Review of Eutrophication Control Strategies
Draft Final Report

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1. INTRODUCTION

Eutrophication is the general term given to the increase in concentration of plant nutrients in standing waterbodies and watercourses. The biological effects are manifest primarily in increased biomass of plants together with secondary responses of the primary producing community such as shifts in relative abundance of species and taxa through competition for nutrients or other limiting resources (e.g. light). This is followed by food-chain responses to direct (e.g change of food organism abundance) and indirect (e.g. altered oxygen regime caused by decay of plant biomass) effects of the changed primary production base. Publications which explain in detail the causes and effects of nutrient enrichment in rivers, lakes and coastal waters are Henderson-Sellars and Markland, (1987), Golterman and de Oude, (1991) Harper, (1992) and in more direct applied fashion, Cartright et al., (1993).

A decade ago the general consensus in water management was that eutrophication was a minor problem in the UK outside particular wetland areas such as the Norfolk Broads and the Shropshire/Cheshire Meres (Collingwood, 1977). This opinion was not so prevalent in continental countries. Perspectives on eutrophication, however, changed rapidly in both north and southern hemispheres following long hot summer periods in 1989, which led to pronounced blooms of cyanobacteria dense enough to cause toxic scums (Anon., 1990), (Anon., 1992). Eutrophication is now recognised as a widespread problem for all countries with developed agriculture and urbanisation, after for example, (Vollenweider and Kerekes, 1982).

1.1 Philosophy of the review

This review starts from the recognition that eutrophication is caused by elevated nutrient input from a variety of sources in combination (natural background, atmospheric deposition, industry, agriculture, domestic); in a variety of manifestations (point, diffuse); leading to situations where different limiting factors act upon ecosystem productivity and stability (phosphorus, nitrogen, light). The control strategies adopted therefore have to be holistic, structured in a hierarchical fashion from catchment-level downwards, and linked with other management strategies which initially might have had different objectives (e.g. erosion control, wetlands, buffer strips) (Straskraba, 1994).

1.2 Consultation

During the course of the contract it was possible to consult with active scientists in seven european countries to gain a 'state-of-the-art' perspective on eutrophication control science. Dr De Bernardi, Director of the Italian Centre for Limnology in Pallanza on Lake Maggiore, contributed information on current research in Italy in the fields of biomanipulation and lake restoration. Dr Pagnotta, Director of the Italian Water Research Institute and Dr Tartari, Head Scientist of the Chemical Laboratory, in Milan provided unpublished reports on eutrophication management in the river Po and its delta. In Anguillara, Rome, at the National Research Centre for Energy and Environmental Technology (ENEA), Dr Izzo, Head of the Microbial Ecology Department discussed recent developments in the control of eutrophication in Mediterranean lagoons (the lagoon of Orbetello, the lagoon of Venice, the Pontine lagoons). Electronic contacts with Dr Sechi of the Ente Flumendosa (Cagliari, Sardinia), a state-owned institution for the management of the Flumendosa catchment reservoir cascade which received a grant from the European Community (DG 11) for its multidisciplinary research activities on the prevention of the effects of eutrophication in drinking water reservoirsyielded a report and final recommendations from the three year project concluded in 1993.

In the Czech Republic discussions were undertaken with research teams involved in the monitoring and recovery of eutrophication in Czech reservoirs (Drs. Straskraba, Hejzlar, Matena, Vyhnálek, Seda, Kubecka). This visit included the collection of several internal reports, published and unpublished documents both in Czech and in English. A visit to Drs. Pokorny and Pechar of the Hydrobotany Department in Trebon produced information on the successful recovery of Czech fish culture ponds by sediment removal. In Prague Dr Korínek was consulted on recent advancements of a European Union financed, international research project on new biomanipulation techniques aimed at the recovery of shallow water bodies.

In Germany, Profs. Uhlmann and Benndorff of the Technical University of Dresden provided information on the most recent advancements in nutrient immobilisation techniques in reservoirs and biomanipulation, integrating the published work of their Limnology Department with unpublished data and internal reports both in English and German. Dr Ripl at the Technical University of Berlin was consulted on lake management and restoration methods in which he has long-standing experience.

Dr Eddy Lammens at RIZA (the state water research laboratory of the Netherlands), Dr Eric Jeppesen at the Danish Environmental Protection Agency, Dr Rheinhard Heerkloss at the University of Rostock and Professor Zalewski at the University of Lodz, Poland, will be consulted during a second mission in early April and the conclusions incorporated into the final report.

Consultation within the UK was less extensive, because the major scientists and research institutes have published reviews or reports within the last three years. In particular the national scene was reviewed in 1992 by the Royal Commission on Environmental Pollution (Anon, 1992). Two reviews of nutrients have been contracted by major charities (Stansfield and Moss, 1992), (Irving, 1993) and three recent reviews on lake management have been carried out for the water supply industry (Parr and Clarke, (1992), Parr, (1992), Cartright et al., (1993)). Three publications have also been produced by the NRA in the field of biomanipulation (Anon, 1994), (Phillips and Moss, 1993), (Reynolds, 1994); two on lake classification (Moss, et al., unpublished), (Johnes, et al., 1995) and several by the NRA in conjunction with other river management agencies on estuarine/coastal water eutrophication (Parr, et al., 1993), (JONUS, 1994), (Anon., 1994), (Elliot, et al., 1994). British and European agencies have also reviewed the subject area: English Nature from the perspective of conservation standards (Mainstone, et al., 1994) and selected SSSIs (Moss, et al., 1993), 'Sniffer' (Scotland and Northern Ireland Forum for Environmental Research) from the perspective of risk assessment (Johnson, et al., 1992) and SCOPE (Scientific Committee on Phosphates in Europe) from the perspective of remedial measures (Johnston, 1994). Coastal eutrophication is of concern to other countries with semi- enclosed marine waters including the Baltic (Moller Andersen, 1986) (Anon, 1991) and Adriatic (Italy) (Vollenweider et al., 1992).

River eutrophication is a more nebulous issue than lakes, often because rivers are merely considered as transporters of nutrients to standing waters, or with problems manifest most obviously in the lower reaches where they become similar to long thin lakes (Descy, 1992). Almost nothing is known of the effects of nutrient enrichment separate from organic pollution on rivers (Sweeting, 1994) with the exception of specific work such as (Holmes and Newbold, 1984), (Mainstone et al., 1994) directed towards plant communities. Separate consideration is currently being given to rivers in a literature review of the OI project 'Mesotrophic Rivers in Anglian Region' and to biomanipulation in the OI feasibility study 'Biomanipulation of Deep Lakes and Reservoirs' and so the important summaries will be merged into the final report of this project. The large amount of 'grey literature' in the listings of the United States EPA has been scanned and a number of pertinent reports of the EPA on lake and nutrient management reviewed (e.g. (Moore and Thornton, 1988)).

2. THE PROBLEM OF EUTROPHICATION

The problems of eutrophication are primarily caused by excess quantities of nitrogen and phosphorus; both draining from agricultural land, urban areas, and as point sources of industrial, agricultural and domestic sewage. In almost all standing water situations in the UK, phosphorus is the nutrient limiting primary production and so whose concentration is of most concern, but for short periods of time where phosphorus has raised production above background levels, silicon (in spring) nitrogen (in summer), and light (in summer) may be limiting (Harper, 1992).

2.1 Causes

Conventionally we have assumed for several decades that nitrogen is primarily derived from agricultural land and phosphorus from urban effluents, the origin of the latter approximately 50% human and 50% detergent. Even rural catchments derive much of their phosphorus loads from a multitude of small domestic sewage effluents (Turner, 1993). There is increasing evidence however, that phosphorus loadings from diffuse agricultural sources are increasing, e.g. (McGarrigle, 1993), due to association with soil particles in erosion (Sharpley and Smith, 1990), increased stocking rates (Wilson et al., 1993), runoff of applied animal-derived slurry and saturation of the soil-binding capacities through continuous use of phosphate-based fertilisers (Sharpley et al., 1994). Moreover, at times of the year when low flows persist, nitrogen loading from point sources can be considerable - over 50% in one study (Pedersen and Petersen, 1992). In rivers generally, rural point sources (small village sewage works) may be important, and in estuaries and coastal waters effluent discharges may achieve a high importance because the absence of biological treatment stages results in higher loadings of nitrogen (as ammonia) as well as phosphorus.

It is prudent for this review to consider phosphorus and nitrogen both as limiting nutrients. Phoshorus as the usual nutrient in shortest supply has already been mentioned above but some areas of the country naturally rich in phosphorus show nitrogen limitation in summer, e.g. the West Midlands meres (Moss et al., 1993), and coastal waters are often nitrogen limited due to phosphorus recycling from anoxic sediments (Anon, 1991) but nitrogen removal by denitrification (Smith, 1984). Moreover, removal of phosphorus from point-sources or recipeint ecosystems will increase the probability that nitrogen becomes limiting and the two should, therefore, be considered together.

2.2 Effects and Manifestations

Eutrophication is currently only a problem in water management where enhanced nutrients cause ecological changes. The totality of the changes are complex and not yet fully understood, even though we use our current knowledge in aspects of eutrophication management – 'ecotechnical' or 'biomanipulation'. The only area where nutrient levels *per se* give rise to concern without ecological change being important, are where nitrate levels become elevated in drinking water supplies or potentiual drinking water supplies, as there is some possibility that nitrate may be dangerous to human health under certain circumstances (infantile methaemoglobinaemia and nitrosamine-induced cancer). Phosphate in water is not considered a hazard to health under any conditions, and so no standards for drinking wtaer exist. The ecological concern about phosphorus stems from its role in promoting excessive primary production and biomass, and the current high level of concern from the toxic nature of some of the microbial components of this combined with the deoxygenation which results from biomass decay. There remains the possibility that at some future date economic concern over the dissipation of the finite element phosphorus might make recovery technologies important in their own right (Morse, 1993), but this issue is not considered further here.

In all environments the major manifestations of eutrophication are enhanced plant (usually algal) biomasses concentrated in a few 'nuisance' species. In rivers different circumstances may produce enhanced macrophyte growth - usually shallow slow-flowing waters – and in standing waters nuisance macrophyte growth may be an intermediate stage in progressive eutrophication or recovery. The most widespread indication in rivers however, is enhanced growth and cover of the macroalgae *Cladophora* and *Entermorpha* (Fox and Malati, 1985), with phytoplankton in the semi-impounded lower reaches (Marsden et al., 1995). In lakes and coastal waters phytoplankton algae are the primary manifestations (Anon., 1992), with mats of benthic macroalgae on lake muds where light penetrates or coastal inter-tidal mudflats (Anon, 1991).

3 CONTROL OF EUTROPHICATION – GENERAL PRINCIPLES

The principles of eutrophication control are primarily the removal of the causes; scondarily the management of effects and thirdly the removal of manifestations. The assumption is made that removal of the cause is preferable, unless prohibited by political, economic or temporal constraints in which case management of the effects is the objective. In extreme cases removal of the manifestations is also an option.

3.1 Introduction

All methods which have a scientific basis are being reviewed for the efficiency, impacts on the ecosystem, and effectiveness in terms of their original objectives. Methods which are purely political or purely technical (e.g. scum removal by boom) are listed but not being reviewed further as these are part of the separate economic evaluation.

3.2 Hierarchy of Methods

Methods for control of the sources of eutrophication were initially considered under a hierarchy of scale; from trans-catchment methods for nutrient removal, to catchment-wide methods, and point-source removal. Methods for control of the manifestations were considered under recipient ecosystem nutrient removal techniques, or recipient ecosystem nutrient management techniques. The full list is contained in the first progress report.

Trans-catchment methods for nutrient removal or reduction - such as removal of phosphorus in detergent formulation (Morse, Lester et al., 1994), re-formulation of fertilisers or alteration to economics of fertiliser use and replacement of water flushing toilets by dry composting types (Hindmarsh, 1992), were not considered further in this report. This is because methods of such scale are political considerations and better suited to the business needs and cost-benefit aspects of the development of national NRA strategy.

4. CONTROL OF EUTROPHICATION – CATCHMENT-WIDE METHODS FOR NUTRIENT REMOVAL OR REDUCTION

4.1 Diffuse catchment sources

Control of diffuse sources of nutrients from agricultural catchments can broadly be divided into three parts: those methods widely applicable to agricultural land; those methods or adjacent to, watercourses and those methods focused within the river channel.

4.1.1 Reduction of nutrient loss from agricultural land

Methods to reduce input of nutrients to watercourses from agricultural land need to be applied on a fine, field-by-field, scale, and addressing the nature of nutrient pathways. There are three main mechanisms of nutrient transport in water; surface run-off, subsurface run-off and underdrainage and sediment transport.

Subsurface flow is the main transport mechanism of soluble nutrients especially in the winter when the water table approaches the soil surface. Subsurface drains and secondary drainage conduits are an important pathway for nitrogen movement in catchments.

Concentrations of nitrogen between 30 and 50 mg l^{-1} have been recorded in winter in the run-off along artificial subsurface drains (Muscutt et al., 1993).

Surface run-off occurs when the surface of the soil becomes saturated and when rainfall intensities exceed the infiltration capacity of a soil. If a run-off event (such as a storm) occurs following the application of animal wastes or fertilisers to the soil, then surface flow will be the main transport mechanism for soluble pollutants. The problem may be intensified where farming practices have compacted the soils, and also in coarser textured soils which generally exhibit low stability of soil matter (Chambers et al., 1992).

Organic and inorganic sediment movement is a major transport mechanism for phosphorus (Sharpley and Smith, 1990) as well as for organic nitrogen (sediment nitrogen may constitute the greatest proportion of the total nitrogen load) (Muscutt et al., 1993). The risk of erosion is greatest on cultivated soils, particularly when the land is bare. The risk of erosion is usually concentrated in areas such as hillslope concavities, especially on coarse textured soils (Chambers et al., 1992).

A considerable body of evidence, reviewed in (Harper, 1992) and (Johnes, 1995) has developed which accurately estimates nutrient runoff from different kinds of agricultural land based upon 'export coefficients' for the land use type and features of the catchment hydrology. A catchment profile can thus be produced as part of an overall catchment management plan, for example as recently done for the Slapton Ley catchment (Wilson et al., 1993) and the Windrush catchment (Johnes and Burt, 1991). The possibility now exists that satellite imagery could automate and speed up this process, by providing an instantaneous analysis of land use categories for a whole catchment (Wooding et al., 1994; Wright et al., 1994) to which export coefficients could be computed.

Reduction of nutrient inputs from agricultural land is difficult but not impossible, as it can utilise and build upon initiatives already started. These have arisen from the Ministry of Agriculture and latterly the European Union – as Codes of Practices, as the Nitrate Sensitive Areas Scheme, as the Environmentally Sensitive Areas Scheme, as the Set-Aside Scheme and as Extensification. Two aspects of their structure need further attention: the first is the inclusion of phosphorus into those schemes which explicitly deal with water and the second is the education of water quality and agricultural personnel to appreciate the ubiquitous nature of eutrophication and the large number of small steps which are needed to deal with it.

Evidence has been accumulating for the past decade of so that phosphorus from diffuse sources (i.e. from agricultural drainage rather than from industrial or domestic discharges) is increasing in importance (Johnes, 1995). This is particularly true where point source reduction has been achieved. In the catchment of Lough Neagh, (Foy et al., 1995) have shown a steady increase in the phosphorus from agricultural difuse sources, around 1.5 kg P km^{-2} yr^{-1}, and a Swedish study has shown similar results (Krug, 1993). The explanation for this is that, although annual application of phosphorus in fertiliser is not increasing, only a relatively small proportion of applied phosphorus is taken off in the crop each year. IN consequence there is a slow rise in the soil phosphorus content. As the total soil phosphorus increases, the amount of soluble phosphorus lost in drainage also increases. The Lough Neagh study has suggested that this need only be of the order of 2 μg P l^{-1} yr^{-1}, but that this is a figure which achieves a limnological significance over the time span of some 20 years.

Sharpley et al., (1994) have suggested a series of steps towards the management of excess phosphorus runoff. The first is to identify higher phosphorus source areas within a catchment. This can be done using a phosphorus-indexing system, based upon the susceptibility of the soil to erosion and to runoff, together with its soil phosphorus content and the nature and amount of added fertiliser. A simple index such as this allows field agricultural or water quality personnel to identify areas where practices most threaten eutrophication of watercourses. The second step is then to identify and target agricultural practices which are most contributing to the losses and effect remedial measures. Similar conclusions about the need for targetting were reached in the UK for potential nitrate contamination of water supplies (DOE, 1986), (STWA, 1988). Measures which are standard practice for minimising erosion and water pollution risks – contour ploughing, avoidance of bare soil, timing of manure applications as produced by several countries (MAFF, 1991; MAFF, 1993) for the UK, Kunkel (1989 in Klapper 1991) for Germany) – will minimise excessive nutrient runoff but more detailed methods ar also necessary to decrease eutrophication. These are often crop- and site-specific and may be contradictory when considered generally to different practices associated with arable farming. For example the use of no-tillage systems for wheat cultivation may decrease erosion and soil phosphorus loss but increase soluble phosphorus lost through leaching from crop residues left on the surface. Optimising vegetative ground cover may conflict with 'knifing' or sub-surface injection of fertilisers. Hence a catchment-based approach is necessary to eliminate such conflicts by selection of the optimum method for the particular kind of agriculture.

More accurate methods can be targetted at manure management, although here there may be greater conflict with economic requirements because manure tends to be over-rich in phosphorus with an average N:P ratio of 4 compared with crop requirement of around 8 (Sharpley et al., 1994). In specific catchments of the US where this problem is being tackled the main objective is to manage application rates based upon soil phosphorus concentration (rather than crop nitrogen needs), to manage cattle densities and cattle feed rations, and to limit cattle access to watercourses. It is probable that much agricultural pasture with manure applied on the basis of nitrogen requirements is saturated with phosphorus and that regulation of mature application on a national scale will be a major challenge in the next decade (Sharpley et al., 1995).

Inherent in any approach to management of phosphorus in agricultural catchments is the acceptance by agriculturalists that phosphorus is a pollutant and that restrictions on agricultural practices which cause phosphorus loss (or financial incentives for those which conserve it) are necessary. At present, the Codes of Good Agricultural Practice (Heathwaite et al., 1993), (Parkinson, 1993) are directed at prevention of pollution rather than prevention of eutrophication and have minimal effect on reducing the latter. Such acceptance has been more readily achieved for nitrogen because its (albeit small) health risk has resulted in the establishment of WHO standards and a number of initiatives designed to reduce nitrate concentration in water supplies. Nevertheless, the same changes in agricultural practices will reduce both phosphorus and nitrogen runoff into watercourses. (Johnes, 1995) has used export coefficent models constructed for the Windrush catchment to predict the effects of different land use changes on future nitrogen and phosphorus runoff. Catchment-specific changes, involving reduction and relocation of the highest nutrient exporting arable farming practices, might reduce stream nutrient loads by around 20% whilst the application of the Premium Scheme Nitrate Sensitive Area scheme (conversion of arable to unfertilised grass) would yield between 60-80% reduction of both nitrogen and phosphorus. Other changes, such as the NSA Basic Scheme reduction in fertiliser use, produced lesser reductions in runoff (It is worth noting here however that the Premium Scheme NSA was largely unsuccesful in those areas where NSA was introduced (O'Riordan and Bentham, 1993)).

Johnes (op cit) refers to changing the location of particular agricultural crops based upon the soil characteristics; high cation exchange capacity for example and low slope makes soils more retentive of nutrients. This is a parallel aproach to Sharpley's (op cit). Johnes makes the point however, that for the Windrush catchment such soils are predominantly in the floodplain and thus closest to the surface drainage network. This brings into consideration the protection of surface water drainage courses by suitable vegetation which absorbs, or 'buffers' runoff from more susceptible soils and practices.

4.1.2 Riparian buffer zones for water course protection

Concepts of riparian zones protecting watercourses arose about twenty years ago in the US and Australasia (McColl, 1978) and have developed rapidly since then. Buffer zones are vegetated areas extending a prescribed distance from the water's edge for the purpose of protecting the quality of water (Nieswand et al., 1990). The concept is that buffer zones provide both a biochemical and physical barrier between the pollutant source areas and the receiving waters (Muscutt et al., 1993).

Buffer zones alter the flux of nutrients in two main ways. Firstly, they enhance the retention of nutrients during their cathment to watercourse movement and secondly they reduce the the opportunity for direct contamination during agricultural activities (i.e. ploughing, fertiliser application). For example, bank erosion in Canada has been calculated to account for 32% of sediment and 10% of the phosphorus inputs to water from agricultural catchments. Buffer zones decrease bank erosion in at least two ways. The erosivity of the watercourse may decrease due to the increased channel roughness, and vegetation – particularly woody vegetation – may have a binding effect which is capable of increasing the area's stability (Muscutt et al. 1993).

Retention is achieved by the filtering effect of vegetation, reduction of surface flow velocities owing to the increased hydraulic roughness and infiltration within the buffer zone which reduces surface run-off and enhances nutrient retention in soil (Muscutt et al. 1993).

(Jordan et al., 1993) examined the changes in chemistry of groundwater flowing laterally from a cornfield, through a riparian forest, and subsequently to a stream, between July 1990 and August 1991. The forest was located on a hill slope and floodplain on the Delmarva Peninsula, near Centreville, U.S.A.. Groundwater moving through the forest towards a tributary of the Chester River was sampled at increasing distances from the field boundary. They found that the chemistry of the groundwater changed with distance. Nitrate concentrations showed the greatest spatial variation. This fell from 8mg l^{-1} at the edge of the cornfield to less than 0.4mg l^{-1} halfway through the forest. Most of this change was found to be 25-35 metres from the edge of the cornfield. Using estimates from hydrological modelling they concluded that up to 60kg of nitrate ha^{-1} yr^{-1} could be removed by the buffer zone.

(Uusi-Kamppa and Ylaranta, 1992) carried out comparative experiments to determine the effectiveness of buffer strips in preserving the water quality of a stream adjacent to cereal crops. They compared the effect of the agricultural areas upon water quality at sites with buffer strips and those without buffer strips. Throughout the year, the grass buffers decreased total solids in the run-off water by an average of 23%. The buffers also reduced the total phosphorus load by 6%. Nitrogen losses were reduced by 3.9 kg ha^{-1} and the losses of nitrate to the stream were half that of the plots without buffer strips.

Nitrate removal in riparian zones is often greater than phosphorus because of the involvement of microorganisms in nitrogen processing. There are three main mechanisms by which nitrates are removed from run-off water by buffer zones; uptake by the vegetation, microbial denitrification or microbial immobilisation (Groffman et al., 1992). Water-saturated, anaerobic, soil conditions facilitate denitrification. (Groffman et al., 1992) found that denitrification was greater in wetland areas but that the rates sharply decreased from surface to subsurface. (Haycock and Burt, 1993) demonstrated that the proces is independent of vegetation type and occurs wherever there is adequate carbon supply associated with low oxygen concentrations: undrained floodplain sediments removed 60-70% of nitrate in winter. Riparian zones where denitrification in winter can combine with plant uptake and microbial immobilisation in summer provide maximum removal opportunitiues (Haycock et al., 1993).

Vegetation tvpe

Haycock and Pinay (1993) assessed the effectiveness of forest and grass riparian buffer strips in retaining nitrates during winter months. The study focused on the change in groundwater nitrate content as the water passed through the riparian zones. The aim was to assess the dynamics of riparian zone retention and the processes which occur during the high nitrate leaching period of the year. The removal of nitrate was found to be directly correlated to its input, and the mechanism of removal was attributed to denitrification (very little uptake occurred out of the growing season). The forest buffer zone removed 100% of the nitrate load 5 metres into the buffer strip. Grass buffer zones removed 84% of the nitrate. The greatest reduction of nitrates occurred in the first few metres for both types of buffer zone.

(Osborne and Kovacic, 1993) compared grass and forest buffer strips. Forested buffer strips reduced nitrogen in the groundwater by 68-100% and in the surface run-off by 78-98%. Grass buffer strips reduced the nitrogen input by between 10-60%, but no clear pattern emerged for the effects on phosphorus in either the forested or grass vegetated buffer strips. In general terms, it was found that riparian forests are more efficient at removing nitrate from shallow subsurface water than grass buffer strips. They suggest that the reason behind this difference in the nitrogen removal efficiency of forest and grass buffer strips is associated with the form of the carbon available for denitrification. Muscutt et al. (1993) however reported forest buffer zones reducing the mean annual concentrations of nitrogen and phosphorus in surface runoff by up to 83 and 81% respectively. An important factor is the density of the ground vegetation affecting the ability of buffer strips to control inputs from surface run-off, and in particular sediment-associated phosphorus which are likely to be removed if the density of the vegetation is sufficient to cause deposition. Natural forests provide the greatest hydraulic conductivity of soils and the greatest resistance to overland flow hence retention of sediment (Peterjohn and Correll, 1984), (Yates and Sheridan, 1983). The correct buffer is one appropriate to the hydraulic needs of the system: certain grass buffers may offer sufficient resistance to encourage deposition and the removal of nitrogen and phosphorus on one hand whilst on the other the efficiency of artificial forests may be reduced if the degree of ground cover is too low, increasing surface run-off.

Buffer Width and Shape

One major problem is the assessment of just how wide a buffer needs to be in order to sustain a high level of effectiveness. As yet there are few studies which quantify the effectiveness of various buffer vegetation types and widths throughout the year. Pollutants transported in surface run-off may be retained by a relatively narrow buffer strip, however the retention.of fine particles is likely to be achieved only with a wider buffer strip, especially on dry soils. Studies on buffer strips protecting watercourses from other agricultural contaminants such as pesticides (Reynolds et al., 1993) suggested that 10 m was an adequate width, and the natural woodlands referred to above are almost completely effective at 20-30 m.

As well as width considerations, shape is also important. Simple linear buffer strips may not be the optimum design. Surface run-off is spatially intermittent and buffers should be oriented to take into account the path of the run-off water. Their design should also consider the pollutant transport processes that are likely to occur in specific situations (Muscutt et al. 1993). The pollution removal potential of a simple linear buffer strip would be limited if subsurface drains crossed the riparian zone for example. Subsurface drains decrease the rate of nutrient (especially nitrate) removal in two ways. Firstly, by lowering the water table, thereby limiting the frequency of reducing conditions which are required for denitrification. Secondly, through restriction of the time of contact between soil and water, thereby reducing the opportunities for nutrient by plants and bacteria (Chambers et al. 1992). The problems of subsurface drains in decreasing pollutant removal potential can be overcome by the creation of 'Horseshoe Wetlands' (Petersen et al., 1992), (Burt and Haycock, 1993) – semi-circular excavations at subsurface drain outlets occupied by vegetation.

4.1.2.3 Assessing buffer strip effectiveness

The conflicting results given in the literature abourt buffer strip functioning is largely caused by local differences in soil type, water table and fluxes, vegetation and topography.

There is little doubt that buffer strips are effective in removing a substantial part of incoming nutrients and that, on a catchment scale, they could produce the largest reduction in nutrient fluxes. (Wilson et al., 1993), using figures of 10-15% nitrogen and 20-30% phosphorus reduction, calculated that wooded buffer strips in the Slapton Ley catchment would be the most effective nutrient management measure, with 10 m wide strips reducing phosphorus loading to the Ley to 'permissable' under the OECD criteria and 15 m strips would reduce it to the borders of 'oligotrophic'. Current research underway funded by MAFF (Haycock, 1994) should clarify many of the uncertainties about buffer strips but in the present state of knowledge there is no doubt that well-conceived and constructed, they will provide the best nutrient protection for catchments from the available methods.

Several existing incentive schemes enable buffer strip establishment, and using general criteria such as 10 m minimum width, natural riparian woodland vegetation as optimum but grassland vegetation secondary and removal of underdrainage, their widespread will improve water quality. These schemes include (additionaslly to NSAs), Nitrate Vulnerable Zones, and Water Protection Zones and the Water Fringe option of the Habitat Improvement Scheme (Haycock, 1994), Extensification and Set-Aside (Burt and Haycock, 1993), and Environmentally Sensitive Area designation. These latter have been developed by MAFF under the Common Agricultural Policy; in addition the Countryside Commission in England has developed the Countryside Stewardship Scheme, which has a Field Margin option (6 m width along arable fields) and a Waterside Land option for re-creation of wetland and wet meadow grassland features. In general terms, each scheme which promotes good agricultural practices or reduce the overall intensity of agriculture, such as reduced fertiliser application, are likely to have a small impact on nutrient fluxes (Johnes, 1995) whereas those which promote field margins and buffer strips are, once widely implemented, more likely to have a substantial impact (Burt and Haycock, 1993).

None of the schemes above are managed by the NRA, but their diversity alone suggests that a first step should be their coordination within the formal Catchment Management Process, so that schemes initially designed for conservation, landscape improvement or reduction of surplus yields can also be made to produce the maximum reduction of nutrient fluxes down waterways.

4.1.3 In-channel mechanisms for nutrient retention

There is a basic incompatibility between modern land drainage practices which promote agricultural productivity on floodplain and catchment land, and the maximisation of rural water quality. Almost twenty years ago, (Karr and Schlosser, 1978) argued that, not only did 'near-side vegetation' (i.e. buffer strips) of headwater streams protect water quality from agricultural nutrients and sediment, but that natural channels with a pool-riffle morphology, showed around 25% lower Unit Stream Power over engineered agricultural channels; and a 28% reduction in suspended sediment transport due to lessened bank erosion occuring in the natural channels. The sediments of natural river channels possess considerably more heterogeneity of both particle size and depth, providing greater opportunity for infiltration and nutrient metabolism. Denitrification within river sediments; both flood deposition zones and gravel bars (Pinay et al., 1994) removes substantial amounts of incoming nitrate-nitrogen, and this process is enhanced by diffusion caused by invertebrate activity or plant roots (Cooper and Cooke, 1984), (Christensen et al., 1990), (Christensen and Sorensen, 1988). The imposition of natural debris dams in small streams adds a further dimension to these processes, because they reduce hydraulic throughput and hence increase contact time for processes such as denitrification, they enhance in-stream metabolic processes and hence phosphorus uptake by stream biota, and they provide the main natural source of carbon to maintain microbial metabolism and denitrification. Natural channel sediments in lower order headwater streams are therefore, highly likely to substantially reduce the downstream transport of particulate and soluble nutrients. The improvement of water quality which follows restoration of such channels degraded by engineering works has not yet been addressed.

In estuaries and coastal waters the greater area of shallow littoral zone produced by tidal movements plays a similarly important role in the natural ecosystem metabolism of nutrients, with denitrification removing as much as 50% of the nitrogen supply from freshwater flow and direct effluent discharges.

4.2 Point sources – rural

Control of nutrients by treatment of point sources is an essential method for most catchments. Rural catchments contain a multitude of village sewage works which are at present not covered by any nutrient reduction regulations other than the normal water pollution controls which require secondary treatment if communities are not served by household septic tank systems. In the rural catchment of Slapton Ley for example, 45% of the total phosphorus input was derived from small point sources; in the upper river Bure, Norfolk, 73% of total phosphorus was derived from small point sources (Turner, 1993).

Strategies which have been developed for nutrient reduction of such sources are: direct removal (if they are agricultural storage or effluents which can be diverted); chemical methods for nutrient stripping effluents; biological (microbial) methods for nutrient uptake during a secondary-tertiary oxidation stage; and artificial wetlands for nutrient retention and removal.

4.2.1 Nutrient stripping

Chemical methods for nitrogen removal are unusual, because most technical solutions for nitrogen removal involve biological nitrification followed by dentirification in different stages or locations of biological treatment (WPCF, 1983). Phosphorus removal on the other hand, is primarily by chemical means although biological methods utilising 'luxury consumption' physiologies of microrganisms provide an efficient alternative under the right conditions (Tracy and Flammino, 1987) can provide removal efficencies of up to 90% (Convertiet al., 1995).

Chemical methods for phosphorus removal are based upon the precipitation of insoluble salts with either calcium, using lime, or iron/aluminum using the ferric or chloride salt. Physical removal using reverse osmosis or ion exchange is also practiced less commonly (WPCF, 1983). One of these is usually introduced as a tertiary treatment stage on the end of sewage or waste treatment processes and the selection of which method, including a combination of biological and chemical methods (Cooper et al., 1995) depends upon the nature of the waste and the economics of individual sites and situations.

Point source removal of both phosphorus and nitrogen using biological or chemical processes is technically well-developed in countries of Europe and North America, capable of achiving effluent P standards of < 0.5 mg l^{-1} for example, in Sweden and 0.2 mg l^{-1} technically (Boers and Van der Molen, 1993). Removal from effluents as a catchment-level initiative has occurred on a limited scale in parts of the UK (Norfolk Broads, Lake District) and more recently from some large sewage works (over 10,000 population equivalent) as part of the preparation for the EC Urban Waste Water Directive. The problem of the UWWD directive is highlighted by the Great Ouse river system however, where nutrient removal from the major sewage works alone even if maintained will not restore the river system back from a hypertrophic state because of the large number of smaller works which discharge into the catchment.

4.2.2 Artificial wetlands

One suggested solution to the high capital and revenue expenditure required for phosphorus-stripping and nitrogen-removing tertiary treatment plants has been the development of artificial wetlands for effluent treatment. The use of natural wetlands has been studied for several decades in the US (Boyt, 1977), where several studies have demonstrated that wetlands, without any major modification of existing hydrologic conditions, act as sinks, transformers, and sources of chemicals, often with seasonal patterns. These are termed passive wetlands, as there has been no purposeful addition of wastewater, or active design of discharges into these wetlands.

Freshwater Marshes

One of the first studies that identified freshwater wetlands for their role as nutrient sinks was on Tinicum Marsh nr. Philadelphia (Grant & Patrick 1970) That study found decreases in phosphorus, inorganic nitrogen, and BOD as waste flowed through this tidal freshwater wetland. Lee et al. (1975) found that marshes acted as nutrient sinks during the summer and sources in the spring. Kitchens et al. (1975) demonstrated a significant reduction in phosphorus as waters flowed through a swamp forest in South Carolina. Kemp & Day (1984) and Peterjohn & Correll (1984) both demonstrated the roles of wetlands as systems receiving agricultural run-off. The former study suggested that a Louisiana forested swamp acts as a transformer system, removing inorganic forms of nitrogen, and serving as a source of phosphate and organic nitrogen and phosphorus. Van der Valk et al. (1979) concluded that all types of wetlands act as nitrogen and phosphorus sinks. Many of the other wetlands reported in the literature were found to act as a seasonal sinks only, large amounts of nutrients were exported at other times of the year (Sloey et al. 1978, Klopatek 1978, Lee et al. 1975, Kuenzler et al. 1980).

The dominant processes occurring resulted in the transformation of inorganic to organic material. Nichols, (1983) concluded that natural wetlands were efficient at removing both nitrogen and phosphorus (above 70%) at low loading rates (below 10 g N m^{-2} y^{-1}). Vollenweider (1968) estimated that about one hectare of wetland is needed for every 60 people for every 50% phosphorus removal. This estimation was established from removal rates of nitrogen and phosphorus from wastewater effluents (2.2 g P day^{-1} and 10.8 g N day^{-1}). Although high rates of nitrogen removal can be achieved at low application rates in order to maximise application rates, and nitrogen removal it may desirable to add additional carbon. A variety of exogenous carbon sources have been added to artificial wetlands in order to facilitate the denitrification process, with differing degrees of success. Gersburg et al. (1983) found that when methanol was added removal efficiencies of 95 % for total nitrogen and 97% for total inorganic nitrogen were obtained. However, due to the expense of methanol continued addition over an extended period is impractical. As a low cost, renewable alternative to methanol mulched plant biomass was applied to the wetland surface (Gersburg 1983). Removal rates of 87 and 91% for total nitrogen and total inorganic nitrogen respectively were achieved. It was established however, that the addition of mulched biomass did significantly increase the phosphorus pool.

Salt Marshes

Marshes have been widely regarded as sources and/or sinks in coastal marine nutrient cycles (Nixon 1980). Not all data support this view. In general, marshes seem to act as nitrogen transformers, importing dissolved oxidised inorganic forms of nitrogen, and exporting dissolved and particulate reduced forms. While the net exchanges are too small to influence the annual nitrogen budget of most coastal systems, it is possible that there may be a transient local importance attached to the marsh- estuarine nitrogen flux in some areas. Marshes are sinks for total phosphorus, but there appears to be a remobilisation of phosphorus in the sediments, and a small net export of phosphorus from the marsh.

It has been observed that some wetlands have received artificial loading of chemicals for many years and are still functioning as nutrient sinks e.g. (Nessel & Bayley 1984). Others suggest that continual chemical loading of wetlands will lead to a reduction of the wetlands efficiency in a process now called ageing. The ageing of natural wetlands receiving low loadings of nutrients, such as from non-point sources is not well understood.

Ecotechnological designs of wetlands

Over the past 15 years, the idea of purposeful application of wastewater to natural wetlands for water quality, has been explored with many experiments and demonstrations. The idea of using wetlands for receiving wastewater is as a treatment system; to others it is considered a disposal alternative. Wastewater recycling in wetlands has been experimentally investigated in a number of studies, many of which are summarised by Nichols (1983) and Godfrey et al. (1985). Many of the schemes in operation utilise natural wetlands, however it is also possible to construct artificial wetlands to carry out the same role. A constructed wetland is defined as a "man-made complex of saturated substrates, emergent and submergent vegetation, animal life and water " (Hammer & Bastian 1989).

The retention time is one of the most important variables in the use of wetlands as wastewater treatment systems (Hammer & Kadlec 1983). Landers and Knuth (1991) suggest that accompanying retention, optimal management models stress the need for maintaining a level of water in wetlands that is conducive to pollutant removal. Specific recommendations include; maintaining a water level high enough to cover the organic substrate (Sloey et al. 1978), while not so high as to provide insufficient contact of pollutants with pollutant-removing substrates (Kadlec & Kadlec 1978), performing site-specific studies to determine optimal water depth (Chan et al. 1982), and implementing physical methods of water level manipulation to maintain an appropriate depth for pollutant removal (Stockdale 1986, Canning 1988). Reuter et al. (1992) emphasise the importance of designing wetlands which provide a sufficient hydraulic residence time (HRT). The term HRT refers to the length of the period in which wastewater is present within the wetland. An HRT of more than four days is required to remove significant quantities of nitrogen (Reuter 1992). In a constructed wetland in Minnesota, which had an HRT of only a few hours, removal rates of 3% and 1% for nitrates and ammonia respectively were recorded (Reuter 1992). Stockdale (1986) recommended a retention period of 20-36 hours for most wetlands, while Ahern et al. (1982) suggested that a minimum of 5 days retention is required for pollutant removal, especially for phosphorus. Pride et al. (1990) also emphasise the importance of designing, to allow for maximum control of the hydrology in an artificial wetland. They found when constructing a wetland in Alabama, that such control was necessary in order to establish and maintain the growth of emergent vegetation. Gillette (1988) discusses wetland design considerations in a more general way. She quotes from Wolverton stating that the most effective wetland designs are based upon rock filters. These should be graded with the largest rocks in the receiving portion of the marsh so that during the summer months, when the sewage influent is algae laden, filter clogging is minimised. It is important when constructing wetlands to use native plants which are able to withstand the local climate (Gillette 1988). Reeds and duckweed, which are able to withstand cold winter temperatures, are commonly used (Gillette 1988).

Plant biomass was found to be the major nutrient storage compartment, with plant nutrient uptake being the primary removal mechanism Breen (1990) and absorption an important nutrient removal mechanism. Shijun and Jingsong (1989) also found that plants are a major agent in the removal of nutrients after decomposition by micro-organisms in artificial wetlands. Harvesting of the flora permanently removes the nutrients from the wetland. The removal of plants by harvesting, actually lengthens the ultimate life of an artificial wetland.

Gersburg (1985) found that for a constructed wetland in California, mean removal efficiencies for a year were 80% for total nitrogen and 80% for total inorganic nitrogen. This compares favourably with natural wetland removal efficiencies in relatively organic soils. Reuter et al. (1992) states that: "Gravel filled constructed wetlands ..... provide a much greater surface area for bacteria attachment than is possible in natural wetlands, thereby enhancing the substratum to water volume contact ratio. Therefore, these manmade systems potentially need less land area than natural wetlands for equal levels of treatment.". Other advantages of artificial wetlands over natural include the following: Purpose built wetlands allow a much greater flexibility in terms of their shape (Hammer & Kadlec 1983), size, and site selection (Gersburg et al. 1985).

On a simple assessment of the figures for percentage removal of nitrogen and phosphorus, artificial wetlands compare favourably with natural ones. Gale et al. (1993) found that removal rates for nitrogen were 49 mg N m^{-2} d^{-1} for the constructed wetlands studied, and 37 mg N m^{-2} d^{-1} for natural wetlands. These results represented removal by the soil only, removal in the field will be expected to be higher as plant uptake would also be occurring.

Artificial systems have been popular at all scales from small rural treatment works upwards for about the past decade (Hammer, 1989), (Cooper and Findlater, 1990), but often with poorer-than-anticipated removal of nutrients when used as a substitute for conventional sewage treatment (Brix, 1994), (Green and Upton, 1994). They appear to be most promising at present for small wastewater plants focusing on BOD removal, or as 'diffuse filters' in riparian zones rather than for specific removal of point-sourcer nutrients. Nevertheless, there is potential for management and flexibility in artificial systems and little work has been conducted on combinations (e.g. pond/reedbed): most authors advocate continued research into efficiency improvements of artificial systems combined with the use of artificial or natural systems wherever possible to produce small improvements in nutrient output (Chambers, Wrigley et al., 1993).

5. SITE-SPECIFIC METHODS FOR EUTROPHICATION CONTROL

5.1 Creation of inflow wetlands and/or shallow reservoirs by 'pre-dams'

Pre-dams or detention basins have been widely used in continental Europe for the protection of lake or reservoir resources. These structures are more or less hydraulically separated from the main water body and constitute an efficient trap for toxic substances and nutrients. When combined with dredging facilities, pre-dams also significantly increase the life span of reservoirs under threat of rapid siltation.

Sedimentation is a function of particle size and weight. Given ideal conditions of size, depth and retention time, pre-dams may collect between 60 and 80% of the load of adsorbed elements such as P, Cu, Fe, Pb and Zn. In areas under strong anthropogenic impact, where the export of phosphorus in dissolved form can account for a large proportion of the total phosphorus load, pre-dams act as biological filters through incorporation of phosphate by diatoms and successive sedimentation. An ortho-phosphorus retention model developed by (Benndorf and Putz, 1987a; Benndorf and Putz, 1987b) calculates ideal basin size and retention time based upon yearly estimates of phosphorus loading and limnological parameters. One large and six small pre-dam reservoirs and a submerged dam are able to reduce phosphorus loading to the Saidenbach reservoir (Germany) by about 77% (Uhlmann and Horn, 1992). Nutrient trap-efficiency, retention and immobilisation are dependent however on complex interrelated equilibria sometimes specific to the biogeochemistry of the catchment such as the Fe and Al:P ratio of sediments, the abundance of S, Si, NO_3 and $CaCO_3$ in the inflows. Igneous catchments tend to produce sediments with higher Fe:P ratios and consequently higher potential P adsorption (Uhlmann and Horn, 1992). This is due to their relative abundance in Fe and Al and to the fact that, being more resistant than catchments of other geological nature to chemical denudation, they export less phosphorus per unit runoff (Grobler and Silberbauer, 1985).

Small particles characterized by a large surface area available for adsorption may not sediment in the pre-dam and contribute significant phosphorus loading to the main water body. Conditions of low summer flows, high phosphorus loading and silica depletion favour the development of cyanobacteria which are buoyant and therefore are not retained by the pre-dam.. Similarly chrysomonads are motile and avoid sedimentation. The use of floating plastic sheets above submerged dams could prevent the passage of cyanobacteria and increase hydraulic separation between the pre-dam and the main water body. Their use is currently under investigation (Dietrich Uhlmann, pers. comm.).

In the summer high temperatures favour phosphorus release by the rapid degradation of organic matter. Oxygen depletion caused by this microbial activity may trigger anaerobic sediment phosphorus release. The presence of nitrate which behaves as electron acceptor in conditions of oxygen depletion, is crucial for the continued immobilization or redox-dependent phosphorus forms. Similarly anoxic conditions can occur in winter under ice (Klapper, 1991).

High storm runoff events may create turbulence and resuspension in the pre-dam. This can be partly alleviated by the construction of a series of small retention basins and regular sediment removal (Cooke et al., 1993). Pre-dams have been efficiently used in combination with phosphorus coagulants such as Fe and Al sulphate, Fe and Al chlorides or calcium carbonate.

5.2 Sewage diversion

Diversion of sewage effluent has been one of the earliest remedial methods for eutrophic lakes, since many of the early problems (e.g. Lake Washington, Lake Zurich) were caused by urban sewage effluent. Smaller water bodies, such as Bosherton Lakes (Pembrokeshire), Barton Broad (Norfolk) and Rutland Water Leiacestershire) have also been protected by diversion and it is under consideration for Slapton Ley (Devon). From results published in the literature in the past ten years, it seems that the time of recovery of a lake after sewage diversion is extremely variable and could be very long. Beside the amount of nutrient loading and the duration of the period of impact there are specific characteristics of water bodies that have been revealed as important for the determination of the time needed for recovery. Among these are the depth, the flushing rate, the thickness of the sediment and the length of time under cyanobacterial dominance – sometimes referred to as "biological memory" (Feuillade and Davies, 1994). The relative importance of these factors as well as the evaluation of the effectiveness of control measures are often difficult to determine since they presuppose detailed knowledge of complex limnological cycles. Simple nutrient loading models such as the ones of (Vollenweider, 1976) and (Lorenzen, 1976), used within their limits and supported by reasonable assumptions, can be employed in a first approximation to discuss observed changes (Barroin, 1994). Such an analysis carried out on Lake Nantua (Jura Mountains, SE France) revealed that the abatement of external phosphorus loading had an immediate impact reducing internal phosphorus loading while cyanobacteria persisted, moving towards nutrient rich layers in the metalimnion. Water clarity improved but the main target of the sewage diversion scheme – the reduction of phytoplankton biomass, was reached only 18 years after the implementation of the works. Studies of the ecology of bloom forming cyanobacteria revealed that these organisms were able to delay the phosphorus residence time in the lake to 16 years compared to a flushing rate of 0.7 years. There are several mechanisms that can account for self-maintaining cycles of phytoplankton growth and phosphate regeneration. The progressive decrease of available phosphorus reduced the buoyancy of cyanobacteria (a phosphorus -dependent mechanism) forcing them to sink to deeper layer during summer and eventually sediment to the bottom. Slow decomposition with formation of anaerobia and phosphorus release from bottom sediments contributed to a delay in the cleansing of the lake (Davies and Blanc, 1994). Certain species of cyanobacteria such as *Oscillatoria rubescens* under conditions of phosphate limitation find refuge at the limit of the metalimnion reinforcing water column stratification. They do so by three main mechanisms: they constitute a total light screen, they increase pH because of CO_2 consumption and they deplete nutrients preventing nutrient diffusion through the thermocline. In this way *Oscillatoria* actively contributes to the persistence of the nutrient status of Lake Nantua notwithstanding 18 years of sewage diversion (Feuillade, 1994).

A problem specific to sewage diversion is the reduction in water volume which can cause perturbations, bringing the ecosystem back towards a eutrophic state. In Alderfen Broad sewage diversion and isolation of the water body from the polluted river caused reduction of water levels and a fish kill occurred under conditions of particularly hot weather (Perrow et al., 1994). Bottom sediments became closer to the water surface, increasing heat absorption and causing a higher organic phosphorus degradation rate. Microbial activities were enhanced leading to sediment anoxia with iron reduction and P release. Higher pH as a consequence of photosynthesis under such circumstances may induce ligand exchange (OH^- for PO_4^-) on iron hydroxy complexes (Cooke et al., 1993). Reduced flushing contributes to concentrate the nutrients released into the water column.

The cost of sewage diversion can be relatively high as it may require building extensive pipelines, installation of stormwater retention basins and watertight installation of the interceptor to prevent contact with groundwater (Klapper 1991).

Systematic monitoring is necessary to describe the successive phases of lake recovery during abatement of nutrient loading. (Sas, 1989) critically evaluated the effect of sewage diversion from nine shallow and nine deep European lakes. He noted that while deep lakes responded rather rapidly to reduced loading, shallow lakes offered a delayed response independent from the proportion of nutrient reduced. This behaviour is attributed to the effect of phosphorus release from the sediment under various conditions (internal loading). The answer to the questions 'when' and 'to what extent' will lakes recover after reduction of external loading remains problematic.

Although in shallow lakes some correlation exists between internal phosphorus loading and phosphorus concentration in the water column after reduction of external loading (Van der Molen and Boers, 1994) no simple models relate changes in the external loading to expected consequences (Cooke et al., 1993). Some deep lakes, relatively independent of internal loading, such as Lake Washington (Edmonson and Lehman, 1981) behaved as predicted by Vollenweider's equation and recovered quickly after sewage diversion. This rather isolated case of successful recovery is due to specific characteristics of the lake; such as its great depth, the fast renewal rate, its oxic hypolimnion and the relatively short period of enrichment (Cooke et al., 1993).

In a more general sense one of the main assumptions of Vollenweider's model is the equilibrium between internal and external loading. By definition the disequilibrium following restoration precludes the predictive application of the loading model until the lake is in its "transient phase", a state hard to define. The best prediction of lake phosphorus concentration after the reduction of external loading can be achieved measuring independently parameters correlated to internal nutrient loading such as the Fe:P ratio in sediments (Van der Molen and

Boers, 1994). Multiple regressions are preferred to Vollenweider-type of relations for this type of model. Data on internal and external phosphorus loading, retention time, phosphorus concentrations in the water and in the sediments and iron concentrations in the sediments were reported by (Van der Molen and Boers, 1994) for 49 shallow lakes but judged insufficient for the definition of an operative model for the prediction of the recovery of nutrient concentrations.

5.3 Artificial mixing and aeration

5.3.1 Destratification

Artificial destratification may be achieved in three ways:

1. Introduction of compressed air through a perforated pipe or diffuser, so that the fine bubbles released entrap bottom water and circulate it to the surface.
2. The mechanical pumping of water from the hypolimnion to the epilimnion.
3. The 'jetting' of inflow water under pressure into a reservoir if it has pumped inflows.

The first method is the commonest. Commercial designs such as 'Helixor' tubes exist which usually need to be installed in a reservoir at the construction stage or in a lake during temporary drainage if this is possible, at the deepest point. These create a rising plume of bubbles. A simpler alternative is the sinking of a pipe of flexible material with regular small perforations along its length along the deepest axis of the lake connected to a compressor or other source of compresesed air. This creates a 'curtain' of bubbles. Both techniques destratify the immediate region of their operation and then progressively draw in water from further away until the body is fully mixed. The time which this takes depends upon the power of the equipment and the climatic conditions pertaining at the time - hot, calm conditions will act to re-establish stratification while cool windy conditions will aid destratification.

Ideally mixing should be used in combination during day and night cycles. Air-lifting using perforated pipes during the day may cause O_2 release from the supersaturated epilimnion of hypertrophic systems (Klapper 1991). Plastic foam can be used to achieve higher destratification efficiency by reducing air bubble sizes according to observations carried out by Knoppert in Holland (Burns, 1981). This is related to physical water circulation laws which state that the mass of water circulated is directly proportional to the cube root of the discharged air flow. Similarly the more the discharge of compressed air can be spread over the bottom area, the higher becomes the oxygenation efficiency (Verner, 1994).

Providing that the power of the equipment is adequate, artificial destratification is effective in maintaining a fully oxygenated water column with the supression of release of iron, manganese and nutrients. Pumping and inflow jetting (Steel, 1972) achieve the same ends - a fully mixed water column throughout the summer season when stratification would otherwise set in. The latter technique can usually only be operated continuously.

The effects of artificial mixing upon phytoplankton populations are apparently conflicting, however (Fosberg and Shapiro, 1980). If the main effect is to recycle nutrients which would otherwise be trapped in the hypolimnion back into the photic zone of the lake, then algal growth will be enhanced. On the other hand, if algal cells which would otherwise be confined to the narrow photic zone are circulated throughout the depth of the water column, and assuming that the mixed depth under these circumstances exceeds the euphotic depth, they experience a lower overall light regime and their photosynthesis and growth will be inhibited. Changing the light regime or the nutrient regime in a lake can also be expected to change the competitive advantages of one species over another, and thus alter the pattern of algal succession. In the range of studies reported both increases (e.g. Fast, 1981) and decreases (e.g. Cowell et al., 1987) of biomass and/or primary production have been found after mixing.

These apparently conflicting results are explicable on the basis of the capacity of the individual lake's destratification equipment to effectively mix the water column, coupled with the explanation of algal species succession developed by Reynolds and others, which were confirmed in a series of controlled experiments in limnetic enclosures (Reynolds et al., 1983, 1984). Reynolds and colleagues used periods of intermittent mixing which promoted the growth of spring-autumn diatom species; heavy cells adapted to low light conditions. Subsequent stratification eliminated these mixed-column species and resulted in the growth of summer 'r'-strategists such as small green algae. Alterate periods of mixing and stratification of about 3-weeks' duration prevented either type of algae from achieving their maximum potential biomasses. Slow growing, 'K' selected species slowly increased in abundance over the whole period of the project, but probably reached their peak at a lower biomass and later in the year than they would have done had more stable conditions existed. These experimental results support observations on the operation of intermittent mixing in reservoirs where population developments of individual species are not able to achieve their maximum (Ferguson & Harper, 1982) and where succession can be explained in terms of the Reynold's matrix of nutrients and column mixing. These explanations of changes in terms of the reproductive strategy of the taxonomic forms concerned should not be viewed in isolation from explanations which seek the more detailed reasons for shifts; indeed the two approaches go hand-in-hand.

Mixing may also alter zooplankton populations. There are conflicting results here too, with most studies reporting an increase in both individual size and total numbers of zooplankton following mixing (Pastorak et al., 1980) but some reporting the opposite (Cowell et al., 1987). The more usual observation, of increase, is explained by the distribution of zooplankton in a greater volume of water, reducing interactions with fish at lower light intensities, and hence reducing predation pressure upon the zooplankton. In shallower hyper-eutrophic lakes (Cowell et al., 1987) this effect would be lessened.

Artificial circulation is most suitable for stratified hypertrophic water bodies that are not nutrient limited. The hypolimnion is reduced or disappears altogether reducing redox-dependent phosphorus release from the sediments. Algal development is negatively affected as algae are transported below the limit of the photic zone and remain unproductive for part of the day. Often the regression of algal growth and the reduction of trophic conditions which occur under artificial destratification are mainly the consequence of the creation of an enhancement of zooplankton grazing as zooplankters find refuge in the oxygenated deep water escaping predation and become more effective in the control of algal development (Klapper 1991). Destratification is also suitable during the cold winters where low oxygen may cause fish kills under the ice. While winter destratification lowers the temperature of bottom sediments decreasing oxygen consumption and organic phosphorus degradation, in summer it would increase the temperature of the sediments causing opposite effects.

During recovery from eutrophic conditions by destratification, changes in species composition within the phytoplankton will invariably occur (Shapiro, 1981). Renewed abundance of silica in the epilimnion and increased changes in pressure as a consequence of reduced water column stability will favour diatoms at the expenses of bluegreens (Fast et al. 1973). In several water supply reservoirs such as the Biesbosch reservoirs in Germany, the London and the reservoirs in the Anglian Region specific injector nozzles are employed to increase pressure shifts and disrupt the development of bluegreen blooms. Negative impacts caused by sediment stirring during the laying and the operation of destratification pipes for air-lifting can be avoided by laying the pipes just above the sediment and control their position by integrated buoyancy pipes (Verner 1994).

5.3.2 Hypolimnetic aeration

Aeration without disruption of stratification can be used in deep lakes for a similar purpose as destratification with the added advantages that it does not increase the temperature of the hypolimnion and also prevents the advection of nutrient rich water into the epilimnion. Numerous sophisticated devices to achieve hypolimnion oxygenation exist and some have been reviewed by Klapper (1991). Operation is usually started after the spring circulation period to counterbalance the increasing oxygen demand in the hypolimnion and the aerators are left to run throughout the summer (Verner 1994). Oxygen injection is preferred to air to avoid the build up of nitrogen supersaturation which is toxic to fish (Fast, 1981). The impact of the treatment on hypolimnetic anoxia depends on oxygen transfer efficiency (i.e. the ratio between the oxygen supplied with the air flow delivered by the compressor and the oxygen in the water) which is porportional to contact oxygen-water time. Calculations and trials have to be carried out to optimize the oxygenation capacity of the intallation in relation to the number of aerators and flow velocity (Verner 1994). The efficiency of the equipment is to be compared to the lake oxygen consumption rate which can vary over two orders of magnitude among different water bodies. The choice of different techniques is therefore determined by specific conditions.

The comparative effects of destratification and hypolimnion aeration are presented in the following table:

Comparative effects - destratification and hypolimnion aeration (modified after Judell 1981)

Water quality effect	Destratification	Hypolimnion aeration
O_2 in hypolimnion	O_2 restored	O_2 restored
Corrosivity	Reduced by CO_2 stripping	CO_2 may accumulate
Temperature of hypolimnion	Increased	No change
Temperature of epilimnion	Reduced	No significant change
Evaporation	Reduced	No change
Plankton blooms	Reduced	Research warranted
Distribution of algae	throughout the reservoir	Algae maintained in epil.
Eutrophication	significant retardation	Marginal retardation

Several devices have been designed for the two techniques. Capital costs of equipment required for hypolimnion aeration are significantly higher than air diffuser systems employed in destratification. Power requirement (running costs) may not differ significantly (Judell, 1981). Water bodies should be considered individually to decide which technique is more suitable. Because of higher capital costs, hypolimnetic aeration is preferred with water bodies having low oxygen depletion rates and/or small size (Fast 1981). Controversy over the efficiency of either method are sometimes confused by inadequate use or poor performance of the equipment. Under certain conditions inadequate circulation can cause an increase in algal biomass and a shift towards a higher proportion of bluegreens (Shapiro 1981).

Several factors contribute to oxygenation efficiency (modified from Verner 1994):

- the solubility of oxygen increases considerably in cold water. Hypolimnetic aerators benefit from low water temperature;
- the oxygen transfer efficiency depends on the oxygen concentration of the intake water. The more the intake water is depleted of oxygen, the better the transfer efficiency;

- reduced chemical compounds can be instantaneously oxidized in the aerator unit. This must be taken into account during calcuations;
- the air pressure needed and the compressor design will determine specific compressor efficiency and consequently oxygen transfer rates.

Failure to obtain oxic conditions in the hypolimnion after deep oxygenation was recently reported for Lake Nantua in South-East France (Barroin 1994). The author stressed the conceptual subtlties involved in the calculation of the oxygn depletion rate and its relation to required oxygen supply to the hypolimnion. Only estimates of the expected oxygen depletion rate under operative conditions can be made as a consequence of the following destabilizing effects brought by the treatment itself:

1) the increase in dissolved oxygen will lead to a rise in repiratory activity of the indigenous fauna (micro and macro) and the immigration of new aerobic species

2) the increased stirring of the hypolimnion could:
(a) speed up water/sediment exchange and thus the rate of oxygen consumption,
(b) erode the metalimnion increasing the volume of the hypolimnion,
(c) cause a temperature rise increasing metabolic and respiratory rates.

Estimated correction factors were cited by several authors and remain still today probably the best practicable solution. Cooke et al. (1993) recommend 100% and indicate that (Steinberg and Arzet, 1984) propose a factor of 4 for each 10° C when there is a large increase in temperature. The final impact of hypolimnion aeration on P stripping from the water is linked to chemical characteristics of the sediment and the dynamics of sediment accumulation by external loading (Gächter, 1987).

5.3.4 Epilimnetic mixing

This technique aims to decrease phytoplankton growth by increasing the epilimnion. During mixing events within the deepened epilimnion, algal cells travel below the compensation point in such a way that their respiration outweighs photosynthetis. The ideal mixing depth is the depth at which the integral net phytoplankton production is minimized (Straskraba 1994). This depth can be calculated following the model proposed by Steel from his studies on London reservoirs (Steel, 1978). In these reservoirs mixing is used for complete destratification. Partial destratification is discussed by (Krambeck, 1988). Following a similar principal, partial reduction in algal biomass can be achieved by increasing the background attenuation coefficient to manipulate the light regime to the detriment of phytoplankton (Ridley and Steel, 1975).

5.4 Immobilisation or removal of nutrients in lake sediments

Sediment removal

The earliest successful removal of the top sediment as a measure of decreasing internal loading was carried out in Lake Trummen, Sweden. An important pre-requisite for the success was the presence of ancient sediments with low P content beneath the sediments abstracted. Pre-project investigations comprised sediment stratigraphy, chemistry, spatial distribution and a test confirming the suitability of the sediment for agricultural purposes (Björk 1985, 1994). The top 0.5 m sediment was removed by a suction dredger in 1970-71 and pumped into a settling pond. The runoff from the pond was treated with aluminium sulphate for the precipitation of phosphorus and suspended matter. Such treatment achieved 30 µg/l TP in the runoff water returned to the lake. An immediate improvement occurred in lake phosphorus concentrations and algal biomass (Andersson et al., 1973), (Bengtsson and Gelin, 1975).

The cost of sediment removal often prevents the use of such management techniques to large lakes; the smaller Norfolk Broads are a case in point (Harper, 1992). Beside capital costs for the dredging machinery it is necessary to carry out in large lakes a detailed investigation of sediment composition and the phosphorus adsorbing capacity of different sediment layers to ensure efficiency in the reduction of internal loading. Dredging activity does cause some sediment resuspension but its effects are contained. In the design of suction dredgers higher priority shoud be given to the precision of removing targetted sediment layers than to pumping capacity (Björk, 1994). The general problem of pumping excessive water faced by conventional dredgers which oblige the construction of extensive settling ponds, was recently solved by the design of a new highly automated sledge suction dredger able to remove sediment with a much higher content of dry matter. This dredger was designed in Sweden (Industrikonstruktioner AB), built under cooperation by Czech and Swedish firms and operated for the first time in Vajgar fish pond in Czech Republic . Accurate positioning of the suction nozzles is obtained by laser technology (Pokorny' and Hauser, 1994) which requires cleaqring of the lake bottom from living roots by specific techniques and machinery. Some of these are cited by Björk (1994).

Sediment conditioning

The principle behind this technique aims at increasing chemical gradients at the sediment-water interface. Nutrient poor sediments can be abstracted from ancient deposits in the lake and then returned to cover the recent sediment. An improved method of extraction by jet flushing, resuspension and dressing was patented by Klapper and co-workers in 1988

(Klapper 1991). Depending on pH conditions in the lake and abundance of calcite, a simultaneous precipitation of phosphorus and algae can be achieved (Klapper 1991).

Chemical conditioning is carried out primarily with the intent of increasing sediment phosphorus-binding capacity. The techniques are based on the observation that in natural oligotrophic lakes with low phosphorus loading, the phosphorus from the catchment is readily bound to iron or lime compounds and deposits by precipitation. These inorganic self-purification mechanisms are overcome in eutrophic water bodies but can be restored by the addition of phosphorus coagulants.

Examples of addition of iron and aluminium in the form of chloride and sulphate salts are numerous. Additions of iron oxide-hydroxide are also common (Wolter, 1994). The release of protons which takes place after addition of iron and aluminium salts can be buffered by slaked lime ($Ca(OH)_2$), chalk or sodium carbonate (Klapper 1991). Careful pH control is necessary to prevent the formation of forms of aluminium which are inefficient in phosphorus sorption and even toxic; the best pH is in the range 6-8. Theoretical maximum Al additions in relation to lake alkalinity can be computed with the nomogram published by (Kennedy and Cooke, 1982). In reality only empirical trials can give an idea of the best dosage as addition of trivalent Al interacts with complex chemical equilibria. Toxic effects should not be significant below a concentration of 50µg Al/l . Some evidence however testifies long-term bioaccumulation in macrophytes and aluminium toxicity on fish and invertebrates even at low doses (Cooke et al. 1993). Heavy dosages of alum (aluminium sulphate with Al^{3+}>5g m^3) can give formation to a floc active in trapping TP and sedimenting it to the bottom (Wolter 1994).

At pH 4 and above aluminium complexes with silicon to form phosphorus-sorptive hydroxy-alumino-silicate. High levels of silicon are thought by some authors to be as effective as the addition of buffers to prevent aluminium toxicity. The practice of aluminium additions has yielded contrasting results, pointing to the complexity of Al chemical equilibria in freshwaters (Cooke et al. 1993). Guidelines for practical sediment conditioning by the use of aluminium based on a review of several case-studies are given by Cooke et al. (1993).

Iron salts (ferric chloride, ferric sulphate) do not give rise to ions which may be directly toxic (notwithstanding the physical effects on benthic communities demonstrated at Rutland Water) but maximum efficiency of phosphorus entrapment by covalent bonding or sorption, is realized between pH 5 and 7 when the $Fe(OH)_3$ form predominates (Cooke et al. 1993). Buffers may be employed to control the acidity released into the lake by additions of ferric chloride or ferric sulphate. pH shifts during such operations are not thought to cause toxic effects on fish and invertebrates provided the pH remains above 6.

In contrast to aluminium compounds, iron bonds are redox sensitive. During the progressive covering of sediments by successive layers, iron phosphate ($FePO_4$) may occur in progressively lower redox conditions up to a point when it is reduced to Fe^{2+} and migrates along its concentration gradient to the sediment surface. Here, given aerobic conditions in the overlaying water, it is newly oxidized to FeO(OH) and ready to trap more phosphorus. Through this mechanism iron additions are seen as dynamic phosphorus traps compared to aluminium which, once covered by successive sediment layers, remains inert and loses its role of stripping phosphorus from the water column (Wolter 1994).

The addition of calcite at pH>9 leads to precipitation of $CaCO_3$ which readily binds soluble inorganic phosphorus. As for iron addition, the use of calcium as a coagulant is possible only under constant redox and sometimes pH control employing auxiliary forms of management such as artificial circulation, aeration and addition of buffers.

A general problem faced by these techniques of phosphorus inactivation is the effect of aquatic macrophytes which take up phosphorus from below the sediment surface and release through subsequent senescence into the epilimnion. Macrophyte growth is enhanced by light limitation as a consequence of phosphorus limitation of algal growth (Harper 1992). Foxcote reservoir in south-east England experienced extensive growth of rooted macrophytes and filamentous algae after three years of successful ferric phosphate dosing (Young et al., 1988).

Some authors in Europe have proposed the use of fly ash from the combustion of coal or oil as it has a high content of calcium and magnesium oxides and trivalent Al. Experimental applications had toxic effect because of the release of heavy metals and sharp changes in pH. The treatment is not recommended (Wolter, 1994).

The addition of phosphorus coagulants presupposes the maintenance of aerobic conditions to prevent release of redox-dependent phosphorus. Artificial circulation and aeration techniques need to be used in conjunction with ferric dosing. An alternative to permanent artificial aeration is the one-off application of chemical electron acceptors. Addition of nitrate or fully nitrified sewage as an electron acceptor is currently widely applied in Germany and Sweden after the first large scale experiment carried out by Ripl on Lillesjön (Sweden, (Ripl, 1978)). After a combined treatment with nitrate ($Ca(NO_3)_2$) and ferric chloride lake metabolism changed drastically and has not reverted to eutrophic conditions since. The iron treatment was applied first to destabilize hydrogen sulphide and other sulphide compounds in the sediment. This made the nitrate addition more efficient. Care was given to pH control since acid conditions caused by iron salts additions may retard denitrification activity (Ripl, 1994).

No general recommendations can be made about dosages of chemicals for sediment conditioning as these will depend on sediment composition, oxygen consumption rate, reduced sulphur compounds content, Fe:P ratio, pH, and the specific penetration of the chemicals into the sediments. In Lake Lillesjön iron and nitrate dosages were determined by successive experiments. Specific costs are high and depend primarily on the cost of the distribution of the chemicals. Recent experiments in the Schlei estuary (Schleswig, northern Germany) with addition of nitrate into tributaries or dephosphorised and nitrified effluents from treatment plants gave promising results with lesser expenditure (Ripl 1994).

Techniques of sediment conditioning are currently investigated to optimize efficiency and reduce application costs. One of the directions of research speculates the impact of coagulant addition at strategic times of the year in connection with successional phases of phytoplankton development and in combination with other forms of lake management (Jürgen Benndorf, pers. comm.). In general only pluriannual trials can give sure indication of the best time for treatment and correct dosage. As a rule the best time for treatment is when the phosphate fraction is at its relative maximum which is usually with the period between late autumn and spring in temperate regions of the northern hemisphere (Wolter 1944).

6. RECIPIENT ECOSYSTEM ECOTECHNICAL MANAGEMENT

This section includes the following topics, which are concurrently under more detailed examnation in the context of the reservoirs of the Northern Area under a separate project. The report for this latter project is due two weeks after the present draft report and so the conclusions from that will be incorporated into the final report of this.

- Management of fish stock through planktivore removal, predator enhancement and/or spawning control (and see (Parr, 1992), (Phillips and Moss, 1993), (Reynolds, 1994).
- Shoreline and littoral management to enhance nutrient metabolism in both lakes and estuaries
- Management of littoral zone for zooplankton refuges
- Introduction of algal predator/parasites
- Mechanical disruption of cyanobacterial physiology

- Algal inhibition by use of barley straw or similar toxin producers

7 PRINCIPLES OF USE OF REMEDIAL MEASURES IN WATER BODIES AND WATER COURSES

The effectiveness of control strategies for eutrophication will eventually hang upon the extent to which each possible method can be applied to the catchment or waterbody under stress. In order to do this two structural pre-requisites are necessary. The first is an adequate data-base for water bodies and the second is an effective decision-tree so that managers without previous extensive experience can implement it as effectively as those with.

6.1 Information base

The development of infomation bases about water body management has been undertaken in different ways in three different countries. In Denmark (Jeppesen pers. comm.) a national database is being established about the restoration chronology of every lake. In the US, and proposed as an international approach by UNESCO (Ryding and Rast, 1989) is the database built up from the OECD classification system (Vollenweider and Kerekes, 1982). Most recently, a system has been proposed for the UK as part of the Lakes Classification Project and this is being compared with the other national approaches.

6.2 Scientific basis for decision structures

Closely linked obviously with the functions of a lake database is the need for clear decision pathways, both using mathematical models to summarised large data sets and indicate trends and relationships, and using decision trees to ensure that the best decsions are taken in correct sequence. There are a number of practical recommendations which are being evaluated, starting from (Ryding and Rast, 1989), (Sas, 1989), (Straskraba et al., 1993) and (Straskraba, 1994).

7. REFERENCES

Andersson, G., G. Cronberg, et al. (1973). Planktonic changes following the restoration of Lake Trummen, Sweden, *Ambio,* **2**: 44-48.

Anon (1991). Eutrophication of Coastal Waters. Funen County Council.

Anon (1992). Sixteenth Report Freshwater Quality. Royal Commission on Environmental Pollution.

Anon (1994). Lake restoration: the role of biomanipulation. National Rivers Authority and the Broads Authority.

Anon. (1990). *Toxic Blue-Green Algae.* Peterborough, National Rivers Authority.

Anon. (1992). Blue-Green Algae: Final Report of the Blue-Green Algae Task Force. Department of Water Resources, New South Wales.

Anon. (1994). Comprehensive Studies for the Purposes of Article 6 of Dir 91/271 EEC, The Urban Waste Water Treatment Directive. Comprehensive Studies Task Team, Forth River Purification Board.

Barroin, G. (1994). Studies on Lake Nantua. The sewage diversion and hypolimnetic oxygenation projects: techniques and effectiveness, *Archiv für Hydrobiologie Beihefte, Ergebnisse der Limnologie,* **41**(Studies on Lake Nantua (France). A eutrophic lake on the way to rehabilitation): 17-31.

Bengtsson, L. and C. Gelin (1975). Artificial aeration and suction dredging methods for controlling water quality, ed, *Effects of storage on Water Quality,* Medmenham, Water Research Centre. 314-342.

Benndorf, J. and K. Putz (1987a). Control of eutrophication of lakes and reservoirs by means of pre-dams - I Mode of operation and calculation of the nutrient elimination capacity, *Water Research,* **21**: 829-838.

Benndorf, J. and K. Putz (1987b). Control of eutrophication of lakes and reservoirs by means of pre-dams - II Validation of the phosphate removal model and size optimization, *Water Research,* **21**: 839-842.

Björk, S. (1994). Lake Trummen, Sweden, ed, *Restoration of Lake Ecosystems - A Holistic Approach.,* Slimbridge, Gloucestershire, International Waterfowl Research Bureau; I.W.R.B. Publication No 32. 130-140.

Boers, P. C. M. and D. T. Van der Molen (1993). Control of eutrophication in lakes: the state of the art in Europe, *European Water Pollution Control,* **3**: 19-25.

Boyt, F. L. (1977). Removal of nutrients from treated municipal wastewater by wetland vegetation, *Journal of the Water Pollution Control Federation*, **48**: 789.

Brix, H. (1994). Constructed wetlands for municipal wastewater treatment in Europe, ed, *Global Wetlands: Old World and New*, Amsterdam, The Netherlands, Elsevier, Bv. 325-333.

Burns, F. L. (1981). *Comment*, Destratification of lakes and reservoirs to improve water quality, Melbourne, Australia, Department of National Development and Energy, Australian Water Resources Council, Australian Government Publishing Service.

Burt, T. P. and N. E. Haycock (1993). Controlling losses of nitrate by changing land use, ed, *Nitrate – Processes, patterns and Management*, Chichester, John Wiley & Sons Ltd. 341-367.

Chambers, B. J., D. B. Davies, et al. (1992). Monitoring of water erosion on arable farms in England and Wales 1989-1990, *Soil Use and Management*, **8**(4): 163-170.

Chambers, J. M., T. J. Wrigley, et al. (1993). The potential use of wetlands to reduce phosphorus export from agricultural catchments, *Fertiliser Research*, **36**: 157-164.

Christensen, P. B., L. P. Neilsen, et al. (1990). Denitrification in nitrate-rich streams: diurnal and sesonal variation related to benthic oxygen metabolism, *Limnology and Oceanography*, **35**: 640-651.

Christensen, P. B. and J. Sorensen (1988). Denitrification in sediment of lowland streams, *FEMS Microbial Ecology*, **53**: 335-344.

Collingwood, R. W. (1977). A survey of eutrophication in Britain and its effects upon water supplies. Water Research Centre, Medmenham.

Converti, A., M. Rovatti, et al. (1995). Biological removal of phosphorus from wastewaters by alternating aerobic and anaerobic conditions, *Water Research*, **29**: 263-269.

Cooke, G. D., E. B. Welch, et al. (1993). *Restoration and management of lakes and reservoirs*. Boca Raton, FA., Lewis.

Cooper, A. B. and J. Cooke . G. (1984). Nitrate loss and transformation in two vegetated headwater streams, *New Zealand Journal of Marine and Freshwater Research*, **18**: 441-450.

Cooper, P., J. Upton, et al. (1995). Biological nutrient removal: design snags, operational problems and costs, *Journal of the Institution of Water and Environmental Management*, **9**: 7-18.

Cooper, P. C. and B. C. Findlater, Ed. (1990). *Constructed Wetlands in Water Pollution Control*, Oxford, Pergamon Press.

Davies, A. and P. Blanc (1994). Studies on Lake Nantua. Changes in nutrient availability following a large scale sewage diversion: A chemical perspective of blooms of *Oscillatoria rubescens*, *Archiv für Hydrobiologie Beihefte, Ergebnisse der Limnologie,* **41**(Studies on Lake Nantua (France). A eutrophic lake on the way to rehabilitation): 33-53.

Descy, J.-P. (1992). Eutrophication in the river Meuse, ed, *Eutrophication: Research and Application to Water Supply,* Ambleside, Freshwater Biological Association. 132-142.

DOE (1986). Nitrate in Water. Department of the Environment, HMSO, London.

Edmonson, W. T. and J. T. Lehman (1981). The effects of changes in the nutrient income on the condition of Lake Washington, *Limnology and Oceanography,* **26**: 1-29.

Elliot, M., G. K. L. Burrell, et al. (1994). Trophic Status of the Ore/Alde, Deben Stour and Colne Estuaries. University of Hull Reports in Applied Marine Biology.

Fast, A. W. (1981). *Comment,* Destratification of lakes and reservoirs to improve water quality, Melbourne, Australia, Department of National Development and Energy, Australian Water Resources Council, Australian Government Publishing Service.

Feuillade, J. (1994). Studies on Lake Nantua. Foreword by the guest editor, *Archiv für Hydrobiologie Beihefte, Ergebnisse der Limnologie,* **41**(Studies on Lake Nantua (France). A eutrophic lake on the way to rehabilitation): V-IX.

Feuillade, M. and A. Davies (1994). Studies on Lake Nantua. Seasonal variations of long-term trends in phytoplankton pigments, *Archiv für Hydrobiologie Beihefte, Ergebnisse der Limnologie,* **41**(Studies on Lake Nantua (France). A eutrophic lake on the way to rehabilitation): 95-111.

Fox, I. and M. A. Malati (1985). Cycling of Phosphorus in Rivers. Department of the Environment.

Foy, R. H., R. V. Smith, et al. (1995). Upward trend in soluble phosphorus loadings to Lough Neagh despite phosphorus reduction at sewage treatment works, *Water Research,* **29**: 1051-1063.

Gächter, R. (1987). Lake restoration. Why oxygenation and artificial mixing can not substitute for a decrease in the external phosphorus loading, *Schweizerische Zeitung Hydrologie,* **49**: 170-185.

Golterman, H. L. and N. T. de Oude (1991). Eutrophication of rivers, lakes and coastal seas, ed, *The Handbook of Environmental Chemistry,* Berlin, Springer-Verlag. 79-124.

Green, M. B. and J. Upton (1994). Constructed reed beds: a cost-effective way to polish wastewater effluents for amall communities, *Water Environment Research,* **66**: 188-192.

Grobler, D. C. and M. J. Silberbauer (1985). The combined effects of geology, phosphate sources and runoff on phosphate export from river basins, *Water Research*, **19**((8)): 975-981.

Groffman, P. M., A. J. Gold, et al. (1992). Nitrate dynamics in riparian forests: Microbial studies, *Journal of Environmental Quality*, **21**(4): 666-671.

Hammer, D. A., Ed. (1989). *Constructed Wetlands for Wastewater Treatment*, Proceedings of the First International Conference on Constructed Wetlands for Wastewater Treatment. Chelsea, Mighigan, Lewis Publishers Inc.

Harper, D. M. (1992). *Eutrophication of Freshwaters: Principles, Problems and Restoration*. London, Chapman and Hall.

Haycock, N. E. (1994). *Watershed management for the conservation of water quality in England*, Proceedings of the International Symposium on Agricultural Water Quality Management Techniques, Suwon, Korea,

Haycock, N. E. and T. P. Burt (1993). Role of floodplain sediments in reducing the nitrate concentration of subsurface run-off: a case study in the Cotswolds, UK, *Hydrological Processes*, **7**: 287-295.

Haycock, N. E., G. Pinay, et al. (1993). Nitrogen retention in river corridors: European perspective, *Ambio*, **22**: 340-346.

Heathwaite, A. L., T. P. Burt, et al. (1993). Overview - the nitrate issue, ed, *Nitrate – Processes, patterns and Management*, Chichester, John Wiley & Sons Ltd. 3-21.

Henderson-Sellars, B. and H. R. Markland (1987). *Decaying Lakes*. Chichester, John Wiley & Sons.

Hindmarsh, M. (1992). The Biomanipulation of Blue-Green Algae. Churchill Fellowship Study Report.

Holmes, N. and C. Newbold (1984). River Plant Communities - Reflectors of Water and Substrate Chemistry. Nature Conservancy Council (now English Nature).

Irving, R. (1993). Too Much of a Good Thing: Nutrient enrichment in the UK's INland and Coastal Waters. World Wildlife Fund, Godalming, Surrey.

Johnes, P. and T. P. Burt (1991). Water quality in the Windrush catchment: nitrogen speciation and sediment interactions, ed, *Sediment and Stream Water in a Changing Environment: Trends and Explanations*, Wallingford, IAHS Publication. 349-357.

Johnes, P., B. Moss, et al. (1995). The determination of water quality from land use, stock headage and population data – testing of a model for use in conservation and water quality management, *Freshwater Biology*, **in press**:

Johnes, P. J. (1995). Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters; I the export coefficient modelling approach, *Water Resources Research,* **in press**:

Johnson, D., R. L. Norton, et al. (1992). Eutrophication Risk Assessment. Scotland and Northern Ireland Forum for Environmental Research.

Johnston, E. (1994). Eutrophication Problems and Remedial Measures. TNO Institute of Environmental Sciences, Delft, The Netherlands.

JONUS (1994). Establishing the Nutrient Status of UK Coastal Waters. Ministry of Agriculture, Fisheries and Food.

Jordan, T. E., D. L. Correll, et al. (1993). Nutrient interception by a riparian forest recieving inputs from adjacent cropland, *Journal of Environmental Quality,* 22(3): 467-473.

Judell, J. T. (1981). *Destratification versus hypolimnion aeration,* Destratification of lakes and reservoirs to improve water quality, 1979 - Melbourne, Australia, Department of National Development and Energy, Australian Water Resources Council, Australian Government Publishing Service, Canberra.

Karr, J. and I. J. Schlosser (1978). Water resources and the land-water interface, *Science,* **201**: 229-234.

Kennedy, R. H. and G. D. Cooke (1982). Control of lake phosphorus with aluminium sulphate. Dose determination and application techniques, *Water Research Bulletin,* **18**: 389-395.

Klapper, H. (1991). *Control of eutrophication in inland waters.* New York, Ellis Horwood.

Krambeck, H. J. (1988). Partial destratification of eutrophic lakes, *Verhandlungen der Internationale Vereinigung Theoretische und Angewandte Limnologie,* **23**: 801-806.

Krug, A. (1993). Drainage history and land use pattern of a Swedish river system – their importance for understanding nitrogen and phosphorus load., *Hydrobiologia,* **251**: 285-296.

Lorenzen (1976). cited by Barroin (1994), :

MAFF (1991). Code of good agricultural practice for the protection of water. Ministry of Agriculture, Fisheries and Food, London.

MAFF (1993). Code of good agricultural practice for the protection of soil. Ministry of Agriculture, Fisheries and Food, London.

Mainstone, C., J. Gulson, et al. (1994). Phosphates in Freshwater: Standards for Conservation. English Nature.

Parr, W. and S. Clarke (1992). A Review of Potential methods for Controlling Phytoplankton, with Particular Reference to Cyanobacteria, and Sampling Guidelines for the Water Industry. Foundation for Water Research, Marlow, Bucks.

Parr, W., S. C. Nixon, et al. (1993). Classification Options for Trophic Status. National Rivers Authority.

Pedersen, S. E. and V. H. Petersen (1992). Monitoring nitrogen and phosphorus runoff in the county of Funen (DK), ed, *River Water Quality: Ecological Assessment and Control,* Brussels, Commission of the European Communities. 739-780.

Perrow, M. R., B. Moss, et al. (1994). Trophic interactions in a shallow lake following a reduction in nutrient loading: a long-term study, *Hydrobiologia,* **275**: 43-52.

Peterjohn, W. T. and D. L. Correll (1984). Nutrient dynamics in an agricuiltural watershed: observations on the role of a riparian forest, *Ecology,* **65**: 1466-1475.

Petersen, R. C., L. B.-M. Petersen, et al. (1992). A building block model for stream restoration, ed, *River Conservation and Management,* Chichester, John Wiley & Sons. 293-309.

Phillips, G. and B. Moss (1993). Is Biomanipulation a Useful Technique in Lake Management. National Rivers Authority.

Pinay, G., N. E. Haycock, et al. (1994). The role of denitrification in nitrogen removal in river corridors, ed, *Global Wetlands – Old World and New,* The Netherlands, Elsevier, B.V. 107-116.

Pokorny', J. and V. Hauser (1994). Vajgar fish pond, Czech Republic, ed, *Restoration of Lake Ecosystems - A Holistic Approach.,* Slimbridge, Gloucestershire, International Waterfowl Research Bureau; I.W.R.B. Publication No 32. 141-153.

Reynolds, C. S. (1994). The ecological basis for the successful biomanipulation of aquatic communities, *Archiv Fur Hydrobiologie,* **130**: 1-33.

Reynolds, P. E., J. C. Scrivener, et al. (1993). Review and synthesis of Carnation Creek herbicide research, *Forestry Chronicle,* **69**(3): 323-330.

Ridley, J. E. and A. Steel (1975). Ecological aspects of river impoundments, ed, *River Ecology,* Oxford, Blackwell. 565-587.

Ripl, W. (1978). *Oxidation of lake sediments with nitrate - a restoration method for former recipients*. Lund, Sweden, Institute of Limnology, University of Lund.

Ripl, W. (1994). Sediment treatment, ed, *Restoration of Lake Ecosystems - A Holistic Approach.,* Slimbridge, Gloucestershire, International Waterfowl Research Bureau; I.W.R.B. Publication No 32. 75-82.

Marsden, M. W., M. R. Smith, et al. (1995). Review of the trophic status of rivers in the Forth catchment, *Journal of the Institute of Water Engineers and Scientists,* **in press**:

McColl, R. H. (1978). Chemical runoff from pasture: the influence of fertiliser and riparian zones, *New Zealand Journal of Marine and Freshwater Research,* **12**: 371-380.

McGarrigle, M. L. (1993). Aspects of river europhication in Ireland, *Annales de Limnologie,* **29**: 355-364.

Moller Andersen, M. J. (1986). Reversal of eutrophication in the Baltic Sea, *Vattern,* **42**:

Moore, L. and K. Thornton (1988). Lake and Reservoir Restoration Guidance Manual. US EPA.

Morse, G. K. (1993). The Economic and Environmental Impact of Phosphorus Removal from Wastewater in the European Community. Centre Européan d'Etudes des Pholyphosphates E.V.

Morse, G. K., J. N. Lester, et al. (1994). The Environmental and Economic Impact of Key Detergent Builder Systems in the European Union. Centre Europeén D'Etudes des Polyphosphates E.V.

Moss, B., P. Johnes, et al. (unpublished). The monitoring and classification of standing waters in temperate regions by state-changed schemes – a discussion and a proposal based upon a worked scheme for British waters, *in press,* :

Moss, B., S. McGowan, et al. (1993). Current Limnological Conditions of a Group of the West Midlands Meres that bear SSSI Status.

Muscutt, A. D., G. L. Harris, et al. (1993). Buffer zones to improve water quality: a review of their potential use in agriculture, *Agricultural Ecosystems and Environment,* **45**(1-2): 59-77.

Nieswand, G. H., R. M. Hordan, et al. (1990). Buffer strips to protect water supply reservoirs: a model and recommendations, *Water Resources Bulletin,* **26**(6): 959-966.

O'Riordan, T. and G. Bentham (1993). The politics of nitrate in the UK, ed, *Nitrate: Processes, Patterns and Management,* Chichester, John Wiley & Sons. 403-416.

Osborne, L. L. and D. A. Kovacic (1993). Riparian vegetated buffer strips in water-quality restoration and stream management, *Freshwater Biology,* **29**: 243-258.

Parkinson, R. J. (1993). Changes in agricultural practice, ed, *Nitrate – Processes, patterns and Management,* Chichester, John Wiley & Sons Ltd. 322-339.

Parr, W. (1992). Manipulation of Fish Communities to Control Algal Levels in Reservoirs. Foundation for Water Research, Marlow, Bucks.

Ryding, S. O. and W. Rast (1989). *The Control of Eutrophication in Lakes and Reservoirs.* Paris, UNESCO and Parthenon.

Sas, H. (1989). *Lake Restoration by Reduction of Nutrient Loading: Expectations, Experiences, Extrapolations.* Richarz, Academia Verlag.

Shapiro, J. (1981). *Effects of artificial circulation in reservoirs on algae,* Destratification of lakes and reservoirs to improve water quality, Melbourne, Australia, Department of National Development and Energy, Australian Water Resources Council, Australian Government Publishing Service.

Sharpley, A., J. J. Meisinger, et al. (1995). Impacts of animal manure management on ground and surface water quality, ed, *Effective Management of Animal Waste as a Soil Resource,*

Sharpley, A. N., S. C. Chapra, et al. (1994). Managing agricultural phosphorus for protection of surface waters: issues and options, *Journal of Environmental Quality,* **23**: 437-451.

Sharpley, A. N. and S. J. Smith (1990). Phosphorus transport in agricultural runoff: the role of soils erosion, ed, *Soil Erosion on Agricultural Land,* Chichester, John Wiley & Sons. 351-366.

Smith (1984). Phosphorus versus nitrogen limitation i the marine environment.*Limnology and Oceanography,* **29**: 1149.

Stansfield, J. and B. Moss (1992). River Eutrophication and The North Sea Catchment. Friends of the Earth, London.

Steel, J. A. (1978). Reservoir algal productivity, ed, *Mathematical models in water pollution control,* New York, Wiley. 107-135.

Steinberg, C. and K. Arzet (1984). Impact of hypolimnetic aeration on abiotic and biotic conditions in a small kettle lake, *Environmental Technology Letters,* **5**: 151-162.

Straskraba, M. (1994). Ecotechnologial models for reservoir water quality management, *Ecological Modelling,* **74**: 1-38.

Straskraba, M., J. G. Tundisi, et al. (1993). State of the art of reservoir limnology and water quality management, ed, *Comparative Reservoir Limnology and Water Qualtiy Managment,* Dortrecht, Kluwer. 213-288.

STWA (1988). The Hatton Catchment Study. Severn-Trent Water Authority, Birmingham.

Sweeting, R. A. (1994). River pollution, ed, *The Rivers Handbook Volume II,* Oxford, Blackwells Scientific Publications. 23-32.

Tracy, K. D. and A. Flammino (1987). *Biochemistry and energetics of biological phosphorus removal*, Biological Phosphate Removal from Wastewaters, Rome, Italy, Pergammon Press, Oxford.

Turner, T. (1993). The Importance of Diffuse Sources of Phosphorus from Scarrow Beck to the River Bure, Norfolk. Napier University of Edinburgh.

Uhlmann, D. and H. Horn (1992). The significance of sedimentation and sediments to phytoplankton growth in drinking-water reservoirs, ed, *Eutrophication: Research and application to water supply*, Ambleside, Freshwater Biological Association. 94-106.

Uusi-Kamppa, J. and T. Ylaranta (1992). Reduction of sediment phosphorous and nitrogen transport on vegetated buffer strips, *Agricultural Science in Finland*, **6**(1): 569-575.

Van der Molen, D. T. and P. C. M. Boers (1994). Influence of internal loading on phosphorus concentration in shallow lakes before and after reduction of the external loading, *Hydrobiologia*, **275-276**: 379-389.

Verner, B. (1994). Aeration, ed, *Restoration of Lake Ecosystems - A Holistic Approach.*, Slimbridge, Gloucestershire, International Waterfowl Research Bureau; I.W.R.B. Publication No 32. 69-75.

Vollenweider, R. and J. Kerekes (1982). *Eutrophication of Waters: Monitoring, Assessment and Control*. Paris, Organisation for Economic Cooperation and Development.

Vollenweider, R. A. (1976). Advances in defining critical loading levels for phosphorus in lake eutrophication, *Memorie dell'Istituto Italiano d'Idrobiologia*, **33**: 53-83.

Vollenweider, R. A., R. Marchetti, et al. (1992). *Marine Coastal Eutrophication*. Amsterdam, Elsevier.

Wilson, H. M., M. T. Gibson, et al. (1993). Analysis of current policies and alternative strategies for the reduction of nutrient loads on eutrophicated lakes: the example of Slapton Ley, Devon, *Aquatic Conservation: Marine and Freshwater Ecosystems*, **3**: 239-251.

Wolter, K. D. (1994). Phosphorus precipitation, ed, *Restoration of Lake Ecosystems - A Holistic Approach.*, Slimbridge, Gloucestershire, International Waterfowl Research Bureau; I.W.R.B. Publication No 32. 63-69.

Wooding, M. G., A. D. Zmuda, et al. (1994). *Crop discrimination using multi-tempral ERS-1 SAR data*, Space at the Service of our Environment, Hamburg, Germany, European Space Agency.

WPCF (1983). *Nutrient Control*. 601 Wythe Street, Alexandria, VA USA, Water Pollution Control Federation.

Wright, P., P. Saich, et al. (1994). *Crop monitoring with ERS-1 SAR in East Anglia,* Space at the Service of our Environment, Hamburg, Germany, European Space Agency.

Yates, P. and J. M. Sheridan (1983). Estimating the effectiveness of vegetated floodplains as nitrate and orthophosphorus filters, *Agriculture, Ecosystems and Environment,* **9**: 301-314.

Young, S. N., W. T. Clough, et al. (1988). Changes in plant community at Foxcote Reservoir following use of ferric sulphate to control nutrient levels, *Journal of the Institute of Water Environmental Management,* **2**: 5-12.