found that most of the nitrate removal occurred within the first 5–8 m of a buffer strip. Groundwater nitrate levels fell by over 60% within the first 3.3 m of entering an afforested buffer strip in the USA (Schoonover & Williard, 2003), which was attributed to both uptake and denitrification during the spring, but probably only de-nitrification during the summer months. Nutrient retention of 67-94% for N and 81–97% for P, was achieved by wider buffer strips of 31 and 51 m respectively in Estonia where the vegetation was a natural complex of wet meadow and grey alder species (Mander *et al*., 1999). This nutrient retention was always the case when incident concentrations were above 5 mg $L^{-1}$ N (0.15 mg $L^{-1}$ P), but could be zero or negative when low levels of 0.3 mg $L^{-1}$ were in the incoming water (Kuusemets *et al.,* 2001).

Removal of N from the riparian zone by the microbial process of denitrification can be a very effective means, but depends upon the maintenance of anaerobicity in the soil. For this reason it was found to be almost 100% effective over time for nitrate entering the 'near-stream zone' of a riparian buffer strip in the USA, but hardly at all for an 'upslope area' Pollutant removal effectiveness of buffers depends on the design of the buffer, including the use of vegetation types and the consideration given to localised topography and weather variables. Table 10 indicates some of the factors which affect buffer pollutant removal performance, whilst Table 11 provides some examples of quantitative performance results. The upslope area did not maintain anaerobic conditions, nor contain high labile carbon levels in the soil, which have also been found to govern the location of de-nitrification hot-spots (Addy *et al*., 1999). It is thought that afforestation of buffer strips is one means of providing high concentrations of labile carbon in the soil, but care must be taken to maintain an understory vegetation, that does not lead to an eroding surface (Cuttle *et al.*, 2004). Promoting denitrification however, is not necessarily beneficial. If conditions are only partially anaerobic, then increased nitrous oxide ($N_2O$) emissions can result, and indeed a study of $N_2O$ emissions from a riparian zone in the Netherlands, found significantly higher emission rates (20 kg ha$^{-}$1 $a^{-1}$ N) from afforested areas, than from grassland (2 – 4kg $ha^{-1}$ $a^{-1}$ N) (Heftig *et al*., 2003). The main process for N removal is subsurface denitrification, not nutrient uptake (Vought *et al.,* 1994).

Overall, the available information suggests buffer strips of 5-20 m will be effective, for dealing with a significant proportion of sediment based inorganic pollution, but effectiveness is limited for colloidal size particles, infrequent large storm events and steep slopes. In addition, their effectiveness for some pollutants is indirect: prevention of access to water reduces trampling of banksides by livestock and direct input of faecal coliforms to water; uncropped buffers prevent direct input and spray drift of pesticides; biodiversity value may be high, but this depends on active vegetation management. For soluble pollutants, buffer strips need to be modified to slow down the movement of subsurface water.

### *4.1.4 Climate related performance*

The influence of weather variables, seasonality and anomalous climatic events has been noted for their effect on wetland performance by many different authors. For example, Reddy *et al.* (2001) demonstrated that wetland performance can be moderately correlated with temperature, through a study of constructed wetlands that showed a reduced efficiency for N removal during cold months (37-51%) compared to warmer periods (>70%). Jing *et al.*

(2001) also found that seasonality was a key issue for constructed wetland performance, through the seasonal nature in growth of the wetland macrophytes. Regmi *et al.* (2003) also demonstrated the effect of seasonal variations on plant growth and, hence, P removal of a vegetated submerged flow wetland, with removal ranging between 27-100%.

Seasonality is a factor that will affect all forms of wetlands and buffer zones. Raisin & Mitchell (1995) also pointed to seasonality effects on performance of three natural wetlands in Australia, showing that total N and total P during winter was a net release from the wetland with greater flows resulting in greater flushings whilst, in spring and early summer, there was a net retention despite similar loadings.

Temperature is recognised as a key control on many of the internal wetland processes that govern removal rates. Hunt *et al.* (1995) noted that the rate of N removal was temperature dependent, being much higher in warm periods, and related this to the effect of temperature on denitrification.

#### *4.1.5 Dairy wastewater treatment performance*

With respect to treating animal wastewater from dairy and swine operations, many studies have indicated that wetlands demonstrate effectiveness when they are a component of a farm-wide waste management plan, but they are ineffective without pre-treatment of the wastewater (Cronk, 1996; Cooper *et al.,* 1996).

This is supported by Hunt & Poach (2001) who state that, whilst wastewater from dairy and swine operations has been successfully treated in constructed wetlands, it is essential to remove solids prior to wetland treatment to ensure long-term functionality. They conclude that constructed wetlands should be considered only as a component of total animal wastewater treatment. They further noted that anaerobic conditions in wetland sediments may limit the rate of nitrification and P removal and, where very high mass removals are necessary (such as dairy and swine wastewater), pre- or in-wetland procedures that promote oxidation are needed to increase treatment efficiency. Otherwise, they point out, at the high loading rates necessary for substantial mass removal, wetlands do not produce an effluent acceptable for discharge, thus requiring a final land application treatment (e.g. vegetative strips) (Hunt & Poach, 2001).

With the inclusion of a pre-treatment system, however, it appears that wetland systems can be very effective in the treatment of dairy wastewater. Luederitz *et al.,* (2001) showed that, with effective pre-cleaning of wastewater, constructed wetland (both vertical and horizontal flow) systems can remove more than 90% of organic load and of total N and P (Germany). However, other factors may influence a wetland's treatment efficiency of dairy wastewater. Results from a New Zealand study (Tanner *et al.,* 2005) of constructed wetlands treating grazed dairy pasture drainage indicate a number of implications for performance:

- size is an issue; constructed wetlands comprising ~1% of catchment area can markedly reduce N export via pastoral drainage, but may be net sources of ammonium-N and total P during establishment;
- performance of the wetland appeared to be affected by both establishment/maturation factors and year-to-year climatic variations of rainfall and soil water status.

Generally there is a large variation in the results of dairy and swine studies, many of which are based in the US. For example, BOD has been found to range from 53-93%, total N from 37-86% and total P from 42-83% across 9 studies alone (see Cathcart *et al*, 1994; Skarda *et al.,* 1994; Hunt *et al.,*1995; Cooper & Testa, 1997; Hermans & Pries, 1997; McCaskey & Hannah, 1997; Moore & Niswander, 1997; Reaves & DuBowy, 1997).

A further range of dairy and piggery studies in New Zealand demonstrated variations in total N removal of 1-33%; total P removal of 17-42%; and BOD of 32-67% (Tanner & Sukias, 2003). Stone *et al.* (2002) evaluated the design approaches in relation to performance of US swine wastewater wetlands and found that design procedures were reasonably accurate for N but that P removal was overestimated.

## 4.2 Pollutant–specific performance

Aside from the general performance issues, there is a range of performance issues that are pollutant specific (although some are applicable to multiple pollutants). The key performance statistics for over 70 studies of wetlands and buffer zones are provided in the performance matrix (Appendix 1). This list is not exhaustive, but illustrates some of the key removal efficiency figures in many different countries. In this context, the following sections discuss the issues surrounding such results.

### 4.2.1 Suspended solids

Wetlands and buffer zones are generally excellent sediment traps. Although input-output data miss most of the processes in the interior of a surface flow wetland, internal measurements of vertical sediment fluxes show a large cycle of deposition and re-suspension (Fennessy *et al.,* 1994). However, much of the material involved originates within the wetland, and in addition to macroscopic litter, the death of microflora and microfauna can create a fine detritus which is rich in nutrients, undergoes fairly rapid decomposition and is more susceptible to transport by water. The amounts of such materials have been found to be much greater in wastewater enhanced wetlands than corresponding natural wetlands. Moreover, planktonic productivity can impair the apparent sediment removal capability of surface flow wetland via the production of green suspended biomass (Kadlec, 1995).

Braskerud (2003) reviewed the results of seven Norwegian constructed wetlands and found that erosion and transportation processes in arable watersheds influenced the retention capability of the wetlands. Sedimentation was found to be the most important retention process and increased with runoff due to the increased input of larger aggregates. Braskerud suggested that creating shallow wetlands can optimise sedimentation when combined with increased aquatic vegetation, which serves for short particle settling distance and hindering the re-suspension of sediments under storm runoff conditions. By doing this, it was shown that P retention was twice that of deeper ponds.

Sedimentation is also a key retention process in surface runoff buffer zones. In a study of Norwegian buffer zones, Syversen (2001) found that high retention of sediment during major winter runoff and erosion could probably cause the erosion of coarser particles, which are more easily trapped in buffer zones.

### 4.2.2 Biological oxygen demand (BOD)

Treatment wetlands frequently receive large external supplies of carbon in the wastewater and are efficient users of external carbon sources, through successful reductions in BOD. Degradable carbon compounds are rapidly utilised in wetland metabolic process. At the same time, however, a variety of wetland decomposition processes produce available carbon. Ottova *et al.* (1997) found that the numbers of aerobic heterotrophic bacteria in wastewater entering horizontal subsurface flow wetlands are greater than anaerobic ones, but that anaerobic ones prevail in outflow, indicating that aerobic bacteria naturally die off due to the anaerobic conditions. Cooper *et al.* (1996) also pointed out that both groups consume organics but the faster metabolic rate of heterotrophs means that they are mainly responsible for the reduction of BOD. Insufficient oxygen supply to this group will, therefore, significantly reduce the performance of aerobic biological oxidation. However, if the oxygen supply is not limited, aerobic degradation will be governed by the amount of active organic matter available to the organisms (Cooper *et al.,* 1997).

Thus, the balance between uptake and production provides the carbon exports, and it has been found that the amounts of carbon cycled in the wetland far exceed the quantities added in wastewater (Kadlec, 1995).

Some studies suggest that unlike other pollutants, BOD removal is not temperature dependent and does not appear to be effected by seasonal variations in climate (e.g. Vymazal, 1999). This is an important finding, as it could confirm that, assuming there is some level of pre-treatment, wetlands (particularly horizontal subsurface flow wetlands) can successfully treat diluted wastewaters with very low BOD concentrations.

### *4.2.3 Nitrogen*

One particular issue for the ability of wetlands to remove N concerns the pollutant loading levels in the inflowing water. A strong relationship has been found between removal rate and loading rate for total N (Headley *et al.,* 2001), and plant efficiency for removing N has been proposed to be greatest at low level additions (Reddy *et al.,* 1987). This is a particular problem for the treatment of dairy wastewater, which typically contains very high levels of nutrients, including N. A study by Baird *et al.,* (2005) has illustrated that increased inflow N loads are likely to reduce a wetlands efficiency and thus increase the effluent outflow (Table 12)

*Table 12. Nitrogen loading rates and mass removal efficiencies for constructed wetlands treating swine wastewater in North Carolina, US. Source: Baird et al. (2005).*

| **Nitrogen Loading** | **System** | **% Mass Removal** | **Average Annual N Removal** | **Average Effluent N Concentration** |
|---|---|---|---|---|
| 3 kg/ha/day | Rush/Bulrush | 94 | 70 kg/ha/yr | 8.2 mg/l |
| | Cattail/Bulrush | 94 | | |
| 8 kg/ha/day | Rush/Bulrush | 88 | 1880 kg/ha/yr | 24.2 mg/l |
| | Cattail/Reed | 86 | | |
| 15 kg/ha/day | Rush/Bulrush | 85 | 3360 kg/ha/yr | 29.5 mg/l |
| | Cattail/Reed | 81 | | |
| 25 kg/ha/day | Rush/Bulrush | 90 | 5870 kg/ha/yr | 46.0 mg/l |
| | Cattail/Reed | 84 | | |

Nitrogen removal efficiency is one of the major issues in the application of animal wastewater to wetland systems (Fedler *et al.,* 2002). Nitrification and denitrification are the main N transformation processes that require aerobic and anaerobic conditions, respectively, for maximum transformation rates. Thus, management practices that allowed alternate dry (aerobic) and wet (anaerobic) periods in a wetland system would, potentially, prove more beneficial for maximising N removal than a continuously wet system. This, however, may increase operational and management requirements, and subsequent costs. Furthermore, the purpose of many constructed wetlands is to treat more than one pollutant – the cycle of wetting and drying may not prove as beneficial to these.

One key limitation to N removal efficiency is the effect of a wetland vegetation upon N transformation processes. Bacterial N transformation via nitrification and denitrification is supported by emergent wetland macrophytes, however some studies have indicated that plant residues of some common emergent macrophytes can significantly impede N transformation processes (Wetland Research Centre, 2005). The importance of different macrophyte species in supporting bacterial N transformations is not well known, and is further complicated by the spatial distribution of nitrification and denitrification processes themselves. A further issue is the maturity of the wetland vegetation, with suggestions that plant uptake and denitrification is more effective where wetland vegetation is well established (Crumpton *et al.,* 1993).

### *4.2.4 Phosphorus*

As a treatment technology, wetlands and buffer zones face several challenges in providing effective P removal from agricultural drainage waters (DeBusk *et al.,* 2004). At the outset, wetland area requirements for P removal are typically greater than for other agricultural drainage water constituents, such as nitrate-N and oxygen demanding substances. Indeed, the effectiveness of riparian buffers has been found to be lower for P reduction, especially during periods of very high runoff (Meals & Hopkins, 2002), and it may be plausible to equate this to the relative size of buffer zones.

Furthermore, P cycling within wetlands is complex, with exchanges between dissolved and particulate P forms occurring within a wetland on a spatial and temporal basis (DeBusk *et al.,* 2004). Thus, the design criteria must be able to reflect such complexity. The design must also accommodate the gradual accumulation of P-enriched sediments over time, as these can affect biogeochemical P removal pathways and limit the long-term removal effectiveness of treatment wetlands (DeBusk *et al.,* 2004).

Nevertheless, despite these challenges, wetlands are capable of reducing P in agricultural drainage waters to extremely low levels although this is usually only achieved with low mass P loading rates – hence wetland area requirements per unit mass of P removal can be extremely high. Performance data, in terms of P removal, suggest that constructed wetlands should be about twice the size of farmyard areas, although this is dependent on required effluent P concentrations (Harrington *et al.,* 2004).

Other performance design considerations include the siting and geology of a wetland treatment system. Gale *et al.* (1994) suggest that constructed wetlands established on mineral soils with a low degree of P saturation may be more efficient in retaining P than those constructed on natural, organic wetland sites. Liikanen *et al.* (2004) further suggest that this necessitates proper soil analyses for the characterisation of P resources and exchange properties before construction in order to examine the applicability of soil for effective P removal. This might also indicate (albeit tentatively) that constructed wetlands may perform better than natural wetlands for the retention of P.

### *4.2.5 Pesticides*

Whilst the fate and retention of nutrients and sediments in wetlands are quite well understood, the same cannot be claimed for agrochemicals (Schulz & Peall, 2001). Some research has highlighted significantly good removal efficiencies of pesticides with related chemical structures, but significantly poor removal efficiencies of some herbicides (Cheng *et al.,* 2002). Since wetlands have a high ability to retain and process material, it would seem reasonable that constructed wetlands could mitigate the impact of pesticides in agricultural runoff.

There is, however, a distinct gap in the knowledge of wetland processes for pesticides, particularly when compared to the amount of research on nutrients and suspended sediments. Whilst the transfer and removal processes for pesticide treatment are understood (see Table 13), the range of studies is relatively disparate, focusing on very specific issues and only a number of key pesticides.

Friesen-Pankratz *et al.* (2003) provided some insight into the possible role of phytoplankton in determining pesticide fate in wetlands, suggesting that constructed wetlands with high phytoplankton levels may be more efficient in treating pesticide contaminated waters. However, the study only focused on atrazine and lindane and admitted that the results were unexpected (findings indicated that high levels of pesticides had a positive effect on indicators of algal concentrations – an issue previously thought to have a deleterious impact).

*Table 13. Transfer and removal processes in wetlands that are important in mitigation of non-point-source pesticide runoff. Source: Rodgers & Dunn (1992).*

| Transfer processes | Removal processes |
|---|---|
| ● Flow | ● Volatilisation |
| ● Sorption | ● Photolysis |
| ● Solubility | ● Hydrolysis |
| ● Retention | ● Biotransformation |
| ● Infiltration | |

Several studies have shown that atrazine biodegradation can be successfully promoted in unsaturated soils by bioaugmenting with pure cultures of soil microorganisms (e.g. Mandelbaum *et al.,* 1993; Alvey & Crowley, 1996; Struthers *et al.,* 1998). Furthermore, the majority of bioaugmentation studies point to the success at degrading spill-site concentrations of atrazine rather than the dilute concentrations in runoff received in constructed wetlands (Runes *et al.,* 2001).

With regards to design-specific performance, a limited number of studies have examined pesticides in surface flow gravel wetlands, with several reporting significant pesticide removal in relatively large surface flow wetlands (e.g. Schulz & Peall, 2001; Kadlec & Hey, 1994). In a study using a gravel surface flow system, McKinlay & Kasperek (1999) concluded that microbial degradation was the dominant process for atrazine decomposition rather than plant uptake. Further studies have reported varying conclusions on the level of atrazine degradation under anaerobic conditions. Chung *et al.* (1995) showed that 50% of atrazine degraded in 38 weeks under anaerobic conditions, whilst Delaune *et al.* (1997) reported slower degradation of atrazine under anaerobic conditions compared to aerobic, supporting earlier such findings (e.g. Goswami & Green, 1971). However, Gu *et al.* (1992) found no degradation of atrazine under anaerobic conditions whilst Kearney *et al.* (1967) suggested that atrazine 'disappeared' more rapidly under anaerobic *vs* aerobic conditions. This uncertainty is further complicated by the introduction of other factors – Larsen *et al.* (2001) demonstrated that the introduction of nitrate, sulphate and carbon dioxide in a wetland soil prevented anaerobic degradation of atrazine. Stearman *et al.,* (2003) found that metolachlor and simaine removal in gravel surface flow wetland was significantly improved at lower flow rates with vegetated cells.

Further investigations have been made into the ability of natural wetland systems to effectively treat pesticides in agricultural runoff. For example, Kao *et al.* (2001), studied the ability of a natural wetland system to remove non point source atrazine from upland agricultural land. Analysis of one major storm and baseline water quality samples indicated a complete removal of the atrazine, with microcosm results suggesting that atrazine can be degraded under anaerobic conditions and the pesticide itself can serve as the N source for the growth of micro-organisms under anaerobic conditions. Entry (1999), in a study of wetland forest systems in northern Florida, found that that large amounts of N accumulating in wetlands as a result of agricultural operations, may decrease mineralisation of toxic agricultural pesticides.

Further performance studies will need to consider how pesticides are transformed by the wetlands system. Chung *et al.* (1996) studied the anaerobic microbial transformation/degradation characteristics of atrazine in the sediment of a US wetland treating sugar-mill wastewater. Their findings suggested that the biodegradation of atrazine created end products of $NH_3$ and $CO_2$. Thus, some consideration should be given to the potential for higher levels of ammonium-N and carbon dioxide actually being created within wetlands themselves.

### *4.2.6 Pathogens*

The removal or inactivation of enteric pathogens from manure and animal wastewater is important for human health (e.g. Salmonella, *Cryptosporidium, E Coli*). Some research has indicated that enteric microbe removal efficiency in constructed wetlands can be effected by changes in a number of factors, including:

- Hydraulic loading rate and resultant hydraulic residence time (e.g. Tanner *et al.,* 1995a).
- Presence of vegetation (e.g. Soto *et al.,* 1999).
- Wetland design type (e.g. Kadlec & Knight, 1996).

Many wetland-pathogen studies have demonstrated high levels (i.e. >90%) of enteric bacteria and virus removal (e.g. Tanner *et al.,* 1995a; Ottova *et al.,* 1997; Gerba *et al.,* 1999; Gersberg *et al.,* 1989) in constructed wetlands. However, much of this has been limited to the analysis of faecal coliforms and other bacterial indicators, which may not be indicative of the removal of other microbes, such as viruses, protozoan parasites or helminths (Hill & Sobsey, 2001). Limited information suggests that protozoan pathogens such as *Cryptosporidium parvum* may be less effectively removed than enteric bacteria and viruses in wetland systems (Gerba at al., 1999). However, helminth ova has been shown to be removed by 80-90% in subsurface flow wetlands (Stott *et al.,* 1997).

Hill & Sobsey (2001) highlight that loading rate is an important variable to consider when designing wetland treatment systems to remove pathogens from wastewater. The presence of vegetation in the studied subsurface flow wetland also significantly improved the removal of Salmonella, faecal coliforms and *E.*coli. Their data also indicated that subsurface flow systems achieved greater pathogen reductions than surface flow wetlands of the same size and loading rate. Gerba *et al.* (1999) have also illustrated how different types of wetland treatments can be effective at removing some pathogens more than others. Their findings indicated that smaller microorganisms (such as coliphage, total and faecal coliform) were removed more efficiently in subsurface flow wetlands, whilst a duckweed covered pond was more efficient at removing large microorganisms such as *Cryptosporidium*. It was suggested that, in order to achieve highest treatment level, a combination of techniques is necessary – a proposition that has been supported by Karpiscak *et al.* (2001). This study demonstrated how different pathogens responded to constructed wetland treatment as part of a multi-component facility (Table 14). Their results support other suggestions that (a) wetlands cannot treat all pollutants with equal success and (b) that wetland treatment is often much more successful as part of either a tiered system or a complex multi-component system.

As the data indicates, the wetland cells achieved small reductions for most of the parameters, with the exception of total coliforms. They were most effective in the removal of coliphage and enterococci. The authors suggest that further improvements in wetland components of treatment systems need to be considered to improve overall capability in comparison to other more conventional methods.

*Table 14. Percentage reduction in indicator bacteria and pathogens in a multi-component dairy wastewater treatment system. After Karpiscak et al. (2001).*

| **Parameter** | **Solids separators** | **Anaerobic lagoons** | **Aerobic ponds** | **Wetland cells** | **Cumulative removal** |
|---|---|---|---|---|---|
| Total coliform | 23.3 | 98.4 | 91.8 | +20.4 | 99.87 |
| Faecal coliform | 17 | 99.6 | 84.3 | 13.2 | 99.96 |
| Coliphage | 22.4 | 95.9 | 44.1 | 94.9 | 99.9 |
| Enterococci | 22 | 97.95 | 82.5 | 73.8 | 99.9 |
| *Listeria monocytogenes* | *39.1* | 98.1 | 83.2 | 31.7 | 99.86 |
| *Clostridium perfringens* | +35.6 | 52.7 | 50.6 | 19.6 | 74.6 |
| Cryptosporidium | +47.5 | 99.99 | N/a | N/a | >99.99 |

# 5 POTENTIAL ADVANTAGES

There are numerous generic advantages for using constructed wetlands and buffer zones to treat agricultural diffuse pollution. They are often less expensive to build than other treatment methods, can be built and operated simply with low operation and maintenance expenses, and require only periodic labour (Haberl *et al.,* 2003). They employ natural process, are characterised by a high buffering capacity and they can, to an extent, tolerate fluctuations in flow. Additional advantages may occur on an incidental basis, providing important benefits such as water reuse and recycling. Furthermore, treatments of this type can be designed to fit harmoniously into the landscape, creating aesthetic enhancement and an environmentally sensitive approach favourable to public perception (Haberl *et al.,* 2003).

Additionally there is the potential for long-term storage of nutrients, with pollutants retained in wetlands available to be recycled at the end of its lifecycle. For example, plant nutrients, particularly P are valuable as fertiliser and could be re-used (Culleton *et al.,* 2004; Otte, 2004).

## *5.1 Costs, operation and management*

Soft engineering techniques are usually referred to as low cost and low maintenance methods for the treatment of agricultural wastewater. With respect to the more conventional treatment systems, this is most likely to be true, particularly where the design, construction and maintenance are effective.

### *5.1.1 Costs*

Whilst wetland construction is the most significant cost, the subsequent operating and maintenance costs are relatively small due to the elimination of water supply and power requirements (unless additional irrigation or pumping is required) and the small, infrequent demand for labour. Some studies have highlighted high costs involved in maintenance (e.g. Marnhull reedbeds, Nuttall *et al.,* 1997), but these are often found to reflect a lack of proper system management, or inaccuracies in the initial design.

One example of operating costs for a UK reedbed treatment system is the Severn Trent Water subsurface flow reedbeds, which are estimated to require only 68 minutes of labour. Whilst this is not specifically a treatment of agricultural wastewater, the example does highlight the relative expense when compared with conventional treatments estimated at 4-6 person-hours per week (Nutall *et al.*, 1997).

Aside from the economic costs of wetlands, some attempts have been made to provide an ecological cost for the services provided by wetland systems. By comparing the costs and effectiveness of wetland treatments and conventional treatments, some authors suggest that a calculation of monetary value would be helpful for raising awareness of wetland related issues as well as aiding management decisions (e.g. Constanza *et al.,* 1997; Mitsch & Gosselink, 2000). Generally, wetland valuation is not a simple or easy task, yet where attempts have been made, wetlands appear to rate very highly (Otte, 2004).

The cost of buffer zones is less easy to assess, as they are often installed by farmers and, in terms of delivering water quality increments, there is often insufficient evidence to determine cost-effectiveness. However, a UK buffer zone study has suggested that buffer zones may result in a reduction in farm net income due to the switch of agricultural land to buffer zones (Leeds-Harrison *et al.,* 1996). However, this was found to be influenced by the ratio of buffer area(s) and the catchment served, as well as the intensity of the land-use prior to the buffer installation. Under CAP reform, there is also opportunity to receive payment for buffer strips.

### 5.1.2 Operation and management

The level of management required to keep a wetland operational is particularly dependent upon the design type, vegetation used and the intended pollutant to be removed. Whilst many constructed wetlands require little more maintenance than an annual harvest, others may need to undergo a removal of accumulated sediments or even a chemical treatment – this is particularly the case where the primary aim is P removal. However, a number of operational issues need to be accounted for, including the provision of pre-treatment facilities, sludge and accumulated sediments management and, most importantly, vegetation management.

Vegetation management is particularly important during the initial stages of the wetland whilst the vegetation is establishing. There have been some cited cases of poor vegetation establishment, most usually attributed to inaccurate design, a misinterpretation of the wetlands intended function, or simply poor management (Kadlec, 1995).

Reedbeds and buffer zones are particularly notable for requiring high levels of management during both establishment and aftercare. However, several management techniques are available to facilitate and improve management practices, such as water management, reed cutting and harvesting, sediment removal and pest management (Klapproth & Johnson, 2001; DeBusk *et al.,* 2004).

#### 5.1.2.1 Reedbed management

Reedbeds entail a particular set of questions about management depending upon the primary and secondary functions asked of them. The primary function is assumed to be the amelioration of pollutants in wastewater or diffuse pollutants in run-off from a catchment. The management in this case may be little more than periodic cutting of vegetation to remove nutrients from the site and the maintain the site as reedbed, rather than allowing its progression through to fen-carr following the build up of organic material.

The decision about how often to cut can more often be left to depend upon the secondary function. For instance, sites maintained to enhance their wildlife value may favour a long period of 3-15 years between cutting, just so as to allow a more open reedbed, the ingress of more diverse plant species and setting up of territories for certain bird species (Hawke & José, 1996). On the other hand, if reeds are to be cut for thatching purposes, short-term cutting regimes of only 1 or 2 years are favoured, to maintain a high density of stems (*c.*200 stems $m^{-2}$), butts of about 7 mm diameter and lengths of around 2 m (Hawke & José, 1996). At larger sites that may be managed for wildlife and even have resident Bitterns, the paradox is that both open water areas and areas of higher density new stems after cutting are prime features to encourage the increase of this rare bird species (Tyler *et al.,* 1998). A mixed pattern of management is, therefore, optimum at larger sites of several hectares.

## 5.2 Environmental benefits and wildlife value

Wetlands offer wide-ranging environmental benefits, which include water quality upgrading, water conservation, water recycling and re-use, habitat creation and restoration, and protection of downstream ecosystems, as well as commercial benefits such as aquaculture and energy production from harvested reeds. Table 15 summarises the key environmental benefits provided by both natural and constructed wetland systems. One particular example of the broad benefits within wetland systems is the effect of wetland vegetation, which can provide environmental benefits aside from wastewater control such as:

- aesthetics (a primary benefit compared to a simple soil or gravel filter and adds to ecological appeal);
- odour control (a secondary benefit as it acts as a natural odour biofilter, thus making it possible to position relatively close to the community it is to serve);

- insect control (surface plant/litter mass also limits the development of nuisance insects, such as mosquitoes and gnats by adsorbing the wastewater into litter mass and over-shadowing any open water (Wood, 1995).

*Table 15. Summary of environmental benefits. Source: Nuttall et al. (1997); Knight (1992).*

| | | | |
|---|---|---|---|
| Resource protection (habitat restoration and creation) and protection of water resources | Catchment management (water quality upgrading and re-use, land use management) | Water conservation and recycling | Energy conservation |
| Provision of a scientific and educational resource | Provision of recreation, amenity and passive enjoyment | Provision of commercial benefits | Maintenance and amelioration of biodiversity |
| Protection of downstream ecosystems | Reduction in pollution | Strengthening of environmental responsibility | Photosynthetic production and secondary production of fauna, food chain and habitat diversity |

However, the extent to which wetlands can play a role in nature conservation, and even whether they should play a role, given that they can be centres of pollution, are debatable. There are four main constraining features of any intended wetland site that will influence any wildlife objective that may also be incorporated into its function. According to Worrall *et al.* (1997) these are:

- Size of the wetland.
- Structural diversity as a habitat.
- Biological stresses imposed by the nature of the influent (pollution).
- Design features of the wetland, especially surface *vs* subsurface flow characteristics.

One of the most visible and widely studied benefits is the wildlife value that wetlands and buffer zones can provide. Reedbeds in particular are considered essential in wildlife terms, particularly in relation to the habitat provision for the Bittern in the UK which is included on the Red Data list (Tyler *et al.,* 1998). Natural reedbeds across the UK predominantly have designated Habitat Action Plans, as part of the UK Biodiversity Action Plan, and are considered to be amongst the most important habitats for birds in the UK and support a range of distinctive breeding birds (English Nature, 2005). Furthermore, they provide habitats for a range of invertebrates, including Red Data invertebrates such as the Red Leopard Moth and Rove Beetle (English Nature, 2005).

Size of the planned reedbed is of great importance in deciding its value for wildlife. Although Bitterns may only breed successfully on sites of the order of 10-25 ha in size (Worrall *et al*., 1997), even small areas of 0.25 ha typical of constructed wetland reedbeds can support a healthy and diverse specialised wildlife community, that may indeed be an addition to the wider area. Hawke & José (1996) list the species in Table 16 as those that may be supported by a small dry reedbed of 0.25 ha.

Under the UK BAP, there is an objective to create 1,200ha of new reedbed by 2010 (English Nature, 2005). However, this would primarily be on land of low nature conservation interest, and hence may only provide incidental benefits for wildlife. Moreover, the quantitative magnitude of wildlife functions will likely be highly variable between wetlands. Thus, the potential for constructed wetlands to further support such wildlife requires further investigation, particularly as their primary purpose is to treat diffuse pollution. An example of how an area of low nature conservation interest can be turned into valuable wetland wildlife habitats is the “Eye brook set-aside wetland habitat plan at the Allerton Trust estate at

Loddington (Boatman *et al.,* 1994). Here, an area of couch dominated set-aside grass in riparian buffer zones along the Eye brook, was reseeded and to a mixture of 'wet grassland', reedbeds and willow coppice, and managed by a new grazing regime.

*Table 16. Wildlife that may be supported by a 0.25 ha reedbed.*

| Numbers | Species |
|---|---|
| < 4 pairs | Reed Warblers (breeding) |
| 1-2 pairs | Reed Buntings (breeding) |
| 1 pair | Sedge Warblers (breeding) |
| 1 pair | Wrens (breeding) |
| 1 pair | Moorhens (feeding – possibly nesting) |
| ? | Bearded Tits (winter feeding) |
| ? | Harvest mice (breeding) |
| ? | Short-tailed Voles (feeding – possibly breeding) |
| ??? | Wide variety of invertebrates in food chain of above species |

In the long-term, it is conceivable that most buffer zones may also offer significant wildlife benefits, through the protection of riparian habitats by minimising sediment influx, reducing nitrate concentrations and possibly through the reduction of direct pesticide inputs (Leeds-Harrison *et al.,* 1996). However, as buffer zones are designed to be compact and distributed across a wide area, the habitat provision may also be slight.

### *5.3 Diversification*

Many natural wetlands in the UK are important areas for tourism, for example the Norfolk and Suffolk Broads, Somerset Levels and Moors, Lake District, Insh Marshes and Loch Lomond (Water Policy Team, 2005). Providing significant amenity value, they provide opportunities for recreation and general outdoor activities. Tourism plays a significant role in supporting many rural economies, so maintaining a diverse wetland landscape is important not only for its biodiversity, but also for the local community.

The application of constructed wetlands and buffer zones for the provision of tourism and recreation services may perhaps not be as significant. In the UK at least, the use of soft engineering techniques for agricultural wastewater treatment is in its early stages and not on a large scale. Furthermore, the actual size of constructed wetlands may not be enough to attract or support diverse activities, whereas many natural wetland areas will likely have the whole 'package' (i.e. designated as country parks, public parking, gift shop/café, information centre, presence of wardens). However, with an increasing need for many farmers to diversify, offering constructed wetland and buffer zone systems as part of a wider diversification package may add extra value. For example, the visual aesthetics associated with these systems could be part of the publicity for other attractions such as bed and breakfast/self-catering accommodation, working farm education and entertainment, leisure pursuits, novel livestock attractions and so on (see SAC, 2005; MFEP, 2005). However, it would be necessary to ensure that the wetland systems in particular are visually attractive.

#### *5.3.1 Biomass energy generation*

Further opportunities for diversification may be through the provision of alternative energy, primarily *via* biomass production from wetland vegetation. This would benefit particular wetland systems and buffer zones that require harvesting and cropping as part of the design maintenance. However, some consideration would need to be given to the choice of vegetation and its suitability for both wastewater treatment and biomass energy. Although no use of the dominant reedbed species of *Phragmites australis* as a biomass feedstock for power generation is known of by the authors, there is no reason why it should not be used. There is at least one example of a created wetland designed to grow another common biomass crop, namely willow (Gregerson & Brix, 2001). In this case, the wetland system uses effluent from domestic sewage, to provide both water and nutrients to an artificially raised bed of willow (*Salix alba*) and no effluent from the site occurs at all, the water in the system being transpired by the crop. The willow is cut every three years only to remove excess nutrients and contaminant heavy metals from the system, but the authors (Gregerson & Brix, 2001) do acknowledge the potential biomass power usage (crops for which are also harvested on a three year basis).

Buffer zones may be particularly suited to the growing of crops for biomass power generation, as many different types of trees, shrubs and plants can be used to form a buffer, whereas in wetlands the vegetation has specific requirements. Moreover, a key to creating buffer zones is to employ a vegetation community which will become rapidly established, and crops such as willow, poplar hybrids and switchgrass not only establish quickly, but are also very suitable for biomass energy production (Schultz, 1996). Another biomass crops that has great potential in buffer zones is *Miscanthus,* which grows to over 2 m high, but requires no fertilising and re-grows from rhizomes every year for over 20 years. Although no experimentation of its use as a vegetative strip or buffer zone crop is known of, its use as a buffer to aerial spray drift has been considered (Nixon – *pers. comm.)*. It has, however, been shown to be an excellent ameliorant of heavy metals in sewage sludge, concentrating them in the rhizome (Nixon – *pers. comm.*) and grows well using landfill leachate as a water supply, absorbing excess salts and nutrients in both shoots and rhizomes (Nixon, 2001). It, therefore, lends itself to buffer zone planting where it would maintain a permanent barrier to surface particulate movement (with careful management) and an absorptive sink for nutrients in both soluble and particulate form.

Wetlands may also have further potential to produce alternative energy through the anaerobic digestion process that treats wastewater. Biological degradation of organic waste produces a mixture of methane and carbon dioxide which can potentially be used as fuel (Waste Research Station, 2005). This technique is commonly employed at landfill sites and could be applicable to constructed wetlands but at a much smaller scale. The production of heat and electricity would require specialised equipment and could only provide small scale, on–site energy. However, the possibility of working in a local co-operative could potentially provide an integrated wastewater approach at a catchment scale, as well as the possible provision of energy to a local community.

#### *5.3.2 Thatching reeds*

Another potential crop application of constructed wetlands that has so far not been explored, may indeed be the traditional one of using the reeds for thatching material. Reedbeds as soft engineering solutions to water pollution problems use the same species as that used traditionally by thatchers from wild reedbeds, namely *Phragmites australis*, though it is often called ‘Norfolk Reed’ in the industry (reflecting its traditional industrial origins in East Anglia). Although most of the thatching in the UK was predominantly wheat straw, this declined significantly after the second world war, and further changes in the stem length of wheat as dwarf varieties became popular in the 1970s, meant that this supply of material has all but disappeared. The continued popularity of thatched roofs in certain sections of English society has, however, meant an increase in demand for suitable materials in recent years and Norfolk Reed has once again become a sought after commodity (Anon., 2005). The current demand is such that British commercial reedbeds cannot meet it, and a large proportion of the thatching needs are met by imported reed from Europe (Anon., 2005). There are considerable difficulties in creating the necessary links and infrastructure to the market for supply from outside of traditional areas, but there may be a niche in the market for quality reeds from new reedbeds. Commercial beds are typically 1-2 ha in size, and growers often hold 3-10 beds (Anon., 2005), so it is conceivable that the harvest from several constructed wetlands and reedbeds within a catchment could become a commercial proposition.

# 6 POTENTIAL LIMITATIONS

## *6.1 According to Wetland type*

### *6.1.1 Natural wetlands*

A number of issues for natural wetlands illustrate the potential limitations of their use for wastewater treatment. Primarily, these are concerned with the negative impacts on the wetland vegetation and wildlife. A particular problem is the degradation of natural wetlands through changes in vegetation, which could result from increased nutrient inputs (Kadlec & Bevis, 1990). This may particularly be the case for oligotrophic wetlands containing plants adapted to limited amounts of nutrients, and any intention to use natural wetland systems for treatment purposes would need to establish the trophic status of the existing vegetation (Blackwell *et al.,* 2002). Any change in vegetation will likely hold consequences for the wildlife it supports, and numerous authors note the potential threats to biodiversity of habitat change and also the introduction of increased pollutants (e.g. English Nature, 2005; George, 1992; Tyler *et al.,* 1998). Because many wetland sites have a protected status for conservation, it is likely therefore, that their suitability for wastewater treatment is limited.

A further limitation is that as they are natural systems, compared to constructed wetlands which are designed to meet a purpose their capacity to remove pollutants from runoff will have a natural threshold. Therefore, during extreme rainfall events, the capacity to cope with high levels of runoff may fail and little pollutant removal may occur (Blackwell *et al.,* 2002). This could not only lead to soil erosion and changes in vegetation, but it also poses a significant risk to the water body it would usually protect. Most wetlands in the UK are also unlikely to be sustainable in the long-term because of potential threats in their wider catchments such as abstraction and low river flows. Many sites already have often severely modified hydrology (Benstead, 2000), and the high cost of management that would be involved in continuing operational treatment may render them unsuitable for wastewater purposes.

### *6.1.2 Constructed wetlands*

One of the most important limitations for constructed wetlands is that there are limits to what can flow into them. For example, they cannot be used for the runoff effluent from out-wintering pads, as the BOD concentration are considered too high (McGarrigle, 2004). High nutrient levels, especially total ammonium-N, and high BOD are of particular concern because wetland vegetation cannot tolerate extremely high concentrations in wastewater (no greater than100-200 mg $L^{-1}$ for total ammonium-N) (Skarda *et al.,* 1994). This necessitates the use of pre-treatment to reduce concentrations within wastewaters (Prantner *et al.,* 2001).

A further problem is that farmers may, unwittingly or otherwise, misuse wetlands by putting everything from slurries, silage and other effluents into them, resulting in unacceptable discharges as the wetlands fail to cope (McGarrigle, 2004). This necessitates, at the very least, some level of knowledge provision and education by interested parties/authorities, and at the most, the introduction of an approved standard or certification for all wetlands installed.

A future problem may occur as a wetland matures and when it reaches the end of its lifespan (15-20 years), where it could be laden with P laden rotting vegetation and nutrient rich silts (McGarrigle, 2004). Tanner *et al.* (1998) illustrated this problem with a study of organic matter accumulation, which, over time, can be substantial and significantly reduce hydraulic retention time and the capacity of the wetland to retain suspended sediments and nutrients (Tanner & Sukias, 1995). The study found that around 50-60% of the accumulation occurred within the gravel substratum, the remainder forming surface sludge often exceeding 50 mm, whilst mean wastewater retention time decreased to ~50%.

Numerous other design-related limitations may occur with constructed wetlands (e.g. land area requirements, inaccurate designs etc), but one important point is that constructed wetlands cannot replace natural ones. Constructed wetlands will differ in soil composition and have a lower biodiversity. Because the efficiency of ecosystem services typically increases with increasing biodiversity (Callaway *et al.,* 2003), constructed wetlands may, therefore, be less efficient and stable compared to natural wetlands (Otte, 2004).

### *6.1.3 Reedbeds*

Although reed will generally grow well in poor water quality, there have been a number of cases of reedswamp decline as a consequence of high levels of pollutants (Crook *et al.,*1983). Furthermore, other flora and fauna may suffer. As a result, there is often a need for further measures to reduce or prevent high levels of pollutants from entering into a reedbed system, adding costs associated with increased maintenance of the system.
The nature of reedbeds means that usually large areas of land are required and the lead-in time for reed establishment (typically 3 years) may initially imbalance the cost:benefit ratio (Hudson, 1992). Thus, the use of the reedbed method may be unsuitable for many farms, unless intended to be used at the catchment scale.

Natural reedbeds are recognised as important for biodiversity, with many in the UK designated SSSIs/ASSIs, Wetlands of International Importance (Ramsar Convention) and SPAs (EC Birds Directive). Several of the UK's larger reedbeds are managed as NNRs by English Nature and the Countryside Council for Wales, and as reserves of the RSPB and County Wildlife Trusts (English Nature, 2005). Therefore, it is questionable whether or not they can provide a useful contribution to the control of agricultural diffuse pollution, being a primary function for habitat and biodiversity conservation. It has been demonstrated that pollution of freshwater supplies to reedbeds poses serious problems for biodiversity; toxic chemicals may lead to loss of fish and amphibian prey for key species, accumulation of pollutants in the food chain and eutrophication could lead to a loss of habitat through reed decline. This has been recorded in the Broadland areas of Eastern England, where increased N and P inputs are thought to have been one of the major causes for marginal reedswamp loss, which are key breeding and feeding areas for Bitterns (George, 1992). Eutrophication is also likely to accelerate the rate of successional change in reedbeds, increasing the management needs to maintain the reed system (Tyler *et al.,* 1998).

### *6.1.4 Buffer zones*

The most obvious limited of buffer zones is that they will not protect water from diffuse pollution in every instance, and their success is closely linked to the hydrological setting of a particular catchment (Leeds-Harrison *et al.,* 1996). Furthermore, the two principle agricultural pollutants, N and P, move by different routes and would, therefore, require a range of management methods. Buffer zones for nitrate control require that the subsurface flow passes through the root zone with sufficient retention time for plant uptake and nitrification to occur. Therefore, an ideal would be for the buffer to be waterlogged and situated in the riparian zone in order to receive as much runoff as possible. However, as P primarily moves in particulate form via soil erosion, the siting of a P buffer would need to reflect field topography and hedge/track/gateway positions. It is evident that, unlike wetland techniques, the creation of 'hybrid' buffer zones to simultaneously treat N and P is not possible. Therefore, they should be considered as part of a catchment wide strategy for diffuse pollution control, ideally using a zoned approach whereby different buffer zones are situated specifically to treat either N or P (Leeds-Harrison *et al.,* 1996).

## 6.2 Specific Limitations

### 6.2.1 Life span

The longevity of constructed wetlands is of potential concern (Drizo *et al.,* 2002). It is difficult to accurately predict longevity because most systems are at most a few decades old. One way to address this is to look at natural wetlands. Estimates vary, but it is thought that as long as a wetland is large enough relative to its loading rates, its life expectancy should be sufficient to make constructed wetlands viable long-term alternatives for traditional methods of wastewater treatment (Otte, 2004). The life expectancy of reedbeds is around 20 years (Green & Upton, 1994) to 100 years (Hudson, 1992), although accurate management practices (e.g. rotational systems of two or more beds) could greatly improve this.

### 6.2.2 Design

Numerous studies point to the fact that wetland systems are much more effective as combined systems with pre-treatment and in tiered form. One study that illustrates this effectively was conducted by Moir *et al.* (2001) looking at the treatment of dairy washings in Scotland. The study used pre-treatment with and initial horizontal flow reedbed feeding into 3 sets of vertical flow reedbeds in a 4:4:2 formation, with the output then re-circulated. Results showed the systematic breakdown of pollutants as they went through the system (see Table 17). Results also indicated that some pollutants respond better/faster than others and this suggests that, if constructed wetlands are to be designed to treat multi-pollutants, then this level of complexity must be considered. A further study by Rivera *et al.* (1995) showed that the support medium used in the design also needs consideration.

*Table 17. Reedbed treatment system average results (mg $L^{-1}$) for pre-treated dairy washings, March-December 2000. Source: Moir et al. (2001).*

| Sample location | SS | $BOD_5$ | $NH_3$ | $NO_3$ | $PO_4$ |
|---|---|---|---|---|---|
| HF Inlet | 64 | 77 | 1.0 | 9.0 | 26.1 |
| HF Outlet | 13 | 23 | 1.2 | 6.3 | 16.5 |
| VF 1-4 Outlet | 13 | 15 | 0.7 | 7.0 | 16.6 |
| VF 5-8 Outlet | 13 | 13 | 0.5 | 6.6 | 15.5 |
| VF 9-10 Outlet | 13 | 13 | 0.4 | 7.2 | 14.3 |

*Table 18. Removal rates of total bacteria and faecal coliforms from wastewater in the UK by 4 consecutive reedbeds. Source: Rivera et al. (1995).*

| | Type of Reedbed | Total Pathogen Removal % | Faecal pathogen removal % |
|---|---|---|---|
| Summer | Soil reedbed | 98.9 | 99.7 |
| | Soil without reeds | 97.8 | 92.1 |
| | Gravel reedbed | 99.7 | 99.6 |
| | Gravel without reeds | 99.8 | 99.8 |
| Winter | Soil reedbed | 34.1 | 95.0 |
| | Soil without reeds | 36.6 | 74.9 |
| | Gravel reedbed | 65.1 | 87.1 |
| | Gravel without reeds | 66.9 | 99.9 |

### *6.2.3 Climate variables*

Table 18 above also illustrates the effects of seasonality on wetland performance, through its impact on wetland vegetation. This is further supported by Dunne *et al.* (2005) during their study of soluble reactive P in an integrated CW in Ireland. They found that despite no seasonal variation in SRP (Soluble Reactive Phosphate) total input, there was distinct seasonal variation in wetland SRP output rates. Retention of spring, summer and autumn was similar with rates above 80%. However, for winter, when the wetland generally discharged the highest levels of output rates, SRP retention was only 5% and in some instances released P. These results were suggested to reflect the high levels of rainfall experienced at the time, which may distort the true processes. However, what this does illustrate, if this is the case, is the impact of extreme storm events and/or unseasonably high levels of rainfall on a wetlands performance.

Bere *et al.* (1995) also showed that varying weather conditions could influence influent strengths due to varying levels of agricultural practices. In particular, the warm summer of 1995 was shown to relate to very high levels of waste strength over a prolonged period. The authors suggest that high loadings can be effectively contained without long-term harm to the wetland system, although this is only in relation to one weather event, and there is a need to consider the consequences if such conditions became more frequent and/or prolonged. Borin *et al.* (2001), for example, showed the influence of particularly above normal conditions in a surface flow wetland receiving agricultural wastewaters. Whilst N performance was particularly good, in the latter half of the 1 year study period the reducing capacity of the wetland was lower by about 85%, which was suggested to be related to above-average rainfall causing flooding in SFW from field drainage immediately after slurry application.

Rushton & Bahk (2001) also identified the impacts of extreme weather events in the US in relation to an unseasonable amount of rainfall induced by an El Nino phase and the succeeding dry La Nina year. This weather phenomena directly influenced the performance of the wetland – about 90% of all the pollutant loads for toxic metals entered the ponds during 5 El Nino storms and this contributed to a greater percentage reduction (>90% compared to ~60% in the following year) of metals during the first year as higher pollutant loads are often more easily reduced (Rushton & Bahk, 2001). Furthermore, total suspended solids, total organic N concentrations increased from the inflow to outflow during the subsequent year – possibly related to the conversely dry conditions. Another important factor illustrated by this study is that rainfall can have a significant input into treatment systems – in this instance, rainfall directly falling onto the pond accounted for 26% of the hydrologic inputs and 50% of all the ammonium loads to the pond. Whilst the UK is unlikely to directly experience El Nino induced weather, similar climatic phenomena – the North Atlantic Oscillation – does influence rainfall, particularly in winter. Thus, considerations must be given to the design of constructed wetlands in order for them to be able to cope with variable interannual conditions (Clemence, 2005, *pers comm.*).

# 7 DISCUSSION

## 7.1 *Agricultural pollution – the challenges*

There is a range of different potential pollutants on farms that can be lost to water:

- Concentrated point sources – potent (slurry, silage effluent), and their safe *storage* is controlled by other regulations (e.g. SSAFO regulations). This does not cover their *use/disposal*, where land application is the preferred option.
- Diffuse sources of nutrients, sediment, agrochemicals and pathogens – by their very nature, diffuse sources are difficult to isolate and treat.
- Dilute point sources (e.g. dirty water from farmyards, parlour washings, etc.) – need to be contained. Usually disposed of/used by land application. Pesticide washings from pesticide handling operations on hard surfaces could also be considered a point source.

The question underlying this report is whether soft engineering options can be used to manage these potential pollutants.

We have identified wetlands and buffer strips as potential management options.

## 7.2 *'Soft engineering' – the options*

Most of the activity to control agricultural pollution relates to adopting best practices as exemplified in Codes of Good Agricultural Practice. These try to reduce losses entering water courses, for example by adopting better fertiliser and manure management: reducing application rates or better timing.

By necessity, Water Companies have to adopt 'end of pipe' solutions (i.e. water treatment plants for nitrate and pesticides), which is a recognition that tackling diffuse pollution at source has not yet been successful.

Soft engineering options possibly offer an alternative, or complementary, method for tackling pollution. This report has focused on two main approaches:

- Wetlands – natural or constructed (including reedbeds).
- Buffer strips.

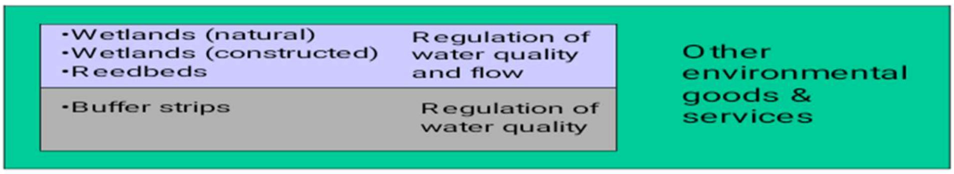


*Figure 1. Representation of the function of different soft engineering options.*

The report shows that these approaches are quite different from farm management practices that aim to control source of pollutants, such as fertiliser recommendation systems. Buffers affect transport of pollutants (by intercepting them). In some ways, however, wetlands could be considered an 'end of pipe' solution. Their main differences to other 'end of pipe' approaches is that they:

- Rely on natural processes.
- Can be incorporated into the landscape.
- Offer other environmental goods and services (Figure 1).

#### *7.2.1 Natural wetlands*

Shallow, permanently flooded or wet marshy ground populated with macroyphytic vascular plants (i.e. reeds) are known to trap and hold large amounts of solids, particulates and dissolved constituents of waters that pass through them (Forbes *et al.*, 2004). Wetlands have an inherent ability to filter/degrade potential contaminants (Table 19).

*Table 19. Percentage removal of several pollutants from secondary effluent in Natural Wetlands. Source: United States EPA (1988), cited in Forbes et al. (2004).*

| Pollutant | % Removal |
|---|---|
| BOD | 70-96 |
| Suspended solids (SS) | 60-90 |
| Nitrogen | 40-90 |
| Phosphorous | Seasonal |

This review shows that wetlands, as a functioning biological system, clean water by a mix of physical and biological mechanisms, which include:
- Plant uptake
- Adsorption
- Sediment deposition/retention
- Microbial degradation
- Chemical precipitation
- Natural die-off or predation (pathogens)
- Gaseous losses

Natural wetlands take many forms – reedbeds, grazing marshes, fens and lowland raised bogs, for example. Although all systems may regulate water quality to some degree, it is the reedbed systems that are seen as having the greatest potential to act as a treatment system.

However – and this is important – the primary function of wetland systems in the UK and Europe is now primarily provision of biodiversity. Most wetlands are under conservation designation, which may compromise their function to clean water. In fact, many would benefit from clean water entering the system, because pollutants can compromise the biodiversity of the ecosystem.

While wetlands play a role in reducing pollutant levels of inflowing water, they also require protection as water resources. The USEPA states that the use of natural wetlands for water quality treatment for either point or non point pollution sources is inappropriate.

#### *7.2.2 Constructed wetlands*

Constructed wetlands are man-made wetlands designed to mimic the action of natural wetland systems (Forbes *et al.*, 2004). Designs of constructed wetlands can vary but basically involve water being channelled into a series of man-made ponds with an impermeable synthetic liner or clay base, filled with either the original soil from the site or with selected substrates (normally sands and gravels) and aquatic plants.

Constructed wetlands are not a new concept and research has been ongoing for several decades (Forbes *et al.*, 2004). Our report shows that they have been primarily been used for point source treatment – sewage, industrial effluents, storm water and drainage water. Hawke & Jose (1996) suggest that they cannot be used to treat agricultural diffuse pollution such as 'fertiliser run-off'. We suggest that this is primarily a problem of containing such a diffuse source, unless novel approaches are used (see later), or unless large, extensive natural wetlands are used to treat the river systems.

However, constructed wetlands are being used as alternatives to traditional methods of farm waste storage and treatment, trying to deal with dilute farm effluents from manure, silage and dairy parlour washings and general farmyard wastewaters. Because these wastes have to be contained during production, we consider these to be point, rather than diffuse, sources of pollution. Nevertheless, alternative approaches to dealing with these materials would be to spread them to land, thus risking more diffuse losses.

The literature suggests that no single design type can deal with the wide range of potential pollutants. Consequently, systems tend to be set up with different elements to filter, allow sedimentation, etc. These are known as integrated systems.

#### *7.2.3 Buffer strips*

As with natural and constructed wetlands, there are many types of buffer strip/zones that can function in a catchment, varying with width vegetation cover and management. They protect water from diffuse pollution through a number of mechanisms:

- Acting as a physical barrier to prevent sediment and sediment bound contaminants from entering the stream.
- Increasing the retention time of sediment bound contaminants to allow degradation or utilisation by vegetation to occur.
- Maximising the uptake of nitrate by the vegetation in the buffer strip.
- Maximising the potential for denitrification within the buffer strip.

They can also act as a physical barrier, reducing the likelihood of direct spreading of manure, fertiliser or pesticide into a surface watercourse. In addition they offer some degree of protection from air borne pollutants, such as spray drift, reaching water sources.

Their importance and use is likely to increase, as the use of buffer strips is an option in UK's agri-environment schemes.

### ***7.3 General evidence for effectiveness***

The literature shows that soft engineering systems can be effective in improving water quality. However, they are not foolproof – and this has perhaps limited their uptake in agricultural situations.

#### *7.3.1 Wetlands*

Water quality processes in natural wetlands are much more challenging to study than those in constructed systems (WSWM, 2005). One main reason is that their water sources, rainfall and runoff, are climatically driven, making them highly variable hydrologically. It is also frequently a challenge to quantify all of the input sources and output paths. As a result, researchers tend to use differing approaches to study different systems, making their results more difficult to compare than those for the more controlled environments of constructed wetlands.

Treatment efficiencies measured in natural wetlands have proven to be more widely variable than those in constructed systems, probably due only in part to differences in experimental methods, and more so to the diversity in natural system structure, function, and historical loading trends.

Research does show that wetlands can provide a low maintenance method for reducing the input of nutrients and other agriculturally derived pollutants to surface and shallow groundwaters (Blackwell, *et al.,* 2002).

However, their performance is variable depending, as shown in Appendix 1, because the action of wetlands depends on many factors:

- nature, quantity and timing (seasonality) of pollution inputs;
- wetland size;
- wetland ability to interact with pollutants;
- the appropriateness and accuracy of design;
- management practices and;
- availability of site-specific data for wetland design.

A constructed wetland also seldom functions as a true sink for nutrients and other contaminants and is more likely to have multiple roles as source, sink and transformer depending on location, season and environmental factors. But, importantly, one particular performance issue is that most agricultural effluents are much too strong to be economically treated using reedbed treatment systems alone – hence the need for integrated systems.

Where failure does occur, this has often resulted from over-optimistic design or unsuitable configurations, poor management or the rigid application of a particular design principle to a wastewater problem. Further research suggests that the different design types may also be limited in their treatment performance for particular pollutants.

Where constructed wetlands are being used for point source control and are discharging directly into a controlled water, then a discharge consent is required from the Environment Agency. This will set strict limits on discharges and will require some level of monitoring. Therefore, this serves as a safety check on the efficacy of a system. Whether such discharge consents would be required if constructed wetlands were installed to address diffuse pollution is a grey area. However, by the letter of the law, and depending on the Environment Agency's interpretation, it is probable that a discharge consent would be required.

Seasonal considerations also need to be taken into account when investigating the effectiveness of constructed wetlands.

The majority of studies collated in Appendix 1 show some improvement – but whether this is sufficient on its own, or whether wetlands just form a part of the solution depends on the target being set.

### *7.3.2 Buffer strips*

Riparian buffer zones can only be effective for local pollution problems and have to be carefully sited within catchments according to local hydrological flows and pathways. Their effectiveness is relative to the nature of the run-off, the type of sediment and pollutant loading. In many intensive agricultural areas, the land is under-drained, which means that soluble pollutants such nitrate N simply by-pass buffer zone methods. In such cases, buffer zones will only be fully effective when operates in concert with in-stream methods to remove N, before the main waterways. In addition, the problem of groundwater flow (carrying a high nitrate loading) may also make buffer zones ineffective in some intensively farmed areas.

Hefting (2003) found that the groundwater level was important in such cases, in determining which process dominated in the N dynamics of the buffer zone. If the ground water was at less than 10 cm depth ammonification dominated and ammonium ions accumulated in the surface soil, whilst at depths between 10 and 30 cm, denitrification dominated. At depths lower than 30 cm nitrification dominated the system and the end product of nitrate was more freely available. Hefting (2003) also found that N was taken up and retained better in a wooded zone compared with grassland, and that the main mechanism for removal in woodland was denitrification. The drawback to this was that nitrous oxide flux was also highest under woodland.

Nitrogen removal by vegetation, and trees in particular, is only effective during the summer growing season, and leakage from the zone can occur during winter. Uptake occurs mainly in the spring, and during summer months denitrification becomes the dominant removal process, providing anaerobic conditions can be maintained. For these conditions, buffer zones of between 5 - 20 m are necessary and can only be operated successfully on slopes of less than about 10 % where there is no significant channel flow occurring.

The removal of P in buffer zones is less efficient than N, and can also be a problem in constructed wetlands. This is due to complex recycling taking place between vegetation and soluble and insoluble forms of P, and the build up of P in sediments. Accumulated sediments are at risk of being scoured out from wetland during extreme storm events and causing pollution downstream. This is especially a problem in older sites and constitutes a limit on wetlands sustainability.

## *7.4 Effectiveness on specific nutrients of constructed wetlands*

### *7.4.1 Wastewater treatment*

With respect to treating animal wastewater from dairy and pig units, many studies have indicated that wetlands demonstrate effectiveness when they are a component of a farm-wide waste management plan, but they are ineffective without pre-treatment of the wastewater (Cronk, 1996; Cooper *et al.,* 1996).

For example, they cannot be used for the runoff effluent from out-wintering pads, as the BOD concentration are considered too high (McGarrigle, 2004). High nutrient levels, especially total ammonium-N, and high BOD are of particular concern because wetland vegetation cannot tolerate extremely high concentrations in wastewater (no greater than100-200 mg $L^{-1}$ for total ammonium-N) (Skarda *et al.,* 1994). This necessitates the use of pre-treatment to reduce concentrations within wastewaters (Prantner *et al.,* 2001). Also, although reed will generally grow well in poor water quality, there have been a number of cases of reedswamp decline as a consequence of high levels of pollutants (Crook *et al.,*1983).

With the inclusion of a pre-treatment system, however, it appears that wetland systems can be very effective in the treatment of dairy wastewater.

### *7.4.2 Individual pollutants*

This review of the literature (e.g. Appendix 1) shows a wide range of performances in reducing levels of pollutants, mainly because of the wide range of operating conditions during the projects. Clearly input loadings and residence times (as well as season, as described earlier) have impacts on effectiveness. However, we are able to draw broad conclusions about the efficacy against individual pollutants.

**BOD** – generally very effective treatment for effluents will elevated BOD, further supported by the review of Forbes *et al*. (2004). Effective even with short residence times.

**Sediment** – constructed wetlands are good sediment traps. This will be improved by including a series of settlement ponds in an integrated system. Sediment trapping is better in shallow (compared with deep) wetlands, and is better in situations of high run-off (larger aggregates that settle quickly).

**Nitrogen** – N is removed from the water by a range of mechanisms, and can be lost completely from the system if denitrified. There is a strong relationship between efficacy of N removal and loading rate, with systems operating more effectively at lower loadings. Clearly, this has implications for treating, e.g. wastewater, because this can have elevated levels of N. This could make treated water unsuitable for direct discharge to watercourses (Hunt & Poach, 2001). For example, a re-analysis of the data presented in Table 12 (Baird *et al*., 2005) suggests the efficiency of removal does not decrease substantially at higher daily N loadings (80-90%, Table 12), but the effluent concentration is higher, the greater the N input (Figure 2). If we were to use constructed wetlands to treat diffuse pollution then, clearly, efficacy would depend on daily N loadings and residence time.

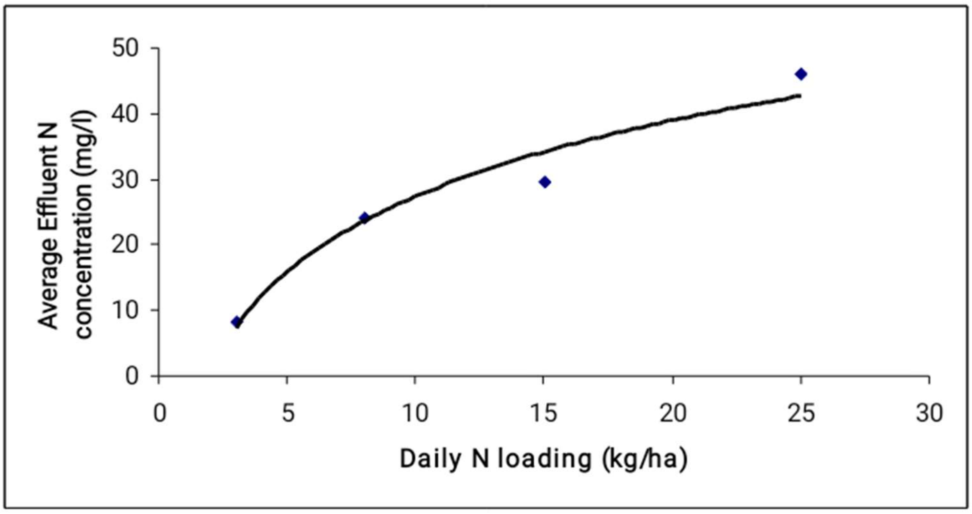


*Figure 2. Relationship between daily N load entering constructed wetlands and the average effluent N concentration post treatment.*

**Phosphorus** – Forbes *et al*. (2004) state that P investigations and constructed wetlands is a contentious issue. Our report confirms this. Phosphorus cycling within wetlands is complex, and there is a gradual accumulation of P-enriched sediment with time, which can limit the long-term removal efficiency. As with N, efficacy is improved at low input rates.

**Pesticides** – Our survey of the literature identifies this as an area of further work, with limited studies on a disparate range of chemicals. However, reports generally show good levels of breakdown of agrochemicals, particularly those based on N compounds.

**Pathogens** – Many studies have demonstrated high levels of pathogen removal. Much of the work has been limited to detection of faecal coliforms or other bacterial indicators. Longer residence times are preferred.

Our review suggests that the effectiveness of constructed wetlands in removing pesticides and pathogens is generally an under-researched area and the mechanisms for both are poorly understood. For instance, it is not known whether aerobic or anaerobic systems are best for dealing with pesticides (it seems highly specific to the chemical species involved), though it does seem that subsurface flow wetlands are generally more efficient at removing FIOs than surface flow designs. However, only bacterial organisms have been studied at any length and very little is known of the fate of viruses, protozoan parasites and helminths.

### 7.5 Constructed wetland for diffuse pollution control

A hypothesis of this review is that perhaps (constructed) wetlands, as well as dealing with dilute point sources, could be adapted to deal with diffuse sources by careful placement in the landscape.

This review shows that America has the most experience of using constructed wetlands for dealing with non-point sources, both 'storm water run-off' (urban flows) and agricultural diffuse pollution. A further advantage is that they can regulate storm flows and also contribute to groundwater recharge.

The use of constructed wetlands for storm water treatment is still an emerging technology, hence there are no widely accepted design criteria (WSWM, 2005). However, certain general design considerations do exist:

- Decrease water inflow velocities to provide opportunity for initial sediment deposition.
- Maximise the hydraulic residence time and the distribution of inflows over the treatment area. Avoiding designs that allow for hydraulic short-circuiting.
- Emergent macrophytic vegetation plays a key role. Thus, it is important to design for a substantial native emergent vegetative component.
- Anaerobic sediment conditions should be ensured to allow for long-term burial of organic matter and P.
- A controlled rate of discharge.

For agricultural NPS runoff, researchers in Maine have developed and tested a multi-step constructed "nutrient/sediment control system" for cropland runoff (Reed *et al*., 1994). Components of the system include, in sequence: a sediment basin; a level spreader, which disperses flows across an overland grass filter; the filter, which provides fine sediment and nutrient removal; an emergent marsh that grades into open water, primarily for nutrient removal; and a final grass filter to capture solids and nutrients in the form of algae that is produced in the pond. These systems have removed 90-100% of suspended solids, 85-100% of total P, 90-100% of BOD, and 80-90% of total N from potato field runoff in northern Maine (Hammer, 1992).

We can conclude, therefore, that use of constructed wetlands does hold possibilities for tackling diffuse agricultural pollution.

### 7.6 Other considerations

The literature suggests that there is some scope for soft engineering approaches to support other technical measures to decrease agricultural pollution. However, we also need to consider other factors such as operation and costs of these approaches.

#### *7.6.1 Cost and cost-benefit*

The literature tends to make general statements about 'low cost' and 'low maintenance costs' of constructed wetlands. However, detailed data are difficult to find, as also noted by Forbes *et al*. (2004) in their review of constructed wetlands.

Whilst construction will be the most significant cost, the subsequent operating and maintenance costs will be relatively small due to the elimination of water supply and power requirements (unless additional irrigation or pumping is required) and the small, infrequent demand for labour. Where studies have highlighted high costs involved in maintenance, these have often been because of a lack of proper system management, or inaccuracies in the initial design.

Examples of cost can be found for non-agricultural uses. The CMCH (2005) quote some costs for constructions in 1994, based on the reports of Kadlec (1995) and Reed *et al*. (1994). The cost obviously varied depending on size and site conditions. In general, larger constructed wetlands involve higher construction, installation, maintenance, and waste disposal costs:

- *Construction Costs* - Using data from municipal systems, Kadlec (1995) cites construction costs from 18 North American surface flow wetlands ranging from £4,000 to £200,000 per hectare, with a mean of £65,000. Reed *et al*. (1994) cited a range of £65,000 to £180,000 per hectare for the same type of system. NB: all 1994 costs.

- *Operations and Maintenance Costs* - Once established, the operation and maintenance costs for constructed wetlands can be lower than for alternative treatment options, generally less than £1000 per ha per year (Kadlec, 1995), including the cost of pumping, mechanical maintenance, and pest control.

Details on the size and features of these constructed wetlands were unavailable. Other reports suggest that costs are 50-90% of the alternative 'conventional' approaches for dealing with storm water run-off (i.e. urban sources of polluted water).

It is argued that the area required for a constructed wetland is relatively small compared with the source water catchment, e.g.
- 840 $m^2$ in a 22 ha catchment (Haarstad & Braskerud, 2003), Norway.
- 1200 $m^2$ in a 80 ha catchment (Blankenberg & Braskerud, 2003), Norway.
- 'small' constructed wetland serves 55 ha catchment (CERE, 2005), USA.

In very broad terms, if we take an average construction cost of £65,000 per ha (previously quoted, above) at 1994 prices, this equates to *c*. £80,000 today (assuming 2% per annum inflation). If we assume that a 0.25 ha wetland will serve 50 ha (i.e. 0.5% of the catchment area, within the 0.3-0.7% range quoted by Vinten *et al*., 2005), this works out to an investment cost of £400/ha. Annual running costs (£1000 per ha of wetland, 1994 costs, inflated to £1,220) are calculated to be *c*. £6/ha of farmed land.

These are very rough estimates, and are very dependent on the cost of construction and the area required. For example, the areas cited by Haarstad & Braskerud (2003) and Blankenberg & Braskerud (2003), above, correspond to 0.4% and 0.15% of the catchment area, respectively. Applying these areas of wetland to our same theoretical 50 ha catchment reduces the construction costs to £120-320 per ha of catchment.

Clearly, the cost of buffer strips creation and maintenance is much less, being a less-engineered solution. There is some loss of productive land, but this can now be offset by payments under UK agri-environmental schemes.

Much is spoken about the costs, but there are also benefits. Again, quantifying these has proved difficult. Also, as with many environmental projects, the question of boundaries for economic calculations is inevitably raised. Do we value all of the environmental goods (e.g. biodiversity, tourism, etc.) that might arise from an improved rural environment? And if so, how?

Perhaps this is outside the scope of this review, and it would be better to focus on the direct impacts on water quality. Even so, this is not simple. Again, drawing on non-agricultural examples, it has been suggest that costs are 50-90% of the alternative 'conventional' approaches for dealing with these urban sources of polluted water. However, this may not be a fair comparison, because alternative 'conventional' approaches for urban storm water management are quite different to approaches that might be taken for agricultural diffuse pollution control.

After reviewing the literature, our thoughts are that the costs and benefits will depend very much on individual circumstances – and how the soft engineering approach is used. Cost-benefit analysis is perhaps less of an issue for buffer strips. The real challenge lies with constructed wetlands. Conventionally, these have been used to treat point sources – and a fairly straightforward cost benefit analysis can be made for the individual circumstances. Costing the benefit of constructed wetlands for diffuse pollution control, however, is complex, and warrants a separate project.

#### *7.6.2 Longevity*

This review reports that longevity of *constructed wetlands* is of potential concern (for example, see Drizo *et al.,* 2002). A problem is that the concept of constructed wetlands is relatively new and so most systems are, at most, a few decades old.

One approach is to learn from natural wetlands. Estimates vary, but it is thought that as long as a wetland is large enough relative to its loading rates, its life expectancy should be sufficient to make constructed wetlands viable long-term alternatives for traditional methods of wastewater treatment (Otte, 2004). The life expectancy of reedbeds is around 20 years (Green & Upton, 1994) to 100 years (Hudson, 1992), although accurate management practices (e.g. rotational systems of two or more beds) could greatly improve this.

So, much comes down to the original design of the system. The key seems to be that the wetland must be sufficiently large relative to the source land/water that accumulation of enriched sediment, for example, does not become a problem. Another risk is sediment build-up affecting the hydrology of the system. Here, there is so simple way of assessing whether the hydrology is functioning correctly (P. Fogg, pers *comm.*) – this is something that perhaps needs further development work.

At the end of its useful life, it will be necessary to remove the sediment, enriched with nutrients and organic matter. This operation will create a risk, both for disposal of the material and, also contamination, of the receiving waters with pollutants (if the process is not managed correctly). There are some suggestions (e.g. Culleton *et al.,* 2004; Otte, 2004) that the cleaned out material is a valuable fertiliser source. Previously, landfill has been an option (P. Fogg, pers *comm.*), though land spreading (with care) would clearly be a more sustainable solution.

The longevity of *buffer strips* will also depend on many of the factors discussed above. Factors that affect their ability to function as a physical and biological barrier will compromise their useful life. This might include decline in vegetation or sediment build up or general compaction from machinery or animals. It is interesting to note that most of the literature that discusses this issue is American. The use of buffer strips or zones is more widespread there than in the UK.

#### *7.6.3 Other environmental services*

We have already discussed some of these issues, above. Soft engineering solutions bring biodiversity benefits. In fact, most of the natural wetlands are managed and valued for biodiversity, rather than diffuse pollution control.

The application of constructed wetlands and buffer zones for the provision of tourism and recreation services may perhaps not be as significant, however. In the UK at least, the use of soft engineering techniques for agricultural wastewater treatment is in its early stages and not on a large scale. Furthermore, the actual size of constructed wetlands may not be enough to attract or support diverse activities, whereas many natural wetland areas will likely have the whole ‘package’ (i.e. designated as country parks, public parking, gift shop/café, information centre, presence of wardens).

However, with an increasing need for many farmers to diversify, offering constructed wetland and buffer zone systems as part of a wider diversification package may add extra value.  Our review also suggests other diversification possibilities, though further research is required:
- Energy crops
- Thatching reeds

# 8 CONCLUSIONS

## *8.1 General conclusions*

1. Natural wetlands, constructed wetlands and buffer zones provide potential for regulating water quality. This review shows that many factors affect their efficacy and, consequently, no system is foolproof. They are, particularly, affected by the level of input. Where constructed wetlands are used to treat dilute point sources of agricultural effluent, this can be taken into account in the design. However, where soft engineering systems are used to deal with diffuse sources, driven predominantly by rainfall events, then these variations in flow and inputs (including seasonality) are more difficult to deal with.

2. Natural wetlands play an important role in the landscape. They are a valued asset, as is noted by the fact that most are designated as protected areas. They are valued and managed mainly for biodiversity benefits. Their role in water management is more likely to be that of regulating flow rather than water quality protection. We noted the suggestion that using for water treatment may conflict with management for biodiversity.

3. Buffer strips are a 'low tech' soft engineering solution. They may appear an attractive option in the UK now that they can attract payments under agri-environmental schemes. Their effectiveness relies on their ability to act as a physical barrier (e.g. stopping sediment) and as a functioning biological system (e.g. microbial transformation and/or breakdown of pollutants, plant uptake). Much is known about their operation and factors that affect their usefulness. However, there are few measurements under UK conditions to prove their worth at the catchment scale.

*The remainder of these conclusions focus on mainly constructed wetlands…*

4. Constructed wetlands are now a very widespread and quite well understood form of soft engineering for pollution mitigation. Most, if not all, water utilities have examples of constructed wetlands in operation to mitigate point-source pollution from a variety of domestic and industrial origins.

5. They are inappropriate for some agricultural effluents – and are better with dilute sources, such as dirty water/parlour washings. There has been very little work, as far as we could see, looking at constructed wetlands to deal with pesticide washings.

6. Systems work better when they have several elements, such as sediment ponds integrated with wetlands.

7. Although originally designed to deal with point sources, there is scope for use to treat diffuse sources of pollution. As far as we could see, most experience of this is in the USA. Construction costs and design to intercept diffuse sources are issues – these are discussed later.

8. When discharging into controlled waters, outputs from the constructed wetlands would need a discharge consent.

9. As with any mitigation method, no system is foolproof, and the level of risk that can be tolerated will determine if constructed wetlands (or buffer strips) are deemed an acceptable approach to pollution control.

10. There is little information on their longevity, but 15-20 years is suggested. Much will depend on the levels of inputs. Removing enriched sediment at the end of their useful

life poses an environmental risk, either by contamination of the receiving waterbody or during disposal of the sediment.

11. In summary, these are the potential advantages of constructed wetlands:
    - Generally effective in decreasing pollutant loads, but much depends on operational conditions.
    - Low cost when the construction price is spread over the catchment area.
    - Generally, low operating costs.
    - Little labour required once operating.
    - They use natural processes and have a high buffering capacity.
    - They 'fit' into the landscape.
    - They are perceived as 'environmentally sensitive' and are generally approved of by the general public.
    - They may have the potential for the secondary use of products, such as thatching reed or biomass energy crops.

12. In summary, these are the potential disadvantages of constructed wetlands:
    - Variable performance, depending on many factors.
    - Potentially substantial construction costs, but these vary considerably depending on site requirements.
    - Extreme weather events may overload the system catastrophically.
    - There are limits to the level of contamination they can cope with, especially with regard to BOD and nitrate concentrations.
    - For some point-sources (livestock holdings) they often need pre-treatment measures in addition.
    - After a working life of 15 – 20 years the system may be laden with nutrient rich silt and organic sediments that are difficult to dispose of.
    - Integrated systems need designing on an individual catchment basis to deal with the local pollution problem.

    - They usually increase the wildlife biodiversity of the local area providing key niches for several important species.

### *8.2 Implications for UK diffuse pollution policy*

#### *8.2.1 Potential for reducing the concentration and loading of agricultural pollutants in agricultural surface water catchments?*

We have to consider point and diffuse sources separately, both contribute to the potential pollution load of a catchment.

**Point sources**

Silage effluent, stored slurry and fuel oil are potential point sources of pollution, if stores fail. Storage is covered by separate regulations (SSAFO).

Silage effluent and slurry are too potent to effectively dealt with by constructed wetlands – they cannot bring N (and P) concentrations down to acceptable levels for discharge.

Constructed wetlands are more appropriate for dealing with dilute point sources such as dirty water/parlour washings. Whether this approach is better than the more conventional approach of, e.g. adding to the slurry store for land spreading or using low rate irrigation systems, will depend on individual farm circumstances. Certainly, given the cost of constructing a wetland treatment system, this is unlikely to be the first choice for a farm –

more likely to be the last resort when all other options have been explored and deemed infeasible or more expensive.

One area that has not been fully explored is their potential use to treat pesticide washings, similar in approach to biobeds.

Buffer strips are inappropriate for point source pollution.

**Diffuse sources**

Buffer strips are increasingly put forward as part of the solution for tackling diffuse pollution from agriculture. Therefore, here, we have focused on the potential for constructed wetlands.

Diffuse sources of pollution should generally produce concentrations of N and P that will be less than those in dirty water. In theory, then, wetlands potentially have a role to play in diffuse pollution mitigation. We have previously argued that natural wetlands are an important landscape feature, but their primary function is biodiversity provision. The question then arises whether we can use constructed wetland as a tool for diffuse pollution mitigation?

In terms of effectiveness of the approach of constructed wetlands, we have already commented on the variable performance of a system because of its dependence on many factors. The literature is full of reports summarising ‘typical’ values of effectiveness (e.g. Table 20). However, behind these averages are wide ranges and the decision on the potential usefulness of the approach comes down to risk.

*Table 20. Summary of contaminant removal efficiency in treatment wetlands, based on the North American Wetland Treatment System Database. Source: DeBusk (1999).*

| Contaminant | Outflow concentration mg $L^{-1}$ | Concentration reduction % | Mass removal kg/ha/day | Removal efficiency % |
|---|---|---|---|---|
| Total suspended solids) | 13 | 72 | 11.9 | 71 |
| Biochemical oxygen demand ($BOD_5$) | 8.1 | 73 | 7.5 | 68 |
| Nitrate (as N) | 2.1 | 62 | 0.54 | 55 |
| Total ammonia (as N) | 2.4 | 52 | 0.38 | 26 |
| Total nitrogen | 4.5 | 53 | 1.5 | 51 |
| Orthophosphate (as P) | 1.1 | 37 | 0.12 | 41 |
| Total phosphorus | 1.7 | 56 | 0.22 | 31 |

It is interesting to note that the conclusions drawn by Forbes *et al.* (2004) were that constructed wetlands were too unreliable to be put forward as a solution, whereas our interpretation of data collated in this review is more sympathetic. However, they were assessing them for point sources, with strict discharge requirements. We are considering them as a diffuse source treatment system.

Our arguments for advocating their use include:

- They generally show some reduction in pollution levels though we accept, in some circumstances, they may act as a source. Improved management and design will minimise this risk.
- They are part of the solution, not the entire solution. They are part of a multi-barrier approach as advocated by the WFD. Therefore, we would expect measures that focus on source and transport controls also to be employed.

This review shows, however, that there is further investigative work to be done on now to construct and manage the systems, as well as how to integrate them into the landscape. Questions also arise over their long-term benefit, the argument being that most monitoring projects tend to last 1-3 years. There are also questions about how to effectively monitor such systems, particularly when using for diffuse pollution control.

There are many parallels with the acceptability of biobeds for treating pesticide washings. This too is a biological system. Even though experiments have shown them to be an effective tool, there is some concern from policy makers over their adoption on farm – will they be managed correctly, will they remain effective, could they make the problem worse by concentrating the problem into small areas? Further demonstration, investigation and use in the catchment (will careful monitoring) will help to assess these risks.

### *8.2.2 How the systems may need to be adapted to meet the requirements of agriculture?*

Because buffer strips are a well-accepted tool for diffuse pollution mitigation, and easy to incorporate into the landscape, here we have focused on constructed wetlands.

Use for point source control, again, is a relatively straightforward design issue.

But can we use constructed systems for diffuse pollution control? The answer from the literature is clearly yes, with systems in USA and Europe.

However, it is essential to properly plan where to place the artificial wetlands, as their effectiveness is dependent on hydrology (i.e., they should be covered by water most of the year and have a sufficient retention time to allow them to treat specific pollutants). They also need to be placed where they intercept the diffuse pollution run-off. Furthermore, it is important to ensure that the wetlands themselves are not sources of potential pollutants, such as P.

The area required is relatively small – 0.1-0.4% of the catchment area, for example. The approach can be relatively low cost, depending on the engineering requirements of the installation and whether the cost can be spread over the area of 'po'luting land'. Systems may need to contain different elements to treat water with a range of potential contaminants. One example of a system in the USA is shown in Table 21.

*Table 21. Example of a five-element constructed wetland system for the treatment of non-point-source pollution. Source: CERE (2005).*

| Cell Type | Function | Water Depth | Vegetation |
|---|---|---|---|
| Sediment Pond (1 pond) | • Collect organic matter, larger sediment particles, provide water storage and regulate flows through CWS. | Variable, depending on inflow and outflow to other cells | only along banks to prevent erosion |
| Primary Filter (8 cells) | • Remove fine sediments and dissolved N.<br>• Inflow from sediment pond outflow to any cell listed below | <10 cm | *Carex nebrascensis*<br>*Juncus balticus*<br>*Eleocharis palustris*,<br>*Schoenoplectus maritimus* |
| Shallow Wetland (4 cells) | • Remove nitrates, ammonia, bacteria.<br>• Inflow from sediment pond, primary filter, or other shallow wetland cells. | 10-50 cm | *Typha latifolia*,<br>*Schoenoplectus acutus* |

| | | | |
|---|---|---|---|
| Deep Water Pond (1 pond) | • Remove dissolved nutrients, fine sediments.<br>• Inflow from any cell listed above. Outflow to final filter. | 100-300 cm | Sago pondweed (floating) |
| Final Filter (1 cell) | • Remove dissolved nutrients<br>• Inflow from any cell. Outflow is directly to American Falls Reservoir. | Variable | *Carex nebrascensis*<br>*Juncus balticus*<br>*Eleocharis palustris*,<br>*Schoenoplectus maritimus,*<br>*Typha latifolia*,<br>*Schoenoplectus acutus* |

Two issues would need to be resolved before they could be used in the UK:

- Ownership – who is liable and who is responsible for the maintenance. Presumably, this would need to be spread across the catchment from where water is draining.
- The need for licensing and discharge consents.

#### *8.2.3 Other diversification opportunities?*

There are clearly other benefits to be obtained from the use of soft engineering systems, as summarised in Table 15 and summarised here:

- Resource protection (habitat restoration and creation) and protection of water resources
- Provision of a scientific and educational resource
- Protection of downstream ecosystems
- Catchment management (water quality upgrading and re-use, land use management)
- Provision of recreation, amenity and passive enjoyment
- Reduction in pollution
- Water conservation and recycling
- Provision of commercial benefits
- Strengthening of environmental responsibility
- Energy conservation
- Maintenance and amelioration of biodiversity
- Photosynthetic production and secondary production of fauna, food chain and habitat diversity

## 9 FUTURE WORK

There are few existing facilities in the UK that could be used for monitoring purposes (Appendix II). We therefore suggest the following actions:

1. Study tour to examine the facilities in the USA. The technology is well developed for application to diffuse pollution control, and a lot could be learnt from visiting example installations.

2. Initiate experimental facilities in England.

3. Initiate long-term monitoring programmes.

4. More detailed information on costs and benefits.

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# APPENDIX I

## (PERFORMANCE MATRIX OF WETLANDS / REEDBEDS USED TO TREAT AGRICULTURAL POLLUTANTS IN ENGLAND)

**NB ‘Type’: N = Natural, C= Constructed, E = Experimental, M = Modelled**

| Source | Location | Type | Design/Type | Study Period | | | Effectiveness | |
|---|---|---|---|---|---|---|---|---|
| | | | | | Suspended Solids | BOD | Nitrogen | Phosp |
| Green & Upton (1994) | UK | C | 5 constructed reed beds for tertiary treatment of domestic sewage | … | Reductions of up to 87% | Reductions of 77-88% | Reductions of 7-97% | Reductions |
| Hudson (1992) | Germany | C | Reedbed in operation since 1974 - thus well estbalished | … | … | 99% reductions | 70% reductions | 56% reduc |
| Sebilo *et al*. (2003) | ? | N | .. | … | … | … | Up to 60% of $NO_3$ exported from agri soils is eliminated either by riparian wetland or in stream wetlands | . |
| Braskerud | Norway | C | 4 different CWs | … | Average retention between 45-75% | … | … | Average re between 2 |
| Prantner *et al*. (2001) | | E | Pre-treatment by soil infiltration | … | … | … | Pretreatment by soil infiltration removed 93% of $NH_4$-N, wetland systems removed 94% of the remainder | Total P lev decreased infiltration a wetlands b 84% respe |
| Cheng *et al*. (2002) | … | E | Vertical flow CW | … | … | … | … | . |
| Runes *et al*. (2001) | USA | C | … | … | … | … | … | . |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Schulz & Peall (2001) | South Africa | C | Vegetated surface flow constructed wetland | 5 months | Retained 15% of suspended solids during dry conditions, 78% in wet conditions | … | Retained 70% of nitrate during dry conditions, 84% in wet conditions | Retained 5 orthophosp conditions, wet conditi |
| Peterjohn & Correll (1984) | US | N | … | … | … | … | 89% removal efficiency | . |
| Cooper (1990) | New Zealand | N | … | … | … | … | 56-100% removal efficiency | . |
| Kadlec & Knight (1996) | US | C | … | … | … | … | 80-90% removal efficiency | . |
| Pommel & Dorioz (1995) | Switzerland & France | N | … | … | … | … | … | 65% remov efficiency |
| Mander *et al.* (1991) | Estonia | N | … | … | … | … | … | 27-88% re efficiency |
| Jenssen *et al.* (1993) | Norway | C | … | … | … | … | … | 98% remov efficiency |
| Braskerud (1994) | Norway | C | … | … | … | … | … | 20-42% re efficiency |
| Mantovi *et al.* (2003) | Italy | C | Subsurface Flow Reedbeds | 2 years monitoring | >90% removal | … | 50% removal | 60% remov |
| Reed & Brown (1992) | | | … | … | … | … | … | . |
| Sun *et al.* (1998) | UK | E | 3-tier constructed downflow reed bed with recirculation | 3 months | About 95% removed | Average BOD reduced by 99.1% from 464 to 4mgl-1 | NH4-N reduced by 61.7% from 253.2 to 96.9mgl-1 | . |
| Cooper *et al.* (1996) | UK | E | 3-tier constructed downflow reed bed | 2 years | Maximum SS reductions of 72% | Maximum $BOD_5$ reduction of 97% | Average reduction of $NH_4$-N of 48% | Average orthophosp reductions |
| McCaskey & Hannah (1996) | US | E | Two-tiered surface flow constructed wetland with pretreatment holding pond | 57 months | Total suspended solid reduction of 87% | Total BOD reduction of 89% | Total N reduction of 84%, ammonia-N reduction of 85% | Total P red 76% |
| Serodes & Normand (1999) | Canada | E | Surface flow constructed wetlands | 1st year of operation | … | … | … | Removal o to 50% for and 63% fo |
| Liikanen *et al.* (2004) | Finland | C | … | 2 years | … | … | … | Total P loa diminished and the dis reactive P 49% |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Reddy *et al*. 1999 | … | … | 83 various wetlands | … | … | … | … | Average P 58% |
| Cooper *et al*. (1987) | US | E | Analysis of riparian buffer areas in 2 watersheds | … | Estimates indicated that 84-90% of sediment was retained by riparian areas | … | … | … |
| Tanner *et al*. (1995) | New Zealand | E | 4 pairs of planted and unplanted gravel bed wetlands | 20 months | Mean annual SS removals of 75-85% recorded irrespective of loading rate. No marked difference between planted and unplanted | Total BOD mean mass removal increased from 50-80% during first 12 months of monitoring, and were highly dependent of influent loadings. No marked difference between planted and unplanted | … | … |
| Tanner *et al*. (1995) | New Zealand | E | 4 pairs of planted and unplanted gravel bed wetlands | 20 months | … | … | As theoretical retention teims increased from 2-7 days, mean reduction of TN increased from 12-41% in unplanted and 48-75% in planted wetlands | As theoreti retention ti increased f days, mear of total pho increased f 36% for un and 37-74% planted we |
| Yates & Sheirdan (1983) | US | N | Heavily vegetated forest wetlands on US coast | 1 year | … | … | 96% of nitrate plus nitrite nitrogen was retained, utilised and transported | 37% of orth phosphoru retained, u transported |
| Abeysinghe *et al*. (1996) | Australia | E | Submerged flow cyclic aerated/unaerated biofilters for fishfarm wastewater | … | … | … | achieved complete nitrification and 40% denitrification | achieved a phosphoru |
| Sun *et al*. (1999) | UK | C | Full-scale combined tidal down-flow reed bed system | … | … | At a mean flow rate of 2.0 $m^3$/d $BOD_5$ of the influent was reduced by 97.6% | NH4-N reduced by 93.1% | … |
| Mandi *et al*. (1996) | Morocco | E | Three horizontal flow reed beds differing in length (30, 40, 50m) | … | … | … | Decrease in total N of 43% in large bed, but 23% in small bed | Decrease i for the larg only 5% fo smallest |
| Jing *et al*. (2001) | Taiwan | C | Pilot scale constructed wetland | … | … | … | Monthly average removal rate of $NH_4$-N between April and October 78-100%, after November this dropped to 16% | Monthly av removal ra orthophosp between A October 52 November dropped to |
| Coveney *et al*. (2002) | US | C | Pilot scale constructed wetland with recirculation | 29 months | Total suspended solid mass removal efficiency was 89-99% | | Soluble $NO_3$ and $NH_4$ compounds increased during passage through wetlands, particulate matter at outlet was enriched in N 2-fold compared to particles in inflow, whilst total nitrogen removal efficiency was 30-52% | Soluble rea increased c passage th wetland, pa matter at th was enrich fold compa particles in whilst total phosphoru efficiency w 67% |
| Biddlestone *et al*. (1991) | UK | C | 2-stage engineered downflow reed beds | … | … | BOD reduced from an average of 1006 mg/litre at the inlet to only 57mg/l at the exit | … | … |
| Schaafsma *et al*. (1999) | US | C | Hyrbrid of 2 settling basins, 2 cells and vegetated filter strip | | Suspended solids reduced by 96% | BOD reduced by 97% | TN reduced by 98%, ammonia 56% but a nitrate/nitrite increase of 82% | total P redu 96%, ortho 84% |
| Moir *et al*. (2003) | UK | C | Combined system accepting pre-treated dairy effluent into 1 horizontal flow reedbed feeding into 3 sets of VF reedbeds in 4;4:2 | 9 months | Reduction of suspended solids from 64 mg/l at HF inlet to 13 mg/l at final outlet | Reduction in $BOD_5$ from 77 mg/l at HF inlet to 13 mg/l at final outlet | Reduction in ammonium from 1.0 mg/l at HF inlet to 0.4 mg/l at final outlet, with reduction in $NO_3$ from 9.0 to 7.2 mg/l | Reduction from 26 mg inlet to 14 r outlet. |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | formation, with recirculation of output | | | | | |
| Rivera *et al*. (1995) | UK | C | 4 consecutive reedbeds | … | … | … | … | … |
| Dunne *et al*. (2005) | Ireland | C | Integrated Constructed Wetland | … | … | … | … | Soluble rea retention w from Feb-C 5% betwee |
| Bere *et al*. (1995) | UK | C | 3 stage complex combined HF and VF reedbed treatment system | 18 months | Removal efficiencies ranged from 80-95% | Removal efficiencies ranged from 80-95% | Removal efficiencies for $NH_4$-N ranged from 76-85% | … |
| Stearman *et al.* (2003) | US | C | Subsurface flow pilot system of 12 gravel bed cells | 2 years | … | … | … | … |
| Kadlec & Hey (1994) | US | N | Reconstructed river wetlands | … | … | … | … | … |
| Borin *et al*. (2001) | Italy | C | Surface flow vegetated wetland | 1 year | … | … | SFW received 205 kg/ha during study period and discharged only 5 kg/ha achieving nearly 100% removal | … |
| Rushton & Bahk | US | C | 2-stage detention pond to treat runoff from winter vegetables | 2 years | … | … | Organic N removal of 20-40% | … |
| Kao *et al*. (2001); Kao & Wu (2001) | US | N | Mountainous wetland site | 9 months | Removed 91% TSS | … | More than 80% N removal | 59% TP re |
| Reddy *et al*. (2001) | US | C | 6 wetland cells were constructed in a combination of ponds and masrhes | 6 months | … | … | Removal of 37-51% N during cold months, >70% N removal during warmer periods | P removal |
| Hill & Sobsey (2001) | US | C | Field scale surface flow constructed wetland treating swine wastewater and laboratory scale surface and subsurface flow constructed wetland reactors | 3 years | … | … | … | … |
| Ottova *et al*. (1997) | Czech Republic | C | 5 constructed wetlands | 2 years | … | … | … | … |
| Gerba *et al*. (1999) | US | C | Duckweed covered pond, multi-species SSF and SF wetlands | … | … | … | … | … |

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|---|---|---|---|---|---|---|---|---|
| Gersberg et al. 1989 | | C | | | … | … | … | … |
| Karpiscak et al. (2001) | US | C | Multi-component treatment system for dairy and municipla wastewater (paired solids separation, anaerobic lagoons, aerobic ponds and 8 constructed wetland cells) | 2.5 years | … | … | … | … |
| Nungesser & Chimney (2001) | US | C | Study of Everglades Nutrient Removal Project - large constructed wetlands known as Stormwater Treatment Areas to reduce P concentrations in runoff | 5 years | … | … | … | Over study ENRP reta metric tons would have entered the Everglades consistentl performanc 75% TP loa reduction. |
| Kantawani chkul *et al*. (2001) | Thailand | C | Combined vertical vegetated bed over horizontal flow sand bed with recirculation to remove nitrogen from pig farm wastewater | … | … | … | Total N removal efficiency from 71-85% in vertical flow bed, nitrate reduction of 60% in horizontal flow bed | … |
| Headley *et al*. (2001) | Australia | C | 4 subsurface horizontal flow constructed wetlands to treat plant nursery irrigation runoff with variation in hydraulic retention times (HRT) | 5 months | … | … | TN load removal of >84% for HRT of 2-5 days; >90% load removal of NH4, NO2, NO3 achieved for all HRTs | TP load rer >65% for H days; >90% removal of achieved fo |
| Sukias & Tanner (1996a) | New Zealand | C | Pre-treated SF and combined SF/SSF constructed wetlands treating piggery effluent | … | 45% reduction in SF; 76% reduction in combined SF/SSF | 53% reduction in SF; 67% reduction in combined SF/SSF | 28% reduction of TN in SF; 33% in combined SF/SSF. 16% reduction in $NH_4$--N in SF; 3% in combined SF/SSF | 25% reduc in SF; 34% combined S |
| Sukias & Tanner (1996b) | New Zealand | C | Pre-treated combined SF/SSF constructed wetland treating dairy farm effluent | … | Estimated mass removal at 56% | Estimated mass removal at 49% | estimated mass removal of TN at 16%; $NH_4$-N 19% | Estimated removal of |
| Sukias & Tanner (1993) | New Zealand | C | Pre-treated SSF constructed wetlands treating piggery effluent | … | Reductions between 40-52% | Reductions between 57-61% | TN reductions between 22-37%; $NH_4$-N reductions between 23-45% | TP reductio between 18 |
| Bruere & Donal (1997) | New Zealand | C | Pre-treated SF and combined SF/SSF constructed wetlands treating dairy farm effluent | … | Estimated mass removal between 70% for SF; 74% for SF/SSF | Estimated mass removal 35% for SF; 32% for SF/SSF | Estimated mass removal of TN 16% for SF; 17% for SF/SSF. Estimated mass removal for ammoniacal nitrogen - 3% for SF; 15% for SF/SSF | Estimated removal of for both SF SF/SSF |
| Sezerino *et al*. (2003) | Brazil | C | pre-treated VF 4-bed Constructed wetland pilot plant treating piggery effluent | 280 days | … | … | ~49% nitrification observed | $PO_4$-P rem 45% |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Braskerud (2002) | Norway | C | 7 constructed wetlands located in first and second order streams | 3-7 years | Mean annual retention of soil particles and organic particles was 45-75% and 43-67% respectively | … | Mean annual N retention of 3-15% | Mean annu retention o |
| Mitsch (1992) | US | N / C | Comparison of natural and constructed wetlands | … | … | … | … | Constructe retained 63 natural wet retained 4- (although h loading rat |
| Rodgers & Dunn (1992) | US | M | Pesticide transfer and transformation model addressing pesticide residence time and transfer half life on removal | … | … | … | … | … |
| van Oostrom & Cooper (1990) | New Zealand | C | Pilot-scale 3-stage SF and 3-stage gravel bed constructed wetland systems treating meat processing effluent | 10 months (SF) and 18 months (gravel) | … | … | N removal by all was low at equal or less than 22% | … |
| Vymazal (1990) | Czech Republic | C | Small reedbed system experimentally treating chicken manure wastewater | 3 months | Removal efficiency for suspended solids of 85% | Removal efficiency for $BOD_5$ of 80% | Removal efficiency of TN of 65% | Removal e TP of 80% |
| Geary & Moore (1999) | Australia | C | 2 pre-treated SF wetlands treating dairy parlour waters | 2 yrs | … | Mean monthly reduction of 61% | Mean monthly reduction of 43% of organic N; 26% of $NH_3$ | Mean mon reduction c TP |
| Vymazal (1999) | Czech Republic | C | Horizontal subsurface flow constructed wetlands | 5 years | … | Average treatment efficiency of 86.6% | … | … |
| Karpiscak *et al*. (1999) | US | C | Integrated wastewater facility using pretreatment solids separators, anaerobic lagoons, aerobic ponds and 8 surface cell wetlands treating dairy wastewater | 1 year | TSS removal by wetland cells 29.6% | $BOD_5$ removal by wetland cells -1.6% | TN removed by wetland cells 23.2%; Organic N 10.2%; NH4-N 40.1% | … |
| Comin *et al*. (1997) | Spain | N | 4 restored wetlands treating ricefield irrigation drainage | 1 year | … | … | Removal of 84-98% of TN | … |
| Worrall *et al*. (1997) | UK | C | Reedbed treatment system with dual objective of improving wildfowl effluent and creating a nature conservation | 2 years | Reduction around 80% | Reduction generally above 60% | … | … |
| Morris & Herbert, (1997) | UK | C | Vertical flow reed bed treating sugarbeet processing wastewater | 6 months (Sept-Feb) | 87.7% TSS removal | … | 79.5% removal of $NH_4$-N | … |
| Raisin (1996) | Australia | N | Small upland wetland receiving storm discharge and downstream nutrient loads | 2 years | … | … | Retained 23% N | Retained 3 |
| Yin & Lan (1995) | China | N | Shallow eutrophic lake with broad ecotone wetlands | 1 month, 2 successive years | … | … | TN removal 42-59% | TP remova |
| Benham & Mote (1999) | US | E | Laboratory scale constructed wetlands with combinations of organic loading rates and presence of vegetation to treat dairy lagoon effluent | … | … | … | 65-81% N removal efficiencies for all treatments | P removal variable wi efficiencies 33-9% |

| | | | | | | [redacted] | | |
|---|---|---|---|---|---|---|---|---|
| Sun *et al*. (1998) | UK | E/M | Full scale downflow reed bed system for treatment of high strength agircultural wastewater | | Removal of suspended solids 39.6% | Average removal od $BOD_5$ 74.3% | Removal or $NH_4$-N 23.1% | Removal o 34.7% |
| Regmi *et al.* (2003) | US | C | 3 vegetated and 3 non-vegetation SF constructed wetlands | 2 years | Up to 98% removal for both vegetated and non-vegetated | Up to 98% removal for both vegetated and non-vegetated | Annual average mass removal for ammonia nitrogen in vegetated beds up to 95% | Dissolved p reduction r 27-100% ir beds; 0-66 vegetated l |
| Williams *et al.* (1995) | UK | C | Tertiary and secondary gravel bed hyrdoponic constructed wetlands | 3 years | Tertiary treatment efficiency 61%; secondary 78% | Tertiary treatment efficiency 92%; secondary 93% | Tertiary efficiency for $NH_4$-N 93%; secondary 6%. Tertiary efficiency for total oxidised nitrogen 53%; secondary n/a | . |

# APPENDIX 2

## DATABASE OF WETLANDS / REEDBEDS USED TO TREAT AGRICULTURAL POLLUTANTS IN ENGLAND

Many individuals and organisations offer the design and construction of wetlands for pollution control now, and at least 16 of these list their services under the web-pages of the Constructed Wetlands Association at http://ww.constructedwetland.org

Details of one highly lauded commercially operating site, at Sheepdrove Organic farm, can be found at http://www.sheepdrove.com/article.asp?art_id=115 .

However, constucted wetlands and buffer zones are still considered largely unproven technology and several actively experimental studies are currently underway in the UK. Several of these are listed on the UK ADAPT database of research into catchment management hydrology and diffuse pollution studies, http://www.uk-adapt.org.uk .

Some of interest are extracted below:

“To find sources and pathways of suspended solids in the Gavenny Brook.” - River Usk, S.E. Wales. *Gloucester University*

“The strategic placement and design of buffering features for sediment and P in the landscape.” - River parrett, Somerset. *NSRI.*

“The National Trust: Farm Buffer Zone Options.” Loe Pool. *National Trust*.

“SOWAP – EU Life Environment, Syngenta, Leuven University, WOCAT, Hungarian Insitute of Sciences. – Porlock Vale and Rutland in the UK (also Belgium and Hungary), West Somerset and Rutland. *Allerton Trust, RSPB, national Trust, Harper Adams University, NSRI, Cranfield University, Syngenta, Pond Conservation Trust, leuven University, Hungarian Institute of Sciences, hydro-Agri., Vaderstad Ltd., Keszthley University, WOCAT, FWAG, Agronomica, NRM.*

“Potential use of willows and poplars as components of practical buffer zones – studentship.” River Avon tribuary, Bristol. *IACR, Long Ashton.*

“Minewaters.” S.E.Wales. *Neath Port Talbot Council.*

“Loch Lomond Catchment Management Plan.” Loch Lomond, Argyll and Bute. *SEPA, Scottish Water, Loch Lomond and the Trossachs National Park, Scottish Natural Heritage.*

“Investigation into nutrient and sediment input into Bassenthwaite Lake.” River Derwent, Cunbria. *ADAS Consulting ltd., University of Sheffiled.*

“Integrated Catchments model of Phosphorus (INCA-P).” Rivers, ANT, Kennet, Lugg & Wye. *EA, University of Reading.*

“Evaluation of the effectiveness of the Water Fringe Option Schem on environmental quality.” *RPS Clouston.*