A review of scientific, technical and socio-economic information about the range and application of biofilters for reducing air and water pollution from UK agriculture

Prepared as an Appendix to the Final Report on Defra Project WQ0102

November 2007
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1. Biofiltration - farm based biofilters for water and air

1.1 Introduction

Biofiltration is a means of ameliorating water and air quality by effective removal of organic and inorganic pollutants such as pesticides and nutrients, greenhouse gases and ammonia, odoriferous compounds and also, in the case of water, by removal of pathogenic organisms and by reduction in biological oxygen demand (BOD). Biofiltration has been proved to be effective in several specific instances, but uptake of methods and technologies by UK farmers has been limited.

Biofiltration utilises various combinations of biological, chemical and physical mechanisms for the removal of substances from streams of liquid or air, that are normally effluents from an industrial or agricultural process. The efficiency in decontamination of these effluent streams is determined by several factors. Biofiltration utilises biological organisms (plants, fungi, bacteria etc) to sequester or metabolise the potentially polluting substance(s) from a stream of air or water passing through or over a culture of the organism (biofilm). The biofilm may be supported on a solid matrix for the purposes of optimising surface interaction with the pollutant, the longevity of biofilm and thus, the overall efficiency of the process. This efficiency depends also on the rate of growth of the biofilm organism, which depends in turn on an adequate supply of pollutants (inorganic nutrients, carbon substrate, or both) either in the liquid/gas stream, or from an auxiliary source. Another important consideration affecting efficiency of biofiltration is the impact of the growing biofilm on the physical interaction of the flow stream with the biofilm and any solid support. Ways of biofilm removal by 'harvesting' must be incorporated into the process to prevent clogging and to maintain or periodically regenerate the efficiency of the system. Biofiltration methods can be classified primarily according to the technology and mechanisms employed (i.e. natures of any solid supporting matrix and the biofilm), application to liquid or gas phase or alternatively according to a range of typical scenarios.

The earliest use of biofiltration has been traced back to a 1923 German publication, where the use of soil based biofilters to treat hydrogen sulphide (H\textsubscript{2}S) from sewage treatment plants was discussed [Bach, 1923]. The concept was further developed in the USA [Carlson and Leiser, 1966; Pomeroy, 1957] and in parts of Europe such as Germany and the Netherlands [Ottengraf, 1986]. Biofiltration has been used in industrial processes at a range of scales for some time. For example, with the liquid phase, in the pre-treatment of drinking water [Madon, et al., 2001; Persson et al., 2005], in fish farming [Zhou, et al., 2006], in the treatment of textile factory wastewater [Chang, et al., 2002], and landfill leachates [Robertson and Anderson, 1999]; and with the gaseous phase, in the use of methane oxidising bacteria for ameliorating effluent gas from landfill sites and in odour abatement at composting plants [Jones and Banuelos, 2000; Sironi and Botta, 2001]. The principles of biofiltration are equally applicable to remediation of agricultural effluents [Oleszkiewicz, 1981] and many successful ‘proofs of principle’ and demonstration projects have been reported. These include degradation of
ammonia (NH₃) from livestock wastes [Kalingan et al., 2004] the removal of the pesticide atrazine from runoff using grass-based filters [Popov and Cornish, 2006] and the removal of nitrate from drainage waters [Dinnes, et al., 2002] and feedlot wastewaters [Zhao and Shao, 2004]. While such 'end of pipe', downstream solutions to water and air pollution are gaining favour in parts of USA [Hill and Sobsey, 2001], Asia [Sommer et al., 2005], New Zealand [Tanner et al., 2003] and Scandinavia [Gustafson, 2000] they have not yet been widely embraced and implemented by the British farming community. Recent Defra funded projects (e.g. ES0132, ES0109) have reviewed the efficacy of wetlands (natural and constructed), farm ponds and buffer strips for the amelioration of agricultural drainage waters. Their effectiveness was found to depend upon whether they are required to treat diffuse or point sources, the strength of the source, their location relative to the source and on weather patterns and hydrology in the catchment. Other projects (CC0222, CC0240) have demonstrated the usefulness of biofilters as part of anaerobic digestion of farm manures for biogas production and as 'biobeds' (PL0544) for ameliorating pesticide waste and washings.

An extensive search of international literature was made in order to produce a broad review of the range and application of biofilters, that could be used in a UK agricultural context. Sources of literature included peer reviewed journals, conference proceedings, and technical reports. The review has focussed mainly on literature produced since 1981.

1.2 Pollutants

The main pollutants of concern are ammonia (NH₃), methane (CH₄), greenhouse gases (GHG’s), volatile organic carbon compounds (VOC’s), odours, nitrate (NO₃), phosphorus (P), pesticides and pathogens. The main sources of aerial pollutants are livestock housing, collection and feeding yards, manure and slurry stores, manure and fertilizer N spreading and grazing animals. The main sources of nutrients and pollutants to water courses include silage effluent, runoff from yards, leaking storage tanks, parlour washings, surface runoff and leaching of slurry, manures, nutrients, pathogens, sediments, pesticides and pathogens.

Target areas where biofilters could be used to control airborne pollutants include livestock housing, manure and slurry stores. Target areas where biofilters could be used to intercept and control liquid borne pollutants include treatment of manures and slurries, riparian areas, and surface runoff and drainage from sloping land.

1.3 Types of biofilter

The term ‘biofilter’ has traditionally been applied to the types of constructed filters that are used in the wastewater industry for the treatment of domestic sewage, or for the screening of gaseous emissions from industrial operations. For the purpose of this review we have considered the definition of a biofilter to be any type of natural or constructed filter that employs biological processes to either immobilise, or degrade harmful pollutants and organisms
in gaseous and liquid effluents from agricultural operations, in order to prevent them from contaminating the wider environment.

1.4 Biofilters for liquids

Types of biofilter that are used in the treatment of liquids include land based filters or buffer zones; storage type filters such as detention ponds, lagoons, settlement tanks; and percolating type filters such as trickling biofilters, clinker beds, rotating arm systems, bacteria beds and filter beds.

Buffer zones and ponds
The term buffer zone is applied to an area of land that separates and protects an adjacent natural ecosystem from potential negative impacts of agricultural operations and management. Buffer zones include natural and constructed wetlands, reedbeds, short rotation forestry, and buffer strips. Buffer zones help to regulate water flow and protect water bodies form the impacts of high nutrient loading, sediments and pesticides [Blackwell, et al., 2002]. The efficiency of buffer zones and wetlands to treat point and diffuse sources of pollutants from agriculture has been extensively reviewed by Fogg et al. (2005).

Wetlands are areas of land that are saturated by surface and groundwater during part or all of the year and support an abundance of vegetation typically adapted to living in saturated soil conditions [Novotny, 2003]. The main attributes of wetlands include low operational costs, minimal reliance on machinery and energy inputs. Mechanisms for nutrient and sediment removal include shallow water to maximise the sediment to water interface, high primary productivity, accumulation of organic matter and anaerobic and aerobic sites in the sediment [Mitsch and Gosselink, 1993]. Factors affecting wetland processes, include hydrology, vegetation, temperature and the medium from which the wetland is constructed [Blackwell et al., 2002].

Although natural wetlands have shown the potential to be effective in the removal of pollutants and nutrients [Cooper, et al., 1986, Gilliam, 1994, Mitsch, 1992], they are limited in their capability to process agricultural wastewaters [Peterson, 1998] and are probably more suitable as controllers of water movement rather than water quality. In addition, they are highly valued for their habitat provision and are considered to be sites for conservation. Thus, the use of natural wetlands to control water quality may conflict with their management for biodiversity [Fogg et al., 2005].

In contrast, constructed wetlands have been specifically designed and engineered to exploit the natural processes of wetland vegetation, soils and microbial population in order to improve water quality from areas of point source pollution. They are considered a high value wildlife habitat, and have a perceived "naturalness" by the general public [Tanner et al., 2003]. Constructed wetlands have been used for the treatment of horticultural effluents [Berghage, et al., 1999], mine wastewaters [Mitsch and Wise, 1998], storm-water runoff from urban environments [Carleton et al., 2000] and for the
secondary and tertiary treatment of sewage [Cooper, 2005; Cooper and Green, 1995; Vymazal, 2001]. For example, where an effective pre-cleaning step is in place, both horizontal flow wetlands (HFW) and vertical flow wetlands (VFW) have the potential to remove up to 90% of the organic load and total N and P from sewage [Luederitz et al., 2001].

In the UK most popular design for constructed wetlands for treatment of point source pollution, is the use of reed-beds [Fogg et al., 2005]. The use of these design systems have been applied to the treatment of agricultural wastewaters [Edwards et al., 2001; Job et al., 1991; Sun et al., 1998]. The construction of reed-beds is based on two main configurations according to the directional flow of effluent through them. Horizontal flow reed beds are generally used for influents with a low contaminant concentration, and that have ideally undergone some form of pre-treatment. They are not effective at reducing NH$_4^+$ concentration, as they can become oxygen limiting which inhibits nitrification. However they help reduce the level of BOD and SS. These types of systems are often used as a tertiary treatment and for final polishing of effluents such as the discharge from a sewage treatment plant. However, they can be more effective if several lagoons are placed in series (e.g. Fig. 1).

In contrast, vertical flow reed-bed systems are more effective at reducing NH$_4$ concentration, BOD, SS and odours than horizontal flow reed-beds. The main difference between the two designs is that the reed-bed is flooded in intermittent batches, in between which, it is allowed to drain, thus, replenishing the oxygen supply. Moreover, vertical flow systems may incorporate perforated pipes which allow air to penetrate the bed. Vertical flow reed-beds are generally smaller and will cope with much stronger effluents than horizontal flow reed-beds. The two designs can be used individually or in combination.

In addition to their use as filters, wetlands can provide a source of economic return. For example, a common wetlands plant species is the common reed (Phragmites australis), which can produce up to 3550 g dry wt m$^{-2}$ yr$^{-1}$ [Kadlec and Knight, 1996] and can be harvested for thatching or weaving. Similarly, some species of willow (Salix spp.), grown as short rotation forestry (SRF), have a high capacity for nutrient uptake, help prevent soil erosion, and can be harvested for construction, as a biofuel, or compost [Borjesson, 1999; Elowson, 1999; Kuzovkina and Quigley, 2005]. The potential of Salix (S. kinuyanagi) to recover N from the effluent output of dairy oxidation ponds has been investigated in New Zealand [Roygard et al., 2001]. The study has shown that an area of 5.4 ha is required to treat a pond effluent discharge output of 7409 m$^3$ yr$^{-1}$ (604 kg N yr$^{-1}$) with a tree uptake of 150 kg N yr$^{-1}$. The authors recommend that effluent application is discontinued post harvest for at least a year to prevent leaching losses of nitrogen due to the resultant reduced evapotranspiration rate and nutrient uptake. On way to reduce leaching losses is to split the SRF into blocks with staggered annual harvests so that blocks can be rested.
Investment costs for constructed wetlands works out at about £400 ha\(^{-1}\) (assuming a 0.25ha wetland area to treat a 50 ha catchment area) with annual running costs at about £1000 ha\(^{-1}\) wetland, or £6 ha\(^{-1}\) farmed land [Fogg et al., 2005].

Figure 1. A wetland system of 3 sequential lagoons being constructed in Hokkaido, Japan to clean the whole effluent from a 300 cow dairy house

Estimates for the effective lifetime of wetlands is suggested as being 15-20 years, though this is dependent on the levels of input, decline in vegetation, sediment build up, and compaction from animals or machinery [Fogg et al., 2005].

**Buffer strips**

Buffer strips are vegetated areas of land that are used to intercept surface runoff. They are usually narrow, linear and are often situated along a field boundary or following a contour [Blackwell et al., 2002]. Vegetated filter strips operate through the processes of infiltration, to reduce water volume, and adsorption and sedimentation to reduce the pollutant load [Popov et al., 2006]. A buffer strip reduces diffuse pollution by acting as a physical barrier to drainage, which increases retention time and allows biological activity to deactivate, remove or immobilise the pollutants. Factors affecting these processes will determine the performance of the buffer. These factors include the type of plant growing in the buffer zone, the slope of the land and the degree of soil structural differentiation. A Defra (MAFF) sponsored PhD project (Macey, 1996; Macey et al., 1996) examined the efficiency of farm woodlands as nitrate sinks. In this 3-year experimental study conducted at 3
scales –farm, instrumented hillslope and replicated field plot- it was concluded that once the flow of polluted water had become highly channelled the buffer was inefficient at intercepting and removing nitrate. In other words, the biofilter had to be close-coupled to the source for optimal efficiency. Where this was the case however, 15 m of nitrogen deficient woodland could effectively buffer pulses of nitrate (75 kg N ha$^{-1}$) applied to an upslope source area subjected to surface runoff applied as artificial rainfall at 5 mm h$^{-1}$. The field-plot experiment showed that there was little difference in buffering efficiency/capacity between a range of tree sapling species and established grass swards.

Earlier experiments measuring nutrients lost in runoff from spring applied fertilisers (Scholefield and Stone, 1995) indicated that the degree of structural differentiation and resulting preferential flow of water also determines the efficiency of the soil itself as a buffer. These factors control the degree of exchange between water flowing rapidly in macropores with water held over longer periods in micropores (Williams et al., 1993) It was thus suggested that buffering capacity is determined by the macropore flux of water:micropore volume and the appropriate pollutant concentration gradient across the interfaces between macropore and micropores (Scholefield et al., 1998)

Microorganisms and aquatic flora play a major role in nutrient cycling and oxygen status of the water in pond ecosystems. The other major factors are abiotic physico-chemical processes between nutrients and pond soil and sediments. The efficiency of ponds for the retention and/or removal of nutrients from agricultural waters was recently reviewed in a Defra funded project ES1009 [Hawkins and Scholefield, 2003].The review identified that effective retention of nutrients within a pond is dependent on the dynamics of and the interaction between physical, chemical, biological and climatological parameters. The retention of available NH$_4^+$, either from fertilizer inputs or from mineralization of organic matter, occurs through immobilisation of NH$_4^+$ by phytoplankton and subsequent conversion to organic nitrogen. Nitrifiers oxidise NH$_4^+$ to NO$_3^-$ which may also be immobilised by phytoplankton and other microorganisms. Organic nitrogen in the form of waste products from the biota or from their dead bodies may eventually deposit onto the pond bottom and become part of the pond soil organic matter. Non-biological retention occurs through exchange of NH$_4^+$ with cation exchange sites on pond soil and sediment.

Gaseous losses of N take place through the denitrification of NO$_3^-$ by anaerobes living in the sediment at the bottom of the pond. Nitrogen gas produced from denitrification diffuses from the sediment into the pond water and eventually to the atmosphere and is the major N removal mechanism from aquatic systems [Busk et al., 1983; Fleischer, 1995; Horberg et al., 1991; Yan et al., 1998].

The majority of N retention is during periods of low flow in the summer. During this period, denitrification may account for as much as 50% loss of the total N retained. Annually though, denitrification accounts for 30-40% loss of total N
retained [Jansson et al., 1994] though losses of up to 90% of total N (external + internal) N load have been reported [Fleischer et al., 1997].

The primary mechanism for retention of colloidal or sediment attached P in ponds is that of sedimentation and co-precipitation with calcium carbonate (CaCO$_3$) [Istvanovics et al., 1990]. The size of the sediment particles is a major control on effective retention. Pond soil and sediments strongly absorb P and absorption increases with increasing clay content of the soil and sediment. Phytoplankton readily uptake P, and this appears to be the most important removal process of dissolved P from waters [Istvanovics et al., 1990].

A summary of the efficiencies of the various methods of pollutant removal discussed in this section, is given in Table 1 below.

Table 1 Summary of efficiencies for several method of control for water borne pollutants. Summarized from Novotny (2003)

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Nitrogen (%)</th>
<th>Phosphorus (%)</th>
<th>Sediment (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetlands</td>
<td>40-80</td>
<td>10-70</td>
<td>80-90</td>
</tr>
<tr>
<td>Filter strips</td>
<td></td>
<td>5-50</td>
<td>35-90</td>
</tr>
<tr>
<td>Detention-retention ponds</td>
<td>30</td>
<td>40</td>
<td>40-87</td>
</tr>
</tbody>
</table>

**Percolating filters**

Percolating filters are also known as trickling filters, biological filters, biofilters, clinker beds, rotating arm systems, bacteria beds and filter beds [Grant, et al., 2000]. Percolating filters typically consist of a container (up to 2m deep) filled with a medium such as stones or clinker. Often these type of filters are preceded by a holding tank or reservoir to encourage settlement of particulate matter and suspended solids.

Percolating filters are traditionally used in the large scale treatment of sewage and urban drainage at water treatment plants, or at small scale, individual domestic situations. Sewage is distributed over the surface of the medium either by a rotating arm or a tipping bucket mechanism, and effluent drains freely from the base of the filter. The raw sewage forms a thin coating on the medium which, together with an ample supply of oxygen, encourages the growth and multiplication of microorganism.

One of the most common ways of removing ammoniacal nitrogen from waters is to exploit the microbiological processes of nitrification and denitrification in the presence of an adequate supply of carbon ©. Carbon sources include sawdust, wood residues, green waste, straw has also been used to promote
the microbial immobilisation of soluble phosphorus from water bodies [Wingfield, et al., 1985]. Simple and inexpensive biofilters using PVC tanks and straw have been used to remove up to 50% of NH$_4^+$ and 65% of suspended solids from irrigation water, prior to its application to land [Avnimelech et al., 1993]. In addition, the authors suggest that the protein (12-15%) and P (0.3-1.0%) enriched straw could be used as a manure or even as an ensiled feed material. Other low cost, innovative ideas to remove N from soil waters include below-ground water inceptors consisting of barriers using sawdust [Blowes, et al., 1994, Robertson, et al., 2000, Schipper and Vojvodic-Vukovic, 1998]

1.5 Biofilters for air

Gaseous emissions from housing (naturally ventilated, fan ventilated), waste storage, and composting

Atmospheric emissions from livestock enterprises include gases, dust particles, and odours. The main gases emitted from livestock housing, and of concern to the atmosphere, are NH$_3$, carbon dioxide (CO$_2$) and methane (CH$_4$). Dust particles can act as carriers of malodorous gases and pathogenic micro-organisms, which have human health concerns. Emissions of aerial pollutants to the atmosphere are generally via the ventilation systems of animal buildings or are from uncovered livestock waste storage facilities.

The annual NH$_3$ emission from pig housing is an estimated 19Kt and represents 7% of the agricultural total and 6% of the UK total NH$_3$ emissions. During summer and winter, emissions of NH$_3$ from pig housing is approximately 9.0 g and 6 g animal d$^{-1}$, respectively [Groenstein, 2000].

The annual NH$_3$ emission from poultry housing is an estimated 24Kt and represents of the 9% of the agricultural total and 7.5% of the UK total NH$_3$ emissions [Defra, 2002].

In livestock housing where there is no artificial ventilation, other abatement strategies for the control of CH$_4$ and NH$_3$ emissions may be more relevant, such as the additional or targeted use of straw [Chadwick, 2005], or dietary modification [Misselbrook et al., 1998; Sutton et al., 1999].

Of the air pollution control technologies, biological systems have been shown to be the most cost effective [Vaith et al., 1996]. The main systems used in the deodorisation of waste gases are bioscrubbers, biotrickling filters and biofilters. These systems are generally categorised by the mobility of the liquid phase and the microorganisms [Le Cloirec, 2006]. Briefly, bioscrubbers remove airborne pollutants by first passing the contaminated air stream through a spray chamber, so that the air comes into contact with a liquid phase. Pollutants are dissolved, and the contaminated liquid phase is treated biologically by suspended microorganisms, typically in a separate chamber. Biotrickling filters operate through piping a contaminated air stream through a medium through which there is a steady flow of water which may be recycled around the system. The media supports a fixed biofilm of microorganisms that
degrade pollutants in the air-stream. A moving liquid phase has advantages in that the biofilm receives a constant supply of nutrients, removes potentially toxic degradation by-products, continual microbial reseeding of the system and help in the diffusion of hydrophilic pollutants into the biofilm [Devinney et al., 1999]. In biofilters, the liquid phase is stationary and is associated with the filter medium, onto which a biofilm is attached.

Despite the different configurations and numerous variations on these, the basic processes for pollutant removal are similar for each of the biological systems, and these are a combination of adsorption, absorption and microbial degradation. Further descriptions and operating conditions of bioscrubber and biotrickling filter systems are not included in this review.

Biofilters were designed primarily for odour control at rendering plants, composting operations, chemical manufacturers and wastewater treatment plants. In the case of the latter they have been used for the last 40 years [Carlson and Leiser, 1966]. Within industrial settings, the removal of pollutants from emissions has traditionally utilised chemical and/or physical treatment technologies. However, the cost of construction and operation of these technologies have meant that these are only cost beneficial for large-scale operations [Lau et al., 1996]. The use of biofiltration for industrial operations is being increasingly recognised as a more economic and environmental than the existing methods such as thermal and chemical decomposition [Li et al., 1996]. In addition, some of the techniques employing combustion and absorption methods produce by-products, which still impact on the environment.

Improvements in design and operation of biofilters have led to improved efficiency and reliability. Key aspects to the use of biofilters are that they have low operating costs, and there is an absence of harmful residuals. However, drawbacks to the use of biofilters include a high area requirement, and moderate to large capital costs. In addition, biofilters are only able to cope with moderate concentrations of relatively soluble and biodegradable substances [Ergas and Cárdenas-González, 2004]. Another concern is the efficiency of biofilters under conditions of fluctuating loading since biofilter operate best with a steady state loading rate [Devinney et al., 1999]. Although some studies have investigated biofilter performance under conditions of fluctuating flow and loading [Chen et al., 2004; Deshusses, 1997], there is little information available for farm-based biofilters.

A range of natural organic and inorganic (soil, compost, heather, straw, woodchips, sawdust, shredded bark, sand, manure, mollusc shells, coconut fibre) and synthetised (perlite, activated carbon) support media are used in biofilters. These materials can either be used individually, or in a combination [Nicolai and Janni, 2001; Tymczyna et al., 2004]. The criteria for selection of an appropriate biofilter media include a large surface area, both for microbial growth and pollutant absorption, the ability to retain moisture in order to maintain an effective biofilm layer, and the ability to retain them and supply nutrients to microbes. The choice of biological support media in closed biofilters is important with regard to achieving a large surface are per unit volume without increasing the resistance to air flow [Hartung et al., 2001b].
One of the main problems that may occur during biofiltration is that of increased backpressure or pressure drop. This is due to an increasing microbial biomass which reduces pore space, thereby restricting the flow of gas through the filter. Therefore, the filter medium should be porous enough at the start so that there is low resistance to air flow whilst providing a sufficiently large surface area for the establishment and growth of microbial organisms.

In basic biofilters, contaminated air is passed through a filter media and the removal of odorous constituents and particulates takes place by two mechanisms; adsorption/absorption and bio-oxidation. Odour compounds and particulates are adsorbed onto the surface of the filter particles and/or are absorbed into the water layer surrounding the filter particles. Compounds are oxidised to relatively benign metabolic by-products, such as carbon dioxide, minerals, and water by the biofilms that are growing on the media. The biological oxidative degradation process is carried out by heterotrophic and chemo-organotrophic microorganisms, and can be described thus:

\[
\text{Organic pollutant} + O_2 \rightarrow CO_2 + H_2O + \text{heat} + \text{biomass}
\]

The efficiency of biofiltration of air is dependent on the type of media used, the moisture content, size and viability of the biofilm, an adequate supply of substrate and oxygen, loading rate of pollutants, air flow direction, process/retention time, pH, and temperature [Devinney et al., 1999; Kalingan, et al., 2004].

Compost is a particularly effective medium as it carries a well established and mixed population of microorganisms and has a good moisture holding capacity. However, for some materials, inoculation or seeding of media may be necessary to establish a required population, such as nitrifying and sulphate bacteria. Inoculants can be obtained from soil or activated sewage sludge [Devinney, et al., 1999]. A short conditioning period of up to 3 weeks may be necessary to allow the microbial population time to adapt to the odorous compounds in the exhaust gases [Nicolai and Schmidt, 2005].

When a biofilter is working correctly, odorous compounds that are adsorbed to the filter material are oxidised which, frees up sites for the adsorption/absorption of additional compounds from the incoming contaminated air stream. Thus, the odour removal capacity of the biofilter is self-regenerating. For maximal removal of odours, the rate of the microbial oxidation of the adsorbed compounds must equal the rate of their adsorption/absorption. If the degradation rate is less than the rate of adsorption/absorption, the filter sites will become saturated, thus, allowing the passage of undegraded contaminated compounds into the atmosphere [Williams and Miller, 1992]. The design and characteristics of biofilter reactors for air pollution control are fully discussed in Devinney et al. (1999).

There are several different designs and configurations of biofilter which may or may not incorporate some of the features used in bioscrubbers and biotrickling filters. Briefly, types of biofilters are categorised by their
configuration (open or closed) and by the direction of air flow (up-flow, down-flow, horizontal-flow) through the filter. In open biofilters, the outflow from the biofilter discharges directly to the atmosphere, whereas in enclosed biofilters, both the inflow and outflow are controlled by the system. Open systems are well suited for applications where there is no restriction on space, and are a low cost and often preferred option to control emissions from livestock operations [Nicolai and Lefers, 2006]. However, open systems are more vulnerable to weather conditions such as rainfall and temperature, which may affect their performance, than closed systems. In contrast, closed systems are more complicated and expensive than open systems, but are less affected by weather, and generally take up less space (Fig. 2). Down-flow direction can be twice as efficient in pollutant removal than up-flow, as maintenance of the water balance is easier [Arnold et al., 1997], though up-flow appears to be the preferred option in low cost open biofilter systems [Hartung et al., 2001a; Nicolai and Janni, 2000]

In open biofiltration of livestock housing, air is passed through a moist organic medium, typically with a high C content, such as wood chips, peat or compost, or an inorganic media such as sand or gravel. The medium is placed on a slatted support, or plenum, and pit fans and/or ventilation fans, push air from a building via an air duct, through the plenum and up though the layer of medium (Fig. 3). In simple low cost biofilters, wooden pallets have been used to construct the plenum [Schmidt, et al., 2004]. Pollutants such as volatile organic carbon (VOC), NH₃, and malodorous compounds are adsorbed/absorbed by the filter material and are oxidised by microbes to CO₂, water, and mineral salts. Some compounds are assimilated into the microbial biomass.
Main design considerations include the properties and depth of the media material(s), the airflow rate through the media, the performance of fans, the retention time of contaminated air in the biofilter, moisture control and the area of the biofilter required [Janni and Nicolai, 2000].

![Figure 3. Cross-section schematic of an open biofilter used to filter odorous emissions from ventilated livestock housing.](image)

A critical factor which affects the performance of a biofilter is the empty bed residence time (EBRT). This parameter is an estimation of the treatment time of a contaminated air stream and is easily determined by dividing the empty filter bed by the volume of airflow. This is an overestimated value of the true residence time since the support medium will occupy a large part of the filter volume. However, as the porosities of media are difficult to determine, EBRT is the preferred calculation [Devinney et al., 1999]. An increase in EBRT, either by increasing the volume of media or by reducing the flow rate will increase the biofilter performance. Typical residence times for biofilters used in industrial and commercial applications range between 25 seconds for odours and low concentrations of VOCs to over a minute for large concentrations of VOCs [Leson and Winer, 1991]. The general operating conditions and performance of biofilters are given in Table 2.
Table 2. Operating conditions of a biofilter. Adapted from Le Cloirec (2006).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Value</th>
<th>Remarks</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gas phase velocity</td>
<td>m h⁻¹</td>
<td>100-500</td>
<td>Low values are due to low kinetic rates of degradation. Depending on the molecule</td>
</tr>
<tr>
<td>Air residence time</td>
<td>s</td>
<td>15-90</td>
<td>degradation kinetic: alcohols &gt; ketones &gt; n-alkanes &gt; aromatics.</td>
</tr>
<tr>
<td>Bed porosity</td>
<td>-</td>
<td>0.4-0.95</td>
<td>High values avoid clogging.</td>
</tr>
<tr>
<td>Specific surface</td>
<td>m² m⁻³</td>
<td>100-400</td>
<td>High values give a better mass transfer and biomass concentration. Compromise between the residence time and pressure drop. Depending on amount of clogging and compression of support media.</td>
</tr>
<tr>
<td>area</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Filter depth</td>
<td>m</td>
<td>0.5-2.5</td>
<td></td>
</tr>
<tr>
<td>Pressure drop</td>
<td>m H₂O</td>
<td>0.1-0.5</td>
<td></td>
</tr>
<tr>
<td>Air Humidity</td>
<td>%</td>
<td>60-100</td>
<td>High values are better.</td>
</tr>
<tr>
<td>Water pH</td>
<td>-</td>
<td>5-9</td>
<td>Depending on solubility of pollutants.</td>
</tr>
<tr>
<td>Temperature</td>
<td>°C</td>
<td>10-40</td>
<td>Suitable for mesophilic organisms.</td>
</tr>
<tr>
<td>Acclimation time</td>
<td>d</td>
<td>8-30</td>
<td>Function of the biodegradability of pollutants. inhibition may occur at higher concentration of pollutants or by-products.</td>
</tr>
<tr>
<td>Pollutant</td>
<td>mg m⁻³</td>
<td>1-1000</td>
<td></td>
</tr>
<tr>
<td>concentration</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Efficiencies</td>
<td>%</td>
<td>90-99</td>
<td>Depending on pollutant molecules. Inorganic material help increase life time.</td>
</tr>
<tr>
<td>Life time</td>
<td>y</td>
<td>2-5</td>
<td></td>
</tr>
</tbody>
</table>

1.6 Biofilters for Livestock Wastes

In their report, Fogg et al. (2005) conclude that constructed wetlands are a widespread and reasonably well understood soft engineering option for the mitigation of pollutants from various domestic and industrial sources. However, the authors point out that wetlands are not suitable for some agricultural effluents such as slurry and silage effluent. They are best suited for treatment of diluted point source effluents such as parlour washings, or used as part of an integrated system that uses a pre-treatment method (e.g. oxidation/settlement ponds) to remove solids; mainly in order to maintain the longevity of the system [Hunt and Poach, 2001; Kadlec and Knight, 1996]. Because of the anaerobic nature of wetland sediments, nitrification is often inhibited and thus, denitrification can be nitrate limited. Phosphorus removal is also limited by the anaerobic conditions [Hunt and Poach, 2001]. Thus, wetlands used in conjunction with other treatment measures that promote oxidation, are considered necessary in order to increase their efficiency [Poach et al., 2004].
A summary of the performance of constructed wetlands used for treating wastewaters from cattle feeding, dairy, and pig operations in the USA, indicates that reductions of 53 - 81% of total suspended solids, 92% of faecal coliforms, 59 - 80% BOD5, 46 - 60% NH3-N (percent), and N (44 to 63 percent) can be achieved [CH2M Hill and Payne Engineering, 1997]. The efficiency of removal of pollutants by constructed wetlands is generally higher for pig wastewaters compared with dairy wastewaters, but this is probably due to smaller loading rates for pig wastewater systems [Cronk, 1996].

Given the cost of their construction, Fogg et al. (2005) suggest that UK farms are unlikely to use constructed wetlands for the treatment of dilute point sources such dirty water over more conventional methods such as addition to the slurry store or low rate irrigation. In addition, there is a large variation in the efficiency of pollutant removal (Table 3) which, may be due to a range of factors including loading rates, design, maintenance, construction and plant cover.

Storage type filters such as waste detention ponds and lagoons can be used to intercept runoff from land, and waste products from hard standings, collecting and feeding yards and buildings, thus preventing pollutants and pathogenic organisms from entering watercourses. These types of systems take advantage of microbiological, and in some case plant processes, to reduce the nutrient content and BOD of liquids.

Ponds or lagoons are widely used in New Zealand to store and treat dairy and piggery farm waste. The set up is generally based on a two-stage pond system, whereby the first stage is anaerobic. The second stage facultative pond, consists of an aerobic top layer over an anaerobic bottom layer. Discharge of the effluent to the land usually follows the second stage. This treatment method can remove up to 75% of total N, and 95% of suspended solids and BOD. However, effluent quality can be very variable and the concentration of ammoniacal nitrogen and faecal bacteria can have an impact on the quality of receiving waters and their aquatic life [Hickey et al., 1989]. Efficiency of the system can be improved by continuous aeration of the second stage, which encourages nitrification and reduces the BOD demand [Sukias et al., 2003]. Constructed wetlands can be an effective secondary/tertiary or final polishing stage for effluent wastewaters [Tanner and Sukias, 2003].
Table 3. Removal efficiency of wetlands used to treat livestock wastes

<table>
<thead>
<tr>
<th>System Details</th>
<th>Size</th>
<th>Input</th>
<th>Loading</th>
<th>Nitrogen</th>
<th>Phosphorus (P)</th>
<th>Suspended solids (SS)</th>
<th>BOD</th>
<th>Pathogens</th>
<th>Reference</th>
<th>Country</th>
</tr>
</thead>
<tbody>
<tr>
<td>Surface flow constructed wetland (planted with burr-reed and cattails).</td>
<td>2 cells each at 3.6m X 33.5m</td>
<td>Diluted piggery waste effluent from anaerobic lagoon</td>
<td>25 kg Total N ha$^{-1}$ d$^{-1}$ at HLR of 2cm d$^{-1}$</td>
<td>92%</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Faecal coliforms, E. coli, and enterococci reduced by 98%, 99%, 80% respectively</td>
<td>Hill and Sobsey (2001)</td>
</tr>
<tr>
<td>Constructed wetlands</td>
<td>0.04 ha</td>
<td>Diluted piggery waste effluent from 2 stage lagoon</td>
<td>Concentrations of Total N, BOD, P and SS were 104, 45, 66 and 118 ppm respectively</td>
<td>97.6%</td>
<td>79.4%</td>
<td>95.9%</td>
<td>91.7%</td>
<td></td>
<td>McCaskey et al. (1994)</td>
<td>USA</td>
</tr>
<tr>
<td>Marsh-pond-marsh (planted with cattail)</td>
<td>6 cells each at 11 m x 40 m</td>
<td>Piggery waste effluent from a primary lagoon of a 2 stage system</td>
<td>138 – 248 N l$^{-1}$ at 7.1-12.6 m$^3$ d$^{-1}$</td>
<td>37-51%</td>
<td>13-26%</td>
<td>35-51%</td>
<td></td>
<td></td>
<td>Poach et al. (2004)</td>
<td>USA</td>
</tr>
<tr>
<td>Marsh-pond-marsh (planted with cattail and bullrushes)</td>
<td>4 cells each at 11 m x 40 m</td>
<td>Pig wastewater</td>
<td>16 kg N ha$^{-1}$ d$^{-1}$ 21 days retention time. 32 kg N ha$^{-1}$ d$^{-1}$ 10.5 days retention time.</td>
<td>51%</td>
<td>37%</td>
<td>30-45%</td>
<td></td>
<td></td>
<td>Reddy et al. (2001)</td>
<td>USA</td>
</tr>
<tr>
<td>System of 3 staged recirculating vertical flow reed beds</td>
<td>12 sub-beds each at 2.56 m$^2$</td>
<td>Mixture of pig slurry &amp; domestic sewage effluent</td>
<td>250 mg NH$_4$N l$^{-1}$ at 0.155 m$^3$ m$^{-2}$ d$^{-1}$</td>
<td>61.7%</td>
<td>48.9%</td>
<td>95.1%</td>
<td>99.1%</td>
<td></td>
<td>Sun et al., (1998)</td>
<td>UK</td>
</tr>
<tr>
<td>Constructed surface flow wetlands treating wastewater from 2 stage lagoon</td>
<td>6 cells each at 405m$^2$</td>
<td>Dilutes Pig slurry from 500 pig yr$^{-1}$ finishing system</td>
<td>104 mg Total N l$^{-1}$ 66 mg P l$^{-1}$ 155 mg SS l$^{-1}$ at 9879 L d$^{-1}$</td>
<td>61%</td>
<td>55%</td>
<td>86%</td>
<td></td>
<td></td>
<td>McCaskey et al., (1994)</td>
<td>USA</td>
</tr>
<tr>
<td>Vegetated filter strip</td>
<td>22.9m x 7.6m with 2% slope</td>
<td>Effluent from detention pond for dairy yard runoff</td>
<td>0.384 kg Total N m$^{-2}$ yr$^{-1}$ 0.174 kg P m$^{-2}$ yr$^{-1}$ 1.694 kg SS m$^{-2}$ yr$^{-1}$</td>
<td>18%</td>
<td>12%</td>
<td>33%</td>
<td></td>
<td></td>
<td>Schelling and Clausen (1992)</td>
<td>USA</td>
</tr>
</tbody>
</table>
Malodorous materials

The reduction of malodorous products from wastes such as slurries can be achieved by aeration [Westerman and Arogo, 2005] which incurs the cost of an energy input. Another option is the use of a polymer based biocover. These types of cover not only help to reduce NH₃ and H₂S emissions by up to 54 and 58% respectively but also help to enhance anaerobic digestion by 25% [Zahn et al., 2001]. In Canada, a low cost option for the control of odour and NH₃ emission from slurry storage facilities, is the use of peat moss and straw as a floating biofilter. The material is blown on to the slurry in spring, using a forage harvester, in order to provide a diffusion barrier between the surface of the slurry and the atmosphere. This has the benefit of reducing N losses in storage by up to 60%, since the peat moss adsorbs NH₃ and improves plant nutrient uptake in the field by 50% [Barrington et al., 1990].

Sulphur containing compounds are not adsorbed, but peat and straw help to reduce their diffusion [Alkanani et al., 1992]. In addition, the straw can be wetted with manure to increase biological activity and encourage biofiltration [Zhang et al., 1999]. This type treatment is effective at reducing odours by 40-90%, H₂S by 80-94%, and NH₃ by 25-85% and can last up to 6 months. Cost are approximately $0.25 – 1 / yd² [Nicolai et al., 2005]. Odour emissions from slurry and manure storage can also be controlled with open bed type biofilters in conjunction with a geotextile cover [Nicolai et al., 2005].

It has been shown that storage of slurry in single and two-stage lagoons reduces the concentration of pathogens such as salmonella, fecal coliforms, and E. coli, though efficiency is dependent on temperature [Hill and Sobsey, 2003]. This is not a problem during summer months but may be during the winter. Depth is also important. For instance, shallow maturation ponds (0.4-1.5m) are more efficient at faecal coliform removal than deeper ones, due to increased pH (>9.0), from increased carbon dioxide production from enhanced algal photosynthesis, and photo-oxidation by ultra-violet light [Curtis et al., 1992].

The efficiency of nutrient removal in storage based systems is dependent on retention time; the greater the retention time, the greater the efficiency of removal [Bolan et al., 2004]. This is especially the case for open or flowing systems which produce an effluent. Storage ponds and lagoons can overflow during period of excess rainfall, or where the influent supply exceeds that of the holding capacity of the system, which can lead to a point source of pollution.

A lagoon system in France, used to treat swine manure, consisted of algal ponds, daphnid ponds and a polishing fish pond. Results showed that low temperatures (<5 °C) inhibited significant biomass production and levels of NH₄⁺N and P stripping were small. However, when temperatures increased in spring, NH₄⁺-N removal, due to stripping, was 71% and algal productivity rose to 3.5g dry mass m⁻²·d⁻¹ [de al Noue, et al., 1994]. Slurries obtained from the de-sludging of anaerobic ponds used to store farm dairy effluents in NZ, had average concentration values of N, P and K in the slurry obtained, of
1650, 290 and 510 mg l\(^{-1}\) respectively and represented an accumulation of the minerals over a 5 yr period [Longhurst \textit{et al.}, 2000].

The recommended system for the treatment of piggery wastes is a system of ponds in series [Costa \textit{et al.}, 2000]. For example, piggery waste in Brazil was treated using a system consisting of an equalizer, a decanter and 4 ponds comprising 2 anaerobic ponds, a baffled facultative one and a maturation pond containing water hyacinths [Zanotelli \textit{et al.}, 2002]. The system was provided with 3 m\(^3\) d\(^{-1}\) of piggery waste and was monitored for a 12 month period. Table 4 shows the pond characteristics and mean nutrient removal at each stage.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Anaerobic 1</th>
<th>Anaerobic 2</th>
<th>Facultative</th>
<th>Maturation</th>
<th>Effluent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Surface area (m(^2))</td>
<td>83.6</td>
<td>83.6</td>
<td>105.6</td>
<td>100</td>
<td></td>
</tr>
<tr>
<td>Base area (m(^2))</td>
<td>44.5</td>
<td>44.5</td>
<td>67.6</td>
<td>46.0</td>
<td></td>
</tr>
<tr>
<td>Depth (m)</td>
<td>1.7</td>
<td>2.2</td>
<td>0.85</td>
<td>0.80</td>
<td></td>
</tr>
<tr>
<td>Volume (m(^3))</td>
<td>106.4</td>
<td>137.7</td>
<td>73.0</td>
<td>58.0</td>
<td></td>
</tr>
<tr>
<td>Flow rate (m(^3) d(^{-1}))</td>
<td>3</td>
<td>3</td>
<td>3</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>Retention time (d)</td>
<td>35</td>
<td>46</td>
<td>24</td>
<td>19.3</td>
<td></td>
</tr>
<tr>
<td>Influent TN (mg l(^{-1}))</td>
<td>1967</td>
<td>1418</td>
<td>781</td>
<td>412</td>
<td>196</td>
</tr>
<tr>
<td>Influent TP (mg l(^{-1}))</td>
<td>495</td>
<td>210</td>
<td>99</td>
<td>46</td>
<td>25</td>
</tr>
<tr>
<td>% TN removal</td>
<td>25</td>
<td>47</td>
<td>47</td>
<td>52</td>
<td></td>
</tr>
<tr>
<td>% TP removal</td>
<td>58</td>
<td>53</td>
<td>54</td>
<td>46</td>
<td></td>
</tr>
</tbody>
</table>

1.7 Biofilters for livestock housing

The use of biofiltration for treatment of livestock emissions began in Germany in the late 1960's [Zeisig and Munchen, 1987] and gained more popularity in the 1980's in Sweden [Noren, 1985]. Li \textit{et al.} (1996) give a good overview of the main theoretical principles of biofilters, used primarily for malodorous control for the livestock industry. Biological treatment of air has been found to be a robust, cost effective and an efficient way to treat low concentration biodegradable odorous compounds that are emitted from livestock facilities [Martinec \textit{et al.}, 2000; O'Neill \textit{et al.}, 1992].

Li (1997) suggests that odour and dust in livestock housing should be controlled by internal recirculation of air through biofilters. This not only helps improve the quality of air emissions from the housing, but also has implications for the control or prevention of possible dust related respiratory problems in both humans who work in the buildings, and the housed animals. However, biofilters are not generally designed for the filtration of particulate materials, and accumulated dust can cause blockages in closed type biofilters which, will require the replacement of the media in order to resume effective operation [Devinney \textit{et al.}, 1999]. The installation of a bioscrubber or pre-filter may be necessary in order to remove the majority of the particulate material before the air enters the biofilter.

Malodorous emissions typically contain low concentrations of H\(_2\)S, mercaptans, and other reduced sulphur compounds. Particulate materials originating predominantly from feed, faeces, skin cells, hair and feathers can adsorb and concentrate odours to levels that a can be greater than an
equivalent volume of air. [Bottcher, 2001]. Thus, trapping particles can help to reduce malodorous emissions. For example, odours can be reduced from between 40 to 70% by trapping 45% of particles sized between 5 – 10 μm and 80% of particles sizes > 10 μm [Hoff et al., 1997]. Trapping of dust may also be beneficial in the reduction of NH₃ emissions, since this also strongly binds to particles [Ullman, 2005].

Up-flow open-bed type biofilters are generally used for livestock operations as they are considered more economical to install and maintain [Nicolai and Lefers, 2006]. In addition, maintenance and operation of open-bed biofilters is relatively easy. Most importantly the filter media must be kept at a moisture level of between 30-60% so that a biofilm containing the microorganisms is maintained on the surface of the material. A lawn sprinkler on a timer can be used during times of no precipitation. In winter, precipitation is usually enough to keep the biofilter sufficiently moist. Oxygen supply to the microorganisms needs to be maintained to at least 4% of their surrounding atmosphere [P. Hobbs, pers. comm.].

Reduced S compounds and chlorinated organic compounds are degraded to inorganic acids [Janni and Nicolai, 2000]. These can accumulate and reduce the pH of the medium, thereby affecting the environmental conditions of the biomass, and hence the performance of the biofilter. Therefore, it may be necessary to treat the filter material periodically with CaCO₃, in order to raise the pH back to an optimum level which should be as near neutral as possible or use a filter material with a high buffering capacity [Devinney et al.,1999].

Other maintenance may include rodent control, and weeds should be controlled with a herbicide. Air leaks around the perimeter of the biofilter should be repaired in order to avoid short-circuiting the system. It is important not to allow people or animals to walk on biofilters, since the media will become compacted and lose porosity which, will cause backpressure. In addition, it is recommended that the layer of media in open biofilters should be no more than 1-1.5m deep in order to avoid compaction [Devinney et al.,1999].

Janni et al. (1999) set up an experiment to demonstrate the effectiveness of low cost open biofilters to reduce emissions of odour, H2S and NH₃ emissions from livestock facilities. The performance of 12 biofilters with a range of different designs, residence times, and medium was measured on dairy, pig and poultry facilities and the results for odour and H₂S removal are given on Tables 5 and 6 respectively. The data show that odour reduction from the poultry facility was poor (<40%) in comparison with that for dairy and pig. Apparently, this was due to the accumulation of dust on the extraction fans which, resulted in inaccurate airflow measurements. In contrast, odour reduction from dairy and pig facilities was generally >80% and improved with increased residence time. Similar trends were found for H₂S reduction. In order to achieve >80% reduction in emissions from dairy and pig facilities, the authors recommend a minimum residence time of 5s.
Table 5. Percent reduction in odour detection threshold measured in odour units (OU). Taken from Janni et al. (1999)

<table>
<thead>
<tr>
<th>Inlet OU</th>
<th>Varying media depth (m)</th>
<th>Varying media depth (m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0.15</td>
<td>0.30</td>
</tr>
<tr>
<td>Residence Time (s)</td>
<td>3.4</td>
<td>7.2</td>
</tr>
<tr>
<td>OU %</td>
<td>OU %</td>
<td>OU %</td>
</tr>
<tr>
<td>Dairy</td>
<td>120 53 56 38 68 44 64 49 60 36 70 32 73</td>
<td></td>
</tr>
<tr>
<td>Pig</td>
<td>757 265 65 85 89 68 91 138 82 49 94 51 93</td>
<td></td>
</tr>
<tr>
<td>Poultry</td>
<td>117 80 31 83 29 71 39 106 9 110 6 76 35</td>
<td></td>
</tr>
</tbody>
</table>

Table 6. Percentage reduction in hydrogen sulphide. Taken from Janni et al. (1999)

<table>
<thead>
<tr>
<th>Inlet ppm</th>
<th>Varying media depth (m)</th>
<th>Varying media depth (m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0.15</td>
<td>0.30</td>
</tr>
<tr>
<td>Residence Time (s)</td>
<td>3.4</td>
<td>7.2</td>
</tr>
<tr>
<td>ppm %</td>
<td>ppm %</td>
<td>ppm %</td>
</tr>
<tr>
<td>Dairy</td>
<td>102 17 83 11 89 10 91 20 81 13 87 7 93</td>
<td></td>
</tr>
<tr>
<td>Pig</td>
<td>786 335 57 65 92 33 96 142 82 45 94 32 96</td>
<td></td>
</tr>
<tr>
<td>Poultry</td>
<td>89 87 3 77 13 66 26 82 8 67 25 62 31</td>
<td></td>
</tr>
</tbody>
</table>

The performance of open biofilters for NH₃ removal is very variable (Table 7) and Martens et al. (2001) argue that, in comparison to odour removal, biofilters are not suitable for NH₃ reduction. This is largely due to reduced rates of removal with increased filter volume load and hence, air retention time. However, improvements in NH₃ removal can be made by increasing the moisture content of the filter material to up to %50 [Hartung et al., 2001a].
Table 7. Removal efficiency of biofilters used to treat gaseous emissions from livestock facilities

<table>
<thead>
<tr>
<th>System Details</th>
<th>Size</th>
<th>Inputs</th>
<th>Loading</th>
<th>Ammonia</th>
<th>Methane</th>
<th>Odours</th>
<th>Reference</th>
<th>Country</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aerobic up-flow polyacrylic tube filter. Peat and inorganic supporting material (e.g. perlite, vermiculite) packed in a perfusion column</td>
<td>50cm height 5cm diameter</td>
<td>Ammonia gas from cylinder</td>
<td>200 mpv at 0.03 m³ h⁻¹ 400 mpv at 0.03m³ h⁻¹</td>
<td>100%</td>
<td>65%</td>
<td></td>
<td>Kalingan et al., (2004)</td>
<td>Taiwan</td>
</tr>
<tr>
<td>Pilot scale. Aerobic down-flow, cylindrical metal container with pre-filter to remove dust particles and humidifier. Woodchips as media @ 64-69% moisture. 3 trials at diff. volumetric loading rate</td>
<td>0.91m height 0.56m diameter</td>
<td>Exhaust air from a 12 pen pig finishing house</td>
<td>769-1898m³ m⁻³ air flow Odour: 476-2149 OU m⁻³; 41-65 OU m⁻³ s⁻¹ LU⁻¹ Ammonia: 7.1 – 8.3g animal⁻¹ d⁻¹</td>
<td>54-93%</td>
<td>77-95%</td>
<td></td>
<td>Sheridan et al., (2002)</td>
<td>Ireland</td>
</tr>
<tr>
<td>Prototype open biofilter with humidifier. Peat, wheat straw, horse and sewage compost @ 64-71% moisture.</td>
<td>10m² area 1.2m height</td>
<td>Exhaust air from laying hen house</td>
<td>Ammonia: 19.9-34.7 mg m⁻³</td>
<td>36-89%</td>
<td></td>
<td></td>
<td>Tymczyna et al., (2004)</td>
<td>Poland</td>
</tr>
<tr>
<td>Plexiglass cylinder. Perlite and garden compost (40:60 v/v). Filter bed volume = 160 L.</td>
<td>0.95m height 0.49m diameter</td>
<td>Ventilation from covered liquid pig manure - storage (6m³)</td>
<td>Methane:500-5500 mg m⁻³ @ 0.75-8.5 m⁻³ m⁻³ h⁻¹</td>
<td></td>
<td>85%</td>
<td></td>
<td>Melse, and Van Der Werf (2005)</td>
<td>Netherlands</td>
</tr>
<tr>
<td>Open biofilter with compost and kidney bean straw at depth of 0.3m. Average residence time of 5s</td>
<td>6.4m x 6.4m</td>
<td>Odours from farrowing house</td>
<td>100-850 odour units</td>
<td></td>
<td></td>
<td>82%</td>
<td>Nicolai and Janni (1998)</td>
<td>USA</td>
</tr>
<tr>
<td>Two open bed biofilters using coconut fibre and peat mixture (6.5 years old) at 0.28m depth</td>
<td>3.0m x 6.0m</td>
<td>Odours and ammonia from pig house</td>
<td>Ammonia : 1179-8938 mg m⁻³ h⁻¹ Odors: 20—1366 OU mg m⁻³ Ammonia:1590-10424 mg m⁻³ h⁻¹ Odors: 16-707 OU mg m⁻³</td>
<td>15%</td>
<td>78%</td>
<td>36%</td>
<td>Hartung et al., (2001)</td>
<td>Germany</td>
</tr>
<tr>
<td>Three pilot-scale biofilters with 50:50 manure compost and coconut peels at depth of 0.5m. Residence time of 151 s.</td>
<td>0.06 m³</td>
<td>Composting dairy manure</td>
<td>Ammonia: 33 -190 ppm</td>
<td></td>
<td></td>
<td></td>
<td>Hong and Park (2005)</td>
<td>South Korea</td>
</tr>
<tr>
<td>Five open-bed biofilters with range of different media at depth of 0.5m</td>
<td>2.18 m²</td>
<td>Odours and ammonia from pig house</td>
<td>Ammonia: 13.8ppm Odors: 1714 OU m⁻³</td>
<td>-43.1-35.5%</td>
<td>40-83%</td>
<td></td>
<td>Martens et al., (2001)</td>
<td>Germany</td>
</tr>
</tbody>
</table>
A 1992 review of seven options to control odours from pig and poultry housing, that included chemical or biological treatments of air leaving the building, or natural dilution of odorants in the atmosphere, concludes that the cheapest treatment was the use of a biofilter. Costs worked out at £8 per pig or £0.18 per broiler, in terms of both capital and annual costs. Capital and annual costs per unit of ventilation demand were £0.52 m$^{-3}$h$^{-1}$ and £0.14 m$^{-3}$h$^{-1}$ respectively [O'Neill et al., 1992].

1.8 Biofilters for pollutants in agricultural runoff and drainage

Buffer strips can remove up to 90% sediment attached P [Withers et al., 1997] but their effectiveness over the long term is in question [Dillaha et al., 1989]. However filter strips are not so efficient at retaining soluble P and NO$_3^-$. This is presumably because filters strips work by infiltration and will not prevent nutrients dissolved in surface flow waters to be transported to ground waters. Vegetative filters can be very effective at trapping herbicides and sediments in agricultural runoff waters. For example, removal rates of 40-85% for atrazine, 44-85% for metolachlor and 57-93% for sediment were achieved by 1.25m x 4m grass filter strips [Popov et al., 2006]. Similarly, in a ratio of drainage area to buffer strip area of 30:1, filters strips were able to remove nearly 80% of a sediment attached pesticide, chlorpyrifos [Arora et al., 2003].

In New Zealand, constructed wetlands have been used to reduce the nutrient loading of subsurface waters from grazed pastures. For example, the removal efficiencies for wetlands, comprising 1% of a catchment area, for the treatment of subsurface drainage from rain-fed dairy pastures for dissolved reactive P (DRP), NO$_3^-$, NH$_4^-$, organic N and total N were 80%, 78%, 41%, 99.8% and 96% respectively [Tanner et al., 2003]. However, these data are from a short-term study only and further work needs to be done in order to determine the seasonal variations and long-term efficiencies of these systems.

Research in Sweden has shown the importance of ponds for NO$_3^-$-N and P retention from agricultural waters. Budget studies were carried out on a series of restored ponds, dominated by the inflow from a stream receiving agricultural runoff. Pond depths ranged between 0.4-2.0m with hydrological loads of 0.14-5.2 m$^3$ m$^{-2}$ d$^{-1}$. The quantities of N and P removed were 150-7000kg ha$^{-1}$ yr$^{-1}$ and 18-404 kg ha$^{-1}$ yr$^{-1}$, respectively [Gustafson et al., 2000]. The study showed that periods of retention were also followed by periods of N release. Despite this, estimates of net retention ranged from 73-7000kg N ha$^{-1}$ yr$^{-1}$. In addition, there was an empirical relationship between the areal loading and the areal retention of N (kg N ha$^{-1}$ yr$^{-1}$) in that an increase in N loading per unit area leads to an increase in areal retention, but a decrease in percentage removal.

The influent and effluent NO$_3^-$-N concentration levels of a pond system (10 ha in total) receiving tertiary effluent from a constructed wetland system, were 16.3 mg l$^{-1}$ and 11.1 mg l$^{-1}$ respectively (32% reduction in N). The water budget for the ponds was: 37% as outflow, 3% was evaporated and 60% was infiltrated. The average NO$_3^-$-N concentration in extracts from beneath the
ponds was 3.6 mg N l\(^{-1}\). A mass balance approach suggests that 62\% of the total NO\(_3\) -N entering the ponds was unaccounted for and was presumably denitrified [Lund, 1999].

Permeable reactive subsurface barriers and reactors are where reactive materials are placed below the soil surface in such a way as to intercept contaminated groundwater, provide a preferential flow path through the reactive media, and transform contaminants into environmentally benign forms. For example, denitrification barriers consisting of a sawdust and soil mixture in a trench (35 m long, 1.5 m deep, and 1.5 m wide) reduced groundwater NO\(_3\) concentrations of 5-16 mg N l\(^{-1}\) to <2 mg N l\(^{-1}\) [Schipper and Vojvodic-Vukovic, 1998]. Similarly, in a 7 year study, a subsurface containerised reactor (1.9m\(^3\)) consisting of coarse wood mulch, was used to intercept groundwater from an agricultural tile drain. Concentrations of NO\(_3\) in the influent groundwater were an average of 4.8 mg N l\(^{-1}\) and the hydraulic loading rate varied from 800-2000 l d\(^{-1}\). The hydraulic retention time in the reactor ranged between 3-7 hours. The reactor reduced NO\(_3\) concentrations by up to 58\% with removal rates equating to 5 -30 mg l\(^{-1}\) d\(^{-1}\) depending on temperature. Bark based barrier walls have been shown to be effective in trapping polycyclic aromatic hydrocarbons (PAHs) in groundwater with an efficiency range of 77 – 99.99\% [Seo, et al.]

Furthermore, results suggest that these type of barriers have the potential to remain effective for up to 10 years, and possibly more, without replenishment of the reactive material [Blowes et al., 1994; Robertson et al., 2000].

2. Overcoming barriers to uptake and implementation of biofilters

2.1 Reputational issues

While it is possible to evaluate and improve the performance of biofiltration in purely scientific terms it is clear that the reputation of these techniques will dictated by their applicability to real world conditions. After all, it is here that operational problems come to be amplified. This point is true of all forms of biofiltration, but seems especially the case in terms of biofilters for air, since precedents for this technique are rarer, and operational certainties less clear. As Westerman et al (2000) put it “these technologies would benefit from demonstration applications in which the robustness of the technology and ease of operation could be tested under typical operating conditions”. It has already been noted, for instance, that the choice of packing materials when using this system has implications for the time-inputs of adopters, but as was suggested, these decisions also raise questions about the depth of care and expertise required. As Hoff et al note, while biofilters appear be simple to maintain, important procedures can sometimes be ignored, especially where substantial time inputs are required. Issues such as the dry out and over-watering of filtering material, as well as the compaction of these materials (which may lead to cracking) are partly a function of human behaviour and emphasise the need to test likely pitfalls at the point of their application. Leson and Winer (1991) may well be correct when remarking that “potential
system failures can be avoided and their causes minimised “ but it is clear that ostensibly simple codes of conduct, such as not allowing humans or animal to tread these materials, or turning and replacing the filter bed at appropriate intervals, cannot be assumed.

The need to work these handling issues of biofilters through in practical settings seems all the more the case when we reflect on Van Lith et al’s (1997) claim, that because biofilters have not always been developed in appropriate real world circumstances, they have been afforded the “reputation of a somewhat unreliable technology”. What comes with this is the need to proceed with caution in propagating these techniques as operationally simple and commonsense; key motifs of the trade press, and sometimes science-based inquiry too. Casual industry claims such as “treating foul air the keeping the filter clean and making sure the blower is in good working order is the only maintenance required” [Deutsch, 2006] can be damaging as they can be helpful when seeking to advance market potential. It is precisely in this sense that Reynolds et al (1999) strike a cautionary note when reporting that market entrants can be “deluded into the idea the biofiltration is a simple technology”, and that “all one had to do was put bugs in the box and blow air across the bed results mostly in failure”. In this instance the authors argue that “failures in poor design and construction have damaged the reputation of the technology and hurt its ability to penetrate the market”.

These points have applicability to other systems of Biofiltration. Jacobson et al. (2001) has noted in the case of straw based biocovers, for instance, that some users of the technique were not always successful in managing materials, while work taking place in the UK for Defra (was MAFF Project WA0641) have highlighted practical reservations with straw based approaches, such as the tendency to lose surface cover due to wind or sinking. Similarly, research into the efficacy of wetlands has pointed to performance problems in light of system management, inaccurate and over ambitious initial design, misinterpretation of the wetlands intended function, as well as lack of adaptiveness to changing management circumstances. It has been noted that reedbeds and buffer zones in particular require high levels of management when being established and maintained. Furthermore, these authors argue 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”. They argue that what this amounts to is the need for adequate provision of training in wetland design and management, backed up with the setting of approved design standards and certification.

In the context of adoption what these issues point to is that, where handling issues have not been anticipated and adequately worked through, uncertainties will be tend to amplified and confidence in techniques jeopardized. Potential users will be keenly aware of the practical, everyday, settings in which these techniques will come to operate and may identify operational problems not arrived at through purely scientific inquiry. Developing a programme of research that explores operational issues at the point of their real world application is therefore crucial. As we suggest below
this could be combined with extension programmes that can demonstrate these techniques in ways that reflect UK circumstances in general and sectoral priorities in particular.

2.2 Responsibilities and aspirations to adopt biofilters

The need to demonstrate the efficacy of these techniques in real world circumstances works alongside the real and perceived benefits that are thought to be accrued in mitigating the environmental and human health implications of enterprise practices. At one level these motivations are likely to reflect how well these techniques help them to respond to the exigencies of regulation and legislation currently surrounding issues of environmental protection in the UK, not least the Environmental Protection Act (1990) the Water Resources Act (1991) and relatedly, the Nitrate Directive (1991), Integrated Pollution Prevention Directive (1996), and Water Framework Directive (2000) respectively. For instance, in terms of air pollution, provisions made under the Environmental Protection Act 1990 give local authorities (and specifically their Environmental Health Department’s) the responsibility to monitor odour nuisances and take reasonably practicable steps to investigate complaints made to them. Farmers face the prospect of being served abatement notices where a local authority is satisfied that a statutory nuisance exists, or is likely to occur or recur. This means that farmers are responsible for the abatement of the nuisance or prohibiting/restricting its occurrence or recurrence; and the execution of such works and measures as may be necessary in respect of them. Similarly provisions made under the Water Resources Act (1991) discharges powers to the Environment Agency to intervene in circumstances where individuals cause or knowingly permit “a discharge of poisonous, noxious or polluting matter or solid waste matter into any ‘controlled waters’ without the proper authority”.

Those who follow ‘Codes of Good Agricultural Practice’ for air and water (1998), which make direct and indirect references to the biological treatment of manure, slurry and odorous air, will minimize the risk of causing water and air pollution/nuisance and therefore go a significant way towards avoiding the prospect of financial penalty. It is at least partly in this vein that Deutsch (2006) reports pointedly in relation to biofilters, regardless of the costs that come with their uptake of these techniques, they “may allow you to stay in business”. Less dramatically, it is clear that in the advent of the Single Payment scheme, following these protocols are increasingly related to farmers’ need to respond to the exigencies of cross compliance, and therefore are, in principle, pathways through which farming industries may begin to innovate further in the area of biofiltration. In other words, responsibilities that come of these external mandates provide a context to innovation which are simultaneously mapped on to questions of farm economy and enterprise viability.

Nonetheless, it remains a vexed question as to the extent to which farmers are fully aware of, and indeed feel they wish to respond to, this culture of compliance. For instance, in a recent Farm Practices Survey (2004) it was estimated that only 55% of farmers own a copy of the air code, and up to
70%, the water code. Moreover, this proportion tends to decrease among the livestock sector, significantly so in the case of smaller enterprises. Similarly, the same survey conducted in 2006 revealed that awareness of flagship water quality programmes, such as the Catchment Sensitive Farming initiative, though steadily increasingly, was at about 50%. And while the authors of this survey note correctly that awareness of programme and regulation is not the same thing as being unaware of the concepts and principles that underpin water quality (which they estimate to be significantly higher at 70%), it remains the case that the nature and importance of some of the terms employed by the regulatory environment, such as BOD, may well be lost on farmers. In many respects, motivations to adopt these techniques in relation to mandatory concerns may well be driven by wider cultural and geographical factors. Odour nuisances, for instance, are likely to be subject to greater public scrutiny and reaction than issues of water quality within locales, but highly variable depending on the geographical configuration of enterprises to wider settlement structures.

The reverse side of argument about regulation and legislation is that it is clear that some of these techniques can offer wider public benefits over and above the mitigation of environmental ‘bads’, such as the promotion of landscapes features for their biodiversity and amenity value, and therefore may be actively adopted in a more aspirational sense by farmers, especially where benefits accrue esteem among peer and professional networks. Importantly, these non-economic incentives for action may in some cases compensate for marginal losses and costs incurred in employing the techniques (Söderqvist, 2002). It is not the place of this review to the precise nature of these wider benefits, but a clear example in the context of this study is the use of wetland systems, whose benefits are widely recognised and valued. The UK Biodiversity Action Plan have identified a number of these as priority habitats and many operate under nationally and internationally recognized environmental designations. Moreover, constructed wetlands, a key system for biofiltration in this review, are considered to be a high value wildlife habitat, and have a perceived “naturalness” by the general public [Tanner et al, 2003].

While other biofiltration techniques do not fall neatly into this public goods and amenity category, it is interesting to note that in their “scientific” review of different odour abatement technique the use of biofilters and bioscrubbers was privileged over chimneys because they felt that the “chimney bring aesthetic disadvantages” [O’Neill et al 1992, p164] That is to say, it may be the case that biofiltration techniques are possibly chosen because they are deemed to fit and extend the overall landscape aesthetic.

Moreover, it is worth noting that capacities to act also take place in the context of the Town and Country Planning (General Permitted Development) Order 1995. This stipulates the need for planning permission before putting up any new buildings and converting existing buildings that house livestock and store slurries and manures where they are within 400 metres of any protected buildings such as houses and schools. Again this reinforces the geographical context to some of the way these biofiltration techniques may be adopted, and
indeed, it may be the case that their adoption helps ‘make the case’ for securing planning permission, where it is necessary.

2.3 Socio-economic perspectives on farmers’ adoption of new technologies: understanding the decision making process

Human decision-making in general, and farmer decision-making in particular, are affected by a complex interaction of a very wide range of influences. In the area of farmers’ approaches to nature conservation and environmental protection, for example, it is clear from the extensive research literature that their behaviour cannot be attributed to a single cause or influence, or even a small number of key factors; rather, farmer decision-making regarding the adoption of innovations and new techniques is driven by a complex interaction of these factors. In simple terms, these may be broadly classified as (a) external effects on the farm business and (b) the state of mind/attitude of the farmer. These are, respectively, objective and subjective factors and it is the interactions between them that govern farmers’ responses to policy signals.

Although policy and economic instruments can be effectively operationalised at the national and regional levels, the adoption of biofilters in a market-led agricultural sector is fundamentally a decision taken by individual farmers who must assess both the economic implications of the new technologies and also place this within the context of their own individual business circumstances. Theoretical and empirical research on farmer decision-making has tended to be framed within two debates, one informed by insights from the analyses suggested by production economics and the other informed by a wide range of behavioural studies. In practice, the development of effective policy instruments must take account of both conceptual frameworks in that these approaches are essentially complementary rather than competitive. However, it is important to note that their individual contributions act in constraining the role of advisory services relative to other influential factors and, moreover, that cultural influences on farmers are also significant influences [Dampney et al., 2001]. In contrast, while production economics can provide valuable insights into the economic factors involved in farmer decision-making, its application in relation to new and emergent agricultural technologies has not been extensive.

One area for behavioural research has analysed farm household behaviour in the context of wider structural forces [Evans and Winter, 1997]. More recent behavioural studies (mainly from geography and psychology) of farmer decision-making have demonstrated the value of focussing on individuals as decision-makers for adopting environmentally-sound practices (see, for example, Burton, 2004). Two theoretical dimensions to farmer decision-making have emerged through a range of behavioural studies, focussing on structural factors, such as economic conditions, farm size, family circumstances and training & development [Austin et al., 1998; Battershill and Gilg, 1997, Falconer, 2000, Wilson, 2000] and motivational factors, such as attitudes towards pro-environmental practices (Glasson and Potter, 1998; Wilson, 1996 and 1997). Ultimately these criteria need to be considered for the key farming operations within some simplified farm typologies, together
with the economic aspects of employing alternative strategies for achieving diffuse pollution mitigation.

Public policy designed to encourage the more widespread adoption of biofiltration techniques in UK agriculture to mitigate diffuse pollution, however well-intentioned, will only be successful therefore if it is designed in the light of the range of factors influencing farmer behaviour. There is in fact already considerable experience in this broad area, for example the adoption of policies aimed at developing more environmentally-friendly farming practices. Typically, such policies to encourage voluntary behavioural change amongst land managers have been designed to target specific segments of the population and utilise interventions around the motivators and de-motivators for change. While it is beyond the scope of this review to provide a detailed review of alternative behavioural theories relevant in a consideration of farmer decision-making (an example can be found in Jackson et al, 2006, which has provided the basis for much of the following discussion) it is important here to identify the principal paradigms shaping understanding of the processes involved.

Theory of Diffusion of Innovations
Even in the narrowly agricultural context, there has been considerable debate over many decades as to how decisions are made and the relative importance of internal and external factors within the farm decision-making process. The classic study by Ryan and Gross (1943) which studied the diffusion of sowing hybrid corn in Iowa provided the core evidence on which the Theory of the Diffusion of Innovations (TDI) was based. Their categorization of farmers into ‘innovators’, ‘early adopters’, ‘early majority’, ‘late majority’ and ‘laggards’ has been widely adopted in a wide range of contexts. Subsequently, this research resulted in a better understanding of the role of communication as a factor within the diffusion model, providing the classic bell-shaped and sigmoid adoption curves which have been so widely used in agricultural and rural sociology research. Indeed, different communication attributes were seen as defining characteristics of the members of each category of farmers.

The approach pioneered by Ryan and Gross (1943) has been corroborated and extended by numerous subsequent studies [Rogers, 1958 and 1995; Fliegel, 1993; Fisher et al, 2000] which emphasize the role played by social networks. Other studies have identified the important role played by extension services, agricultural agencies and mass media in the process of information diffusion [Wilkening, 1950; Saltiel et al, 1994; Longo, 1990] and, especially, by peer relationships [Copp et al, 1958], social class and size of farm [Feder and Slade, 1984]. Other studies have focused on aspects of the diffusion process, now widely accepted to be essentially non-linear. The non-linear approach downplays the role of government institutions in developing a rational, planned approach to diffusion, rather viewing farmers as relatively passive players responsive to (random) influences arising from social participation and communication [Fliegel, 1993; Feder and Umali, 1993]. While research into agricultural diffusion has been extended into aspects of adoption patterns in developing countries, a significant research sub-genre has focused on the adoption of conservation practices in agriculture [Pampel
and van Es, 1977; Feder and Umali, 1993; Neill and Lee, 2001; Forte-Gardner et al, 2004], the use of bio-technology in developed agriculture [Hategekima and Trant, 2002; Stewart et al, 2002] or both [Quaim and de Janvry, 2005; Pelaez and Schmidt, 2002].

Despite this long history diffusion theory has been criticized, however, particularly in respect of variations in the adoption of different technologies, its application in the agricultural sectors of developing countries and its omission of agribusiness firms. Nevertheless, its insights have proved valuable tools and there is broad acceptance of its continued relevance in understanding farmer behaviour in developed agriculture.

Theory of Reasoned Action
According to the Theory of Reasoned Action (TRA) the best predictor of behaviour is intention, in that a person's behaviour is strongly related to their attitude towards that behaviour. Intention is seen as the mediating variable between attitude and subjective norm, capable of predicting any given behaviour of an individual. It provides both a framework and a conceptual basis to guide policy agencies taking a marketing approach to strategies for behaviour change and can be used to evaluate the importance of those factors which have the most direct influence on voluntary behaviour. It was developed by Fishbein and Ajzen (1975) who later extended its application [Ajzen and Fishbein, 1980]. An extended model is the Theory of Planned Behaviour [Ajzen, 1985] which will be considered separately.

The TRA argues that attitude and subjective norm indicate intention, together forming the 'primary determinants of behaviour' [Ajzen and Fishbein, op cit]. The model is concerned with the internal (psychological) determinants of people's behaviour and has been applied for a wide range of physical and social situations [Parminter and Wilson, 2004]. The key assumption made is that 'human beings are usually quite rational and make systematic use of information available to them' [Ajzen and Fishbein, op cit].

Jackson et al (op cit) identify relatively few research studies utilizing the TRA in the context of agriculture, although several UK studies have tested its potential to predict the behaviour of farmers and other foodchain players [Thompson et al, 1994; Thompson and Panayiotopoulos, 1999]. The findings while not wholly consistent did nevertheless identify significant correlations between attitudinal beliefs and behavioural intentions. However, some foodchain studies have yielded contrasting or inconclusive results [Gorddard, 1992; Thompson and Vourvachis, 1995], as have studies from outwith the foodchain sector. With some critical evaluations suggesting that the TRA lacked robustness to handle situations where behaviour was not completely within the control of the individual, giving inadequate or incomplete predictions of behaviour, Ajzen and Fishbein extended the TRA into the Theory of Planned Behaviour.

Theory of Planned Behaviour
This theory is effectively an extended model of the TRA and was published in 1985 [Ajzen, in Kuhl and Beckmann]. While intention still holds a central role,
the Theory of Planned Behaviour (TPB) model includes a further element, ‘perceived behavioural control’, which facilitates the measurement of the strength of an individual’s belief that a behaviour can be controlled [Burton, 2004]. The TPB uses knowledge of attitudes, subjective norms and perceived behavioural control to provide an understanding of beliefs and so predict behaviour. In predicting behaviour the theory uses perceived behavioural control in two ways: through factors associated with motivation and the intention to behave in a particular way (via ‘intention’) and also through direct control of behaviour which is not in any way mediated by intention [Madden et al., 1992]. So the TPB includes perceived behavioural control in a dominant role in determining intention, a key modification of the original TRA. Briefly, the TPB postulates that behavioural intention is determined by (a) a combination of attitudes towards the expected outcomes of the behaviour, (b) the perception of the views of others towards the behaviour (the subjective norm) and (c) the anticipated degree of control over the decision to behave as planned (perceived behaviour control).

There have been a number of studies in agriculture which have tested the TPB in a range of situations and a fairly comprehensive review concluded that ‘the TPB is suitable for agribusiness research’ [Jackson et al, 2006). Of particular relevance for present purposes is work by Beedell and Rehman (1999 and 2000) which concluded that the TPB was a suitable tool for research into farmer behaviour in two respects: to confirm the range of behaviour patterns among a group of farmers, and to explore the extent to which differences in their beliefs are explained by differences in their behaviour. Furthermore, the TPB was used by Lynne et al (1995) to predict the behaviour of strawberry farmers in technology adoption.

More recently Rehman et al (2007) used the conceptual framework of the TPB in conjunction with empirical data to explore and better understand the behaviour and motivation of farmers adjusting to the reform of the CAP and provides particularly interesting insight into how farmers can be expected to use the Single Payment (SP). The research found that some of the normative influences on behaviour are not statistically significant. However, study of the motivation to comply and of respondents’ subjective beliefs identified that, in respect of use of the SP within the business, (a) the family was dominantly the major influence on their planned behaviour; (b) the farmers were least likely to comply with what Defra, consultants and land agents might suggest; (c) they were not likely to follow the views of other farmers except for those they were closely associated with. Attitudes, perceived behavioural control and the views of others were all found to have significant influence on farmers’ behavioural intentions regarding the SP.

All of these conceptual frameworks (the TDI, the TRA and the TPB) have been found to have predictive or explanatory robustness in various studies of the agricultural foodchain sectors, though none have been found to perform equally well in all circumstances. The key point here is that the tools exist for a more rigorous exploration of farmers’ intentions to adopt new technologies, in this case biofiltration systems for the control of diffuse water and air pollution. This review of the methodological background to the socio-
economic study of farmers’ behaviour in respect of the adoption of innovations or new technology points to the potential value of undertaking such a study in order to improve the process efficiency of technology adoption in this area.

A pragmatic, empirically based approach to classifying the non-adopters of agri-environment schemes has been developed and described by Fish et al., (2003). With this scheme (described briefly below in the context of biofilters) it is assumed that a better overall uptake will result if the approach to encourage adoption is tailored to the underlying reasons for non-adoption.

2.4 Non Adoption: ideal types and potential extension activities

It is against this backdrop of economic and non-economic concerns that this review suggests four ‘ideal’ types of biofiltration ‘non-adopter’. Ideal types are designed to help Defra begin discriminating between potential entrants into the biofiltration market place and help shape the targeting of appropriate extension techniques. The four ideal types identified here draw particular inspiration from research into a related area of Defra’s work, the uptake of agri-environmental schemes (e.g. Fish et al., 2003). All types represent a particular combination of capacities and willingness to adopt and are represented diagrammatically in Figure 4 below.

The first ideal type is termed a Disinterested Non-Adopter (DNAs). Like their attitudes to the wider policy arena for environmental protection and enhancement, DNAs are generally defined by highly reticent attitudes towards biofiltration. Such practices are automatically discounted because they are perceived to be obstacles to the carrying out of core tasks, and further, will be understood to be directly affect the viability of enterprise because of presumed capital and labour inputs. DNAs are likely to made up of both intensive enterprises, and smaller, more marginal, units, where the need to achieve short term productive efficiencies is deemed incompatible with interventions of this kind. This world view of DNAs towards biofiltration is likely to be reinforced by endemic attitudes to changing state visions of the countryside and the less assured place of farming within them.
The second ideal type is termed a Reluctant Non-Adopter (RNA). RNAs are effectively DNAs who turn to techniques of biofiltration because ‘needs must’. RNAs are largely a reactive constituency of farmer who finds pathways to these systems because they faced with the prospect of statutory intervention and penalties. Existing systems of production are likely to be highly intensive, and sited in areas where air and water borne pollution are prominent either from the point of view of regulatory bodies or local residents. RNAs will have exhausted low cost and conventional responses to their pollution problems and may become unenthusiastic adopters of these techniques.

The third ideal type is termed a Sceptical Non-Adopter (SNA). Unlike the first two types, SNAs have generally positive predisposition to issues of environmental protection and enhancement, but they will need some convincing regarding these techniques ‘fitness for purpose’, both in terms of their compatibility with existing practices and their affect on the ‘bottom line’ of financial return. SNAs will recognize the intrinsic gains that come from adoption systems of biofiltration, but they are driven primarily by questions of expediency.

The fourth ideal type is termed an Enthusiastic Non-Adopter (ENA). ENAs are willing converts to the idea of biofiltration and are likely to have had a history of pro-action in the environmental arena. While ENAs rely on a convincing evidence base, they are also experimental within their professional communities. ENAs understand well the synergies that may be realised between the economic, environmental and social benefits of adoption. They
are deeply embedded in the proscriptions of agri-environmentalism both through industry led quality assurance systems and state-led environmental stewardship schemes. ENAs recognise the virtues of leading in these areas: adoption minimizes the chances of statutory inventions in enterprise practices; adds a price premium on commodities; increases esteem with peer and public network, and carry with it intrinsic benefits for human and environmental well being.

While each these 4 ideal types are hypothetical, and require empirical testing in terms of prevailing real world attitudes and circumstances, they form a basis upon which extension activities for encouraging the uptake of biofiltration may begin to be envisaged. In particular Enthusiastic Non-Adopters represent important target group because they can be instrumental in winning the hearts and minds of Sceptics. It is likely that many SNAs will have a “if you will, I will” mentality, and may be strongly influenced by those who pioneer in this area. Importantly, because ENAs are likely to be closely aligned to networks of environmental innovation and good practice, it appears important to capitalise on these networks as opportunities to pilot these systems as part of integrated systems of land and livestock management. The relationship to quality assurance systems and schemes seems crucial here. These can act as mechanisms by which the performance of biofilters can be validated objectively and would increase purchasers’ confidence in them. These processes should feed into wider networks of professional self-learning, for as a recent farm practices survey suggests (2004) the most common ways for farmers to become actively involved in the improvement of water quality is to attend a discussion group or go on farm visits (more than 20% for both), or otherwise to visit demonstration farms (15%). These environments become opportunities to both ‘test case’ and ‘show case’ techniques and thus serve as contexts in which their viability – in terms of know-how and practical farm economics – as well as reputation, can be enhanced. In contrast, the case for adoption among the reluctant and disinterested is likely to be best made in the context of regulatory codes and the propagation of biological techniques within the written materials and enforcement networks that underpin them. In all of this need for developing fiscal instruments that support capital investment would appear to be crucial, especially in the context of airborne biofilters.

2.5 UK research on communication methods for changing farmer behaviour

In 2001 Defra commissioned a major study of ‘Communication methods to persuade agricultural land managers to adopt practices that will benefit environmental protection and conservation programmes’, which ranged widely over the industry’s business and regulatory environments, farm decision-making, communication methods and the appropriate public policy response, drawing on international examples [Dampney et al, 2001]. However its principal output was a framework for effective communication with farmers which was set in the context of a detailed discussion of how to achieve behavioural change in farmers. The study is recent enough to have continued relevance and, in light of the objectives for the present study with regard to achieving behavioural change and communicating effectively with farmers, it is appropriate to review those findings which are relevant to the introduction of
biofiltration techniques in UK agriculture. The following sections reproduce much of the study’s executive summary.

2.6 Business context and farm decision-making

A review of studies on contextual factors and the factors influencing farm decision making concluded that the business context in which the industry operates has an important effect on likely responses to pressures for change. Historically low farm incomes at that time meant that income generation and diversification are high priorities for farmers. Political, economic, cultural and physical factors are important forces that can encourage change, but economics is of paramount importance.

While positive attitudes influence the adoption of good practices, the awareness and acceptance of less visible problems such as diffuse pollution is low. Moreover, farmers are heterogeneous in that they have different information needs and use strategies, as well as variable personal and psychological characteristics. In general farmers have a high average age and may thus be less willing to change or adopt new approaches. Conversely, younger, better educated farmers and those with more farm resources generally show more aptitude to adopt good practices.

A range of on-farm and off-farm factors influence farmer behaviour including personal, psychological, sociological, family, business, economic, institutional, innovation and practical factors. In particular, the availability of information and an ability to use it at farm level are inter-dependent factors which affect farmer behaviour. Often, however, other factors (e.g. economics, labour, farmers’ knowledge and skills, quality of advice and support, risk) will constrain the uptake of advice and information.

Farmer use of information is based on quality, cost, accessibility, accountability, timeliness, reliability, relevance and habit. Farmers usually combine information from different sources and add this to their own knowledge when formulating decisions. Farmers see that guidelines and regulations need to be reasonable and practical to be acceptable. Self audit and assurance schemes exist which can encourage or require farmers to consider their own management practices, and encourage changes where needed. Farmers appear to be more aware of the more established agri-environment/conservation programmes and nitrate schemes than other free advice programmes.

2.7 Communication methods and public policy

A review of studies on communication methods and the response of public policy concluded that information and advice is of the greatest benefit when targeted at the right group at the right time, with mass media creating awareness in the early stages and face-to-face contacts becoming more necessary as appreciation of an issue increases. Farmers need to experience visible environmental problems, recognise causes and observe effective solutions.
Face-to-face advice is particularly appropriate when it follows other awareness-raising communication methods. Face-to-face farm visits by advisers are valued by farmers and providers particularly for providing interpretation of technical/regulation information in the context of the individual farm. Credibility, reliability, trust and affordability are important factors to farmers who use advisers. The loss of ADAS as a truly impartial extension service available to all producers, and providing a wide range of skills within one umbrella organisation, is seen as disruptive.

Farmers attribute different values to different communication methods and deliverers based on experience, habit and loyalties. The mode of delivery of information/advice needs to be appropriate for the target audience and for the stage at which it is delivered; message content and tone is also a consideration.

Other farmers are a common source of information for farmers especially in the early stages of awareness-raising but are not particularly valued. Farmers use and value their own experience often above other sources. Furthermore, the use of discussion groups is increasing although usually these involve the more progressive farmers. Farmers have mixed views of their value.

Well planned demonstrations on commercial farms, which provide financial as well as technical data, are valued by farmers and providers. Demonstrations are particularly useful if followed up with adviser visits. Representatives from agrochemical companies are a common source of information, although this rarely includes environment protection or conservation management advice. Opportunities exist for tapping into the well established flow of information from farmer financed research stations to farmer. Uptake of IT by farmers is slow although there are opportunities for farmer networks using IT if good co-ordination is available.

The farming press is popular and has a large circulation. However farmers’ opinion of its value is mixed; TV and radio are well used and valued. The effectiveness of leaflets could be increased if this was co-ordinated among agencies. Technical information provided by Government bodies is criticised as sometimes too long and complex. Messages need to be relevant, not too simplistic or over technical, and not patronising in tone.

Lack of a national focal point for agricultural and rural information has been highlighted, while competition between farmers and information providers is seen as hampering the more effective sharing of information. Crucially, information on the economics, policy and administrative details of Government schemes is as important as the technical details.

Finally, clearly defined criteria are needed to measure the effectiveness of communication. A framework for measuring communications success is suggested covering decision to adopt, successful application, on-farm multiplier and off-farm multiplier.
2.8 Principles for effective communication

The principles for effective communication are outlined and these may be summarised as:

i) The aim of communications is to help move a potential customer or target audience from a state of ignorance towards a position of decision and action. An adoption process of awareness, interest, desire and action is well documented. A good communicator will concentrate on the receiver not their own aims.

ii) Farmers (like any other market) can be segmented to help ensure that the optimal mix of communication methods is used for each market segment. Segments should be unique, sizeable, accessible and with a characteristic difference-responsiveness to marketing. There is currently no adequate segmentation of farmers to meet EP or CM purposes.

iii) Communication methods should be evaluated according to the size of the target market; its likely impact; the nature of message; the desired coverage and penetration; any negative characteristics; the positive characteristics; their cost; their speed; their complexity and convenience; the feedback; the creative scope.

iv) Communications planning is essential before a campaign. A 10 step process is outlined including objectives; issues; strategy; audiences; messages; plan; timetable; budget; measurement; resources.

v) A communications plan for a virtual catchment is presented (see Dampney et al, 2001).

These findings and guidelines are of direct relevance to the issue being addressed in the present project of increasing the adoption of biofiltration techniques in UK agriculture. In essence the challenge now is of a similar order to that addressed in the 2001 study: both concern aspects of ‘greening’ agricultural practice to provide improved common benefits; both involve introducing often significant changes to established practice in commercial agriculture; both have economic impacts on the business; and, to a large extent, both involve the adoption of practices which have already been pioneered.

However, it is likely that the paramount role of the relative state of the business economics of farming, highlighted in the 2001 study as an important issue affecting the adoption of new practices, remains an important factor at the present time given recent trends in the farming industry’s terms of trade (add ref). Moreover, it is acknowledged that changing farmers’ behaviour to tackle less visible problems such as diffuse pollution is much more difficult than where the issue can be clearly identified and, for example, linked to good farming practice. The need for targeting in any communication initiatives must include both the message and the method, together with the identified segments of the farming population.

This suggests that the most effective approach to this issue needs to combine a sound understanding of the factors which will govern farmers’ behaviour change with respect to the adoption of biofilters, in conjunction with a carefully
designed communications programme tailored to the appropriate segment of the farming population (e.g. taking into account factors such as farm system, farm size, location, etc)

2.9 Communication methods in practice

There are a very large number of organisations involved in providing information and advice to farmers (several hundred including private companies). There are concerns that this may be leading to information overload to farmers. Ten categories of organisation can be identified – agricultural supplies; education and training; farmer representatives; financial institutes; food processors and retailers; Government/Agencies/NGOs/local authorities; private sector farm service companies; research organisations; voluntary bodies; water industry.

A commonly preferred approach to persuade farmers to adopt improved farm practices is to start by creating interest and awareness using simple leaflets and mass media communication methods. One-to-group activities (e.g. discussion groups, demonstration farms) are considered to be effective methods once interest has been generated. Direct contact between farmers and advisers is widely considered to be essential in order to persuade farmers to actually implement change. One to one and one to group methods are widely used to achieve this.

More information is needed on the economic implications of new practices. Farm audits and assurance schemes are regarded as useful mechanisms to encourage adoption of ‘best practice’ on farms. There is a plethora of publications available to farmers and rationalisation is desirable.

Organisations involved in knowledge transfer are increasingly keen to work together to provide integrated advice with the aim that this is seen to be credible and trustworthy. There are concerns that some organisations have agendas which are too narrow. Professional competency standards need to be extended to cover CM and aspects of environmental management not currently covered. Farmers should be more involved in deciding on how best to provide them with information and advice.

Defra should develop and manage a centralised (web-based) source of information, agreed messages and contact points for further information that can be accessed (electronically) by all organisations involved in knowledge transfer. Further, Defra should co-ordinate and support the activities of intermediary organisations providing EP and CM information and advice, including campaign management and funding support to campaigns. Finally, Defra should take the lead in achieving industry wide agreement on key messages.

Farm advisers and others with direct contact with farmers need to be better able to provide information on a wider range of EP and CM issues, including sign-posting to other sources of help. In this respect, further training of advisers is needed.
3. References


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