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AN EVALUATION OF ALTERNATIVE MANAGEMENT STRATEGIES FOR SHALLOW EUTROPHICATED LAKES AND RESERVOIRS

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AN EVALUATION OF ALTERNATIVE MANAGEMENT STRATEGIES FOR SHALLOW EUTROPHICATED LAKES AND RESERVOIRS

by

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"In the first place, we have to face that any observed phenomenon can in theory be explained from many different mechanisms. Trivial as it is, this appears hard to keep in mind in every day scientific life. The well known problem associated with it is that one tends to become attached quickly to ones own tentative hypothesis causing an inevitable bias in further research... There is then the imminent danger of an unconscious selection and of a magnifying of phenomena that fall into harmony with the theory and support it and an unconscious neglect of phenomena that fail of coincidence."

Scheffer and Beets, 1994 p.119

THE P SONG

Cycle me, cycle me, you know where
Into the oceans and through the air
And if you don't cycle me in the right place
I'll weed up your rivers and eutroph your lakes.

Professor Bob Garrels
[Cited in Helgeson, 1992]

An Assessment of Alternative Management Strategies for Shallow Eutrophicated Lakes and Reservoirs

Helen M. Wilson

Abstract

External phosphorus loadings on a range of freshwater Sites of Special Scientific Interest (SSSIs) thought to be affected by eutrophication were predicted utilising export coefficients. The effect of such loadings on lake trophic status was evaluated using the Vollenweider-OECD eutrophication model. Estimates of the relative contribution of phosphorus from various sources enabled the selection of possible reduction strategies on a site specific basis. The effect of reduction strategies on trophic status was predicted with the aid of the model.

It was established that diffuse agricultural losses of phosphorus are the most common source of enrichment. However, consented discharges of sewage effluent appear to affect a significant number of sites. Phosphorus in urban runoff is a notable source for lakes situated in less rural areas. Agricultural point sources significantly influence a small number of lakes.

A critique of the methodology concluded that the use of separate export coefficients for organic and inorganic sources may be useful for identifying appropriate management strategies, but that the scientific basis for such an approach is dubious. In addition, the employment of agricultural returns for data on livestock levels may introduce an unacceptable degree of error into the calculations. The Vollenweider-OECD model appears to predict the trophic status of the lakes under assessment reasonably well, but there is a need for a reliable method of ascertaining loading reduction objectives.

A review of current legislation and policy applicable to the alleviation of eutrophication of freshwater SSSIs encompassed laws relating to nature conservation, to water quality, and to agricultural extensification. It confirmed that legislation which directly addresses the problem is nonexistent, but that certain laws may be applied in a piecemeal manner. In general, the form of nature conservation protection adopted in this country is not designed to prevent deterioration of water quality. An aspect of eutrophication control which may prove to be the most problematical in legislative terms is the regulation of diffuse agricultural sources of phosphorus. Proposals for changes in law and policy on this issue included the establishment of a catchment-wide scheme, specifically designed to reduce diffuse agricultural losses of phosphorus, and targeted at eutrophicated SSSIs.

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List of Abbreviations

a	Annum
AAP	Arable Area Payment
AONB	Area of Outstanding Natural Beauty
AOS	Area of Search
BAP	Bioavailable phosphorus
BCC	Birmingham City Council
BOD	Biochemical oxygen demand
ca	Capita
CAP	Common Agricultural Policy
CC	Countryside Commission
CCW	Countryside Council for Wales
CLA	Country Landowners Association
CO	Conservation Officer
COPA	Control of Pollution Act
CPS	Countryside Premium Scheme
CSO	Combined sewer overflow
DoE	Department of the Environment
EA	Environmental Assessment
EC	European Community
EDTA	Ethylenediaminetetraacetic acid
EN	English Nature
EPA	Environmental Protection Act
ESA	Environmentally Sensitive Area
EU	European Union
FWS	Farm Woodland Scheme
FYM	Farmyard manure
G	Guide
ha	Hectare
HMIP	Her Majesty's Inspectorate of Pollution
HS	Habitat Scheme
I	Imperative
IFE	Institute of Freshwater Ecology
ITE	Institute of Terrestrial Ecology
IWEM	Institution of Water and Environmental Management
JNCC	Joint Nature Conservation Committee
LCA	Life cycle assessment
LFA	Less Favoured Area
LPA	Local Planning Authority
MAFF	Ministry of Agriculture, Fisheries and Food
Ml	Megalitre
N	Nitrogen
NAO	National Audit Office
NC	Nature Conservancy
NCC	Nature Conservancy Council

List of Abbreviations (Continued)

NCCS	Nature Conservancy Council for Scotland
NCO	Nature Conservation Order
NERC	Natural Environment Research Council
NFU	National Farmers Union
NNR	National Nature Reserve
NP	National Park
NRA	National Rivers Authority
NSA	Nitrate Sensitive Area
NSCA	National Society for Clean Air and Environmental Protection
NT	National Trust
NTA	Nitrilotriacetic acid
NVZ	Nitrate Vulnerable Zone
OAS	Organic Aid Scheme
OECD	Organisation for Economic Cooperation and Development
P	Phosphorus
PCA	Polycarboxylic acid
PDO	Potentially damaging operation
PDR	Permitted Development Right
p.e.	Population equivalent
PPG9	Planning Policy Guidance Note No.9
RCEP	Royal Commission on Environmental Pollution
RNAS	Royal Naval Air Station
RWA	Regional Water Authority
SA	Soil Association
SAC	Special Area of Conservation
SNH	Scottish Natural Heritage
SPA	Special Protection Area
SRP	Soluble Reactive Phosphorus
SSSI	Site of Special Scientific Interest
STPP	Sodium tripolyphosphate
STW	Sewage treatment works
SWQO	Statutory Water Quality Objective
SWT	Surrey Wildlife Trust
TON	Total oxidisable nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UC	Use Classification
UWWT	Urban Waste Water Treatment
WOAD	Welsh Office Agriculture Department
WPZ	Water Protection Zone
WRA	Water Resources Act
YDNP	Yorkshire Dales National Park
YWA	Yorkshire Water Authority

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Author's Declaration

At no time during the registration for the degree of Doctor of Philosophy has the author been registered for an award at any other University.

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Publications

1. **O'Sullivan, P.E. and Wilson, H.M. [1992]** Evaluation of UK Policies for the Restoration of Eutrophicated Wetlands: The Example of Slapton Ley, Devon. **In:** Finlayson, C.M. (Ed) *Integrated Management and Conservation of Wetlands in Agricultural and Forested Landscapes.* Proceedings, IWRB Workshop, Třeboň, Czechoslovakia, 25th to 31st March 1992. IWRB Special Publication No.22, 69-73.

2. **Wilson, H.M., Gibson, M.T. and O'Sullivan, P.E. [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.

3. **Wilson, H.M. and O'Sullivan, P.E. [1993]** The Control of Eutrophication of Small Shallow Lakes in the United Kingdom: The Legal Framework. *Verhandlungen der Internationalen Vereinigung Limnologie* **25**, 461-464.

Presentations and Conferences Attended

1. **Agriculture, Conservation and Land Use.** Conference, University of Wales, Cardiff, Wales. 16th January 1992.

2. The Control of Eutrophication of Small Shallow Lakes in the United Kingdom: The Legal Framework. Poster Presentation, **XXV International Association for Theoretical and Applied Limnology Congress**, Barcelona, Spain, 21st to 27th August 1992.

3. An Assessment of Alternative Management Strategies for Shallow Eutrophicated Lakes and Reservoirs. Oral Presentation, **Slapton Research Seminar**, Slapton, Devon, 21st November 1992.

4. **Meres Forum**, First Meeting, Shrewsbury, Shropshire, 7th May 1993.

5. **Agri-Environmental Options: Water Fringes Workshop**, Silsoe College, Cranfield University, Bedfordshire, 28th September 1993.

6. **Meres and Mosses Forum**, Second Meeting, Shrewsbury, 24th March 1994.

7. Review - The Macrophytes. Oral Presentation, **Slapton Research Seminar**, Slapton, Devon, 26th November 1994.

8. Policy and Legislation Relating to Eutrophication of Freshwater SSSIs. Oral Presentation, **Meres and Mosses Forum**, Third Meeting, Shrewsbury, 6th April 1995.

Chapter 1

Introduction

1.1 Background

The Nature Conservancy Council for England, also known as English Nature (EN), is the statutory body concerned with nature conservation in England. Its functions include advising government on the development and implementation of policies relating to nature conservation, and the designation of areas of botanical and geological interest. In EN's opinion, eutrophication, or nutrient enrichment, is the principal environmental problem affecting inland waters, and, as long ago as 1977, they described it as "perhaps the most insidious and pervasive influence on wetland wildlife" [EN, 1992 p.1]. Over one hundred freshwater Sites of Special Scientific Interest (SSSIs) are considered to be affected by eutrophication [EN, 1991b]. The extent to which English Nature is able to protect such sites is, however, limited, in that the nutrients involved usually originate outside SSSI boundaries, mainly from agriculture and sewage effluent.

In order to provide informed advice, and to influence government policy on this issue, EN provides funds for, or carries out, a range of research projects related to eutrophication. As part of this research programme, the present study is expected to contribute recommendations, not only on management strategies for specific lakes and reservoirs affected by nutrient enrichment, but also on future policy for the control of eutrophication.

1.2 Aims and Outline of the Study

The primary aim of this study is to investigate and evaluate management strategies for shallow, eutrophicated lakes and reservoirs in England, with a view to identifying those best suited to the restoration of such ecosystems to ecologically desirable conditions. The project will produce recommendations on management and future policy with respect to phosphorus, on a site-specific and a general basis.

For each site, the potential total phosphorus loading will be quantified, using an export coefficient approach. An assessment of a range of possible strategies to reduce phosphorus loads will be made, using Vollenweider's steady state model to evaluate the effects on individual lakes. These strategies may include tertiary treatment of sewage to remove phosphorus, diversion of sewage effluent, changes in agricultural practices, the establishment of riparian nutrient buffer zones, and the use of detention ponds for urban runoff. The aim will be to recommend the most appropriate combination of strategies for each site.

The methodology adopted in the study will be critically examined, with a view to assessing its applicability to the evaluation of present phosphorus loadings and the potential effects of reductions. In particular, the scientific validity of the export coefficient approach to predicting nutrient loads will be explored, and the suitability of the Vollenweider model for application to shallow English lakes of a range of trophic status will be assessed.

A review will be undertaken of existing legislation related to eutrophication, that is, those laws concerned with nature conservation, with agricultural extensification, and with water quality. The aim will be to identify those laws which may be enforced or adapted to implement the proposed strategies to reduce phosphorus loads. Such an analysis is unique, in that a synthesis of apparently disparate legislation will be related to nature conservation goals. Recommendations will be made as to the necessary changes in existing legislation, as well as proposals for future policy enforcement.

Through its assessment of strategies to reduce phosphorus loadings, the project will assist EN in advising government on future policy to combat the problem of eutrophication. Its critique of the methodology will add to the debate surrounding the assessment of eutrophication, especially in relation to water quality objectives.

The aims of this study are based on a number of concepts and processes which warrant early explanation. It is now well-established that the lake-watershed ecosystem concept, discussed in Section 1.3, provides the basis not only of a useful and logical way in which to study interactions between land and water in an interdisciplinary manner, but also a practical means of water quality and resource management. An understanding of the process of eutrophication, and the impact of anthropogenic disturbances and consequent material transfers within the ecosystem, outlined in Section 1.4, is necessary prior to assessment of management options available for its control.

Finally, the need for restoration and conservation of eutrophicated lakes is often considered purely from an anthropocentric viewpoint, that is, the desire to preserve water resources for human use. In this study the intrinsic value of aquatic ecosystems is assumed, and the ecological basis for conservation is presented in Section 1.5.

1.3 The Lake-Watershed Ecosystem Concept

The ecosystem is the basic functional unit in ecology [Odum, 1971]. The term is a contraction of 'ecological system' and was adopted by Tansley in 1935 [Spurr, 1969]. It describes

> "any unit that includes all of the organisms... in a given area interacting with the physical environment, and with each other, so that a flow of energy leads to a clearly-defined trophic structure, biotic diversity, and material cycles... within the system." [Odum, 1971, p.8]

An ecosystem is connected to the surrounding biosphere by a series of inputs and outputs, which may include energy, water, gases, organic matter and nutrients [Bormann and Likens, 1969]. Measurement of critical inputs and outputs, such as nutrients, may be problematical, particularly when the boundary of an ecosystem is ill-defined. However, since nutrient cycles are strongly related to the hydrological cycle, in many instances the watershed ecosystem is the logical unit of analysis for the evaluation of such inputs and outputs.

The lake-watershed ecosystem concept appears to originate from the work of Bormann, Likens and colleagues on the Hubbard Brook catchment [O'Sullivan, 1979]. During this study of undisturbed hardwood forests in the northern USA, hydrological and chemical inputs, outputs, and net internal change were measured [Bormann and Likens, 1969]. An area of forest was later clear-cut in an experiment to determine the effect of such treatment on nutrient and hydrological cycles. The significance of the Hubbard Brook study is evident in at least two respects:

> "To begin with, it represents the first attempt at comprehensive investigation and measurement of all energetic and material pathways in a discrete watershed system. Secondly, it involves the integration of the approach and findings of a number of previously separate areas of research, in particular those of ecology, hydrology and meteorology." [O'Sullivan, 1979 p.273]

Bormann and Likens applied the results of their work to the problems of watershed management and the impact of human activities on water quality [O'Sullivan, 1979]. They concluded that anthropogenic changes in watersheds tend to increase sediment yield and associated nutrient losses. Likens [1972] extended the watershed ecosystem concept to include freshwater lakes, analysing the system in terms of inputs and outputs of energy and matter, illustrated in Fig.1.1.

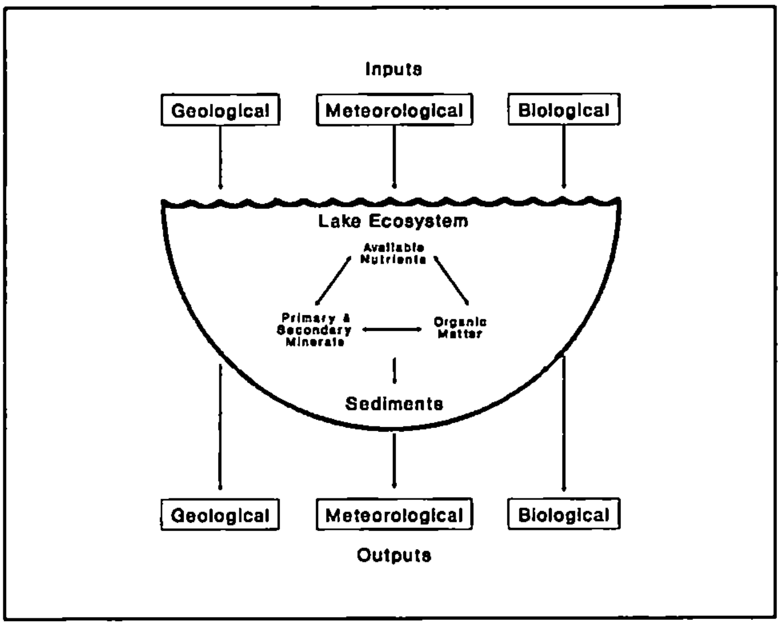

Fig.1.1: Model of Material Pathways in a Lake Ecosystem

Adapted from Likens [1972]

Inputs from meteorological, geological and biological sources are utilised and circulated within the lake ecosystem. Outputs include not only outflowing water and its dissolved and suspended load, but also lake sediments. A close relationship exists between the biogeochemical character of a lake and the biogeochemical nature of its catchment, which is reflected in its sediments [O'Sullivan, 1979]. Oldfield [1977] extended the scope of the lake-watershed ecosystem concept to include lake sediments, which are seen not only as a store of matter, but also of information. Sediments record both lake and catchment history, so that, given modern dating techniques, current environmental problems such as eutrophication, may be closely assessed on a time-related basis.

The lake-watershed ecosystem approach to water quality and resource management is now, at least partially and in theory, brought into force through the efforts of the National Rivers Authority (NRA) to produce and implement Catchment Management Plans. Since the NRA is organised by hydrological regions, the catchment is seen as the natural management unit [Chandler, 1994]. However, a distinction may be drawn between catchment planning and catchment control [Newson, 1991]. The latter approach to water management on a catchment-wide level is seen in the protection zone concept, which aims to address a particular problem in a more proscriptive manner. In dealing with eutrophication, both approaches may be necessary, in order to integrate long-term nutrient export control policies into the planning process, while implementing strategies designed to restrict activities detrimental to water quality.

1.4 The Process of Eutrophication

As outlined above, the biogeochemical character of a lake is closely related to the biogeochemical nature of its catchment [O'Sullivan, 1979]. The nutrient status, or trophic nature, of a water body is, therefore, largely a function of geological, topographical and climatic factors, modified by the degree of human influence within the catchment [Canfield *et al.*, 1989]. Lakes may be broadly classified according to the nutrient status of their waters as oligotrophic or eutrophic [Rast, Smith and Thornton, 1989].

These terms were originally used by Weber in 1907, to describe the nutrient status of peat bogs, but were subsequently applied to lakes by Thienemann in 1918 and Naumann in 1919. Oligotrophic lakes possess low nutrient concentrations, productivity and biomass, generally support greater species diversity, and tend to sustain good overall water quality. In contrast, eutrophic water bodies are characterised by high levels of productivity and biomass, frequent occurrence of algal blooms, development of an anoxic hypolimnion during periods of thermal stratification, generally lower species diversity, enhanced growth of littoral plants and, periodically, poor water quality. Specific plant and animal communities are associated with eutrophic or oligotrophic waters. These categories are not discrete, however, but represent the extremes of a continuum of nutrient status.

In natural lake ecosystems, trophic status is determined by geological, biological and meteorological rates of inputs from the catchment [Likens, 1972]. The process of eutrophication consists of the addition of increased levels of nutrients, mostly as a result of human activity. In its early stages, eutrophication leads to an increase in biomass and productivity within the existing community structure [Moss, 1988]. Continued influx of elevated levels of nutrients may then produce a shift, including a reduction in species diversity, and deterioration of water quality. The specific effects of eutrophication on the ecology of lakes will be assessed in Section 1.5.

The degree of eutrophication of a water body does not, however, depend entirely on the external nutrient loading [Canfield *et al.*, 1989]. Lake basin morphology, flushing rate and internal phosphorus cycling can significantly affect overall productivity. A depth-dependent relationship exists between areal nutrient loading and hypolimnetic oxygen consumption. On the whole, deeper lakes exhibit higher dissolved oxygen concentrations in the hypolimnion than shallow bodies of equal productivity. The flushing rate, which is dependent on rainfall and basin volume, determines the extent to which nutrients can accumulate. A high inflow to basin volume ratio may prevent phytoplankton populations from proliferating to nuisance levels.

Sediments play an important role in the overall phosphorus metabolism of lakes, acting both as sink and source [Boström *et al.*, 1988]. In most lakes, there is a net deposition of phosphorus in the sediments, mainly by co-precipitation with iron (III). Mobilisation of sediment phosphorus is affected by a number of environmental factors, of which temperature, pH and redox potential are thought to be the most important. A rise in temperature primarily produces an indirect effect, by increasing biological activity. During periods of high primary production, the pH of the water column rises, reducing the phosphorus-binding capacity of iron and aluminium compounds. Other physical factors which are known to induce phosphorus release are wind turbulence and bioturbation [Canfield *et al.*, 1989]. The release of phosphorus from sediments can significantly influence the trophic status of lakes, especially in shallow, homothermal bodies, where it may constitute 10% to 30% of total loading [Wetzel, 1983]. For some lakes, the effectiveness of eutrophication control measures, based on reduction of the external nutrient load, may be delayed for many years by the continued supply of phosphorus from the sediments [Sas, 1989]. In-lake treatments may be necessary in such cases, in order to improve water quality.

Eutrophication of fresh waters is often associated with major ecological changes, such as the loss of submerged macrophyte communities and the production of algal blooms. These changes are, however, more symptomatic of the eutrophication of naturally productive ecosystems. From the point of view of nature conservation, the enrichment of oligotrophic waters, through the addition of much lower levels of increased nutrient influx, may be equally important [Wilson, Gibson and O'Sullivan, 1993].

Confusion sometimes arises between eutrophication and the processes of paludification and successional development of lakes, in which in-filling brings about transformation to terrestrial ecosystems [O'Sullivan, 1995]. However, these forms of ontogeny usually take many hundreds of thousands of years to occur, are largely irreversible [Rast and Ryding, 1989] and, after the initial stages, do not necessarily involve a higher level of nutrients or productivity [Wetzel, 1983]. In contrast, eutrophication is a process which occurs relatively rapidly, is reversible and, by definition, involves increased levels of nutrient inputs, but without any appreciable change in depth or form of the basin [Vallentyne, 1974]. The use of the term 'accelerated eutrophication' arises from such misconceptions and implies the speeding up of a natural process. Although instances of natural eutrophication do occur, in the context of this study eutrophication refers to the far more common, human-induced process.

A further area of confusion related to eutrophication concerns the extent of the problem in the UK. A report by Lund [1980], commissioned by the Soap and Detergent Industry Association, concluded that a eutrophication problem does exist, but that it is highly localised. This idea was perpetuated by the government's interpretation of a further report by Lund and Moss [1990]. The Royal Commission on Environmental Pollution

(RCEP) Report on Freshwater Quality noted that the government's view on the subject was that "eutrophication is a localised problem in the UK" [RCEP, 1992 p.64]. This was based on the idea that eutrophication is only a problem if it "causes an increase in the growth of algae or other plants and shows a noticeable deleterious effect on the environment" [*ibid.*].

Perceptions of eutrophication as a problem in the UK appear to depend on the criteria adopted for assessment. A comparison of orthophosphate concentrations in rivers with those of natural background levels, detailed in the RCEP Report, reveals that virtually all major rivers in the NRA's Anglian, Thames and Severn-Trent regions are heavily enriched, as are a high proportion in the North-West, Yorkshire, Southern and Wessex regions. Taking into consideration many other bodies of evidence, the RCEP concluded that "nutrient enrichment would appear to be widespread over large parts of the country" [RCEP, 1992, p.64]. Additionally, EN in its submission to the House of Lords Select Committee on Municipal Waste Water Treatment, stated that nutrient enrichment is the main environmental problem for inland waters in Great Britain [House of Lords, 1991]. A total of 112 river, canal and lake SSSIs were considered by EN to be affected by eutrophication.

The most important nutrient elements involved in eutrophication, under the majority of conditions, are phosphorus and nitrogen [Wetzel, 1983]. Of these, phosphorus is generally the first to impose limitations on productivity, because the amount present in biologically available forms, relative to other required nutrients, is small [Lee, Rast and Jones, 1978]. The tendency for phosphorus to become scarce in nature is largely because its cycle does not include a gaseous phase, between its source in the lithosphere and its sink in the ocean [O'Neill, 1985]. Nitrogen, on the other hand, is abundant in the atmosphere, which serves as a reservoir for biological fixation of this element.

The limiting nutrient concept maintains that productivity is restricted by the essential element which is most scarce in the environment, relative to the specific requirements of the organisms involved [Rast, Smith and Thornton, 1989]. A widely cited atomic ratio for nitrogen and phosphorus uptake and use by algae is 16N:1P, which corresponds to a mass ratio of 7.2N:1P. Based on the limiting nutrient concept, an effective long-term eutrophication control measure is therefore to reduce the external phosphorus load to a water body. According to Golterman [1973], even if phosphorus is not the limiting nutrient, it is the only essential element which can easily be controlled. Additionally, in those lakes for which nitrogen is limiting, a reduction in the relative concentration of this element to below about 7N:1P may further encourage the growth of nitrogen-fixing blue-green algae and consequent toxic blooms [Rast, Smith and Thornton, 1989]. For these reasons, this study is concentrated primarily on the reduction of phosphorus, rather than nitrogen, loads.

The bioavailability of phosphorus, or the extent to which it may be utilised immediately by algae, varies according to source and also within categories. Agricultural runoff generally contains phosphorus of lower bioavailability than that in sewage or leakages from animal waste and silage stores [Källqvist and Berge, 1990]. Table 1.1 illustrates a range of values for total phosphorus and bioavailable phosphorus in these categories.

Table 1.1: Mean Values for Total Phosphorus (TP) and Bioavailable Phosphorus (BAP) in Different Categories of Source

Adapted from Källqvist and Berge [1990]

Category	**TP (μg/l)**	**BAP (%)**
Cereal Cultivation	706	37
Ensilage Leakage	606×10^3	59
Manure Leakage	67×10^3	79
Raw Sewage	4100	60
Sandfiltered Sewage	1255	95
Detergent	105	76

Leakages from ensilage and manure are extremely rich in total phosphorus and the bioavailability of the element from these sources is also high [Källqvist and Berge, 1990]. Raw sewage, although exhibiting very high concentrations of total phosphorus, does not contain proportionately as much bioavailable phosphorus as filtered sewage, since much of the organic component is removed from the latter. Detergents generally contain sodium tripolyphosphate (typically 15% of constituents by weight), which is largely bioavailable. The use of detergents containing phosphates increases the overall soluble phosphorus content of sewage effluent.

Early research and predictive models assumed bioavailable phosphorus to be a constant proportion (20%) of total phosphorus transported [Sharpley and Smith, 1990]. It is now known that the majority (90% to 95%) of dissolved phosphorus and a variable proportion (10% to 90%) of particulate phosphorus is immediately available for algal uptake [Sharpley, Daniel and Edwards, 1993]. Bioavailability is a dynamic function of soil and runoff characteristics, thus producing the wide variations in the proportion of particulate phosphorus which may be assimilated [Sharpley and Smith, 1990]. Transformations between dissolved and particulate forms occur during transport in runoff and streamflow. The direction and extent of such transformations will vary depending on such factors as the adsorption capacity of the suspended and stream bottom material contacted, the concentration of dissolved phosphorus and rate of stream flow.

Several authors [e.g. Boström, Persson and Broberg, 1988; Källqvist and Berge, 1990; Sharpley *et al.*, 1992] suggest that the bioavailability of phosphorus from different sources should be estimated in nutrient control programmes. It is maintained that this approach would improve prediction of biological response to external loadings. However, phosphorus forms which are not bioavailable may provide a long-term, but variable source in streams and lakes [Sharpley *et al.*, 1992]. Additionally, bioavailability of external loads will differ between lakes [Böstrom, Persson and Broberg, 1988]. Factors such as high pH and anoxic conditions in the hypolimnion, which favour phosphorus mobilisation, will increase the bioavailability of external loads of aluminium- and iron-bound particulate forms. As such, total phosphorus input may be a more reliable reflection of change in trophic state than bioavailable phosphorus loading.

1.5 The Need for Conservation and Restoration

As described in Section 1.4, among lakes undisturbed by human activity, there exists a continuum of trophic states [Rast, Smith and Thornton, 1989]. Particular biotic communities tend to be associated with each stage of this continuum. Oligotrophic lakes, although classed as unproductive, generally sustain a diverse assemblage of plants and animals. Naturally eutrophic lakes tend to exhibit an abundant, but less diverse community. Through eutrophication many of these lakes are being forced, over a relatively short time period, to change to new states, with consequent loss of associated flora and fauna. The ecological interactions maintaining and sustaining lentic communities underpin an understanding of why and how such systems need to be restored and conserved.

Primary production in aquatic ecosystems is controlled by two principal mechanisms, that is the rates of supply of light and of nutrients [Valiela, 1991]. These factors interact to produce a set of conditions in the water column to which certain species are adapted. While fluctuations in the levels of light and nutrients do occur, both spatially and temporally within the same lake, such changes lie within definable ranges, to which the community as a whole is adapted [Reynolds, 1984]. Long-term changes in the nutrient supply to a water body may occur naturally, over decades to millennia, with a variety of causes. Eutrophication represents a relatively abrupt increase in nutrient inputs, with consequences for the whole ecosystem.

According to Mann [1991], the biota of lakes and reservoirs may be classified in terms of three types of community: pelagic (open water), benthic (sediment-dwelling) and littoral (shallow water). The pelagic community consists of phytoplankton and zooplankton, suspended relatively passively in the water, and nekton, which possess some mode of active motility. The benthic community is comprised of animals, bacteria, fungi, protozoans and algae. The littoral zone, the extent of which is determined by availability of light, is occupied by a variety of emergent, submerged and floating-leaved macrophytes, which in turn provide food for an array of herbivores and detritivores, and habitat for epiphytes, small fish and invertebrates.

Figure 1.2 illustrates feeding transfers between the most common groups of phytoplankton and zooplankton in the pelagic community, arranged in order of size. Similar transfers occur between biota in the benthic and littoral zones. Although these communities are described separately, they are inextricably linked by numerous interactions [Mann, 1991]. The benthic community, for instance, depends for its nutrients on sinking live phytoplankton, fragments of dead plankton, faecal pellets and various forms of particulate organic matter from the pelagic community. Another important interaction, especially in the context of eutrophication, is the role of the littoral zone in intercepting material flows from the land [Carpenter and Lodge, 1986].

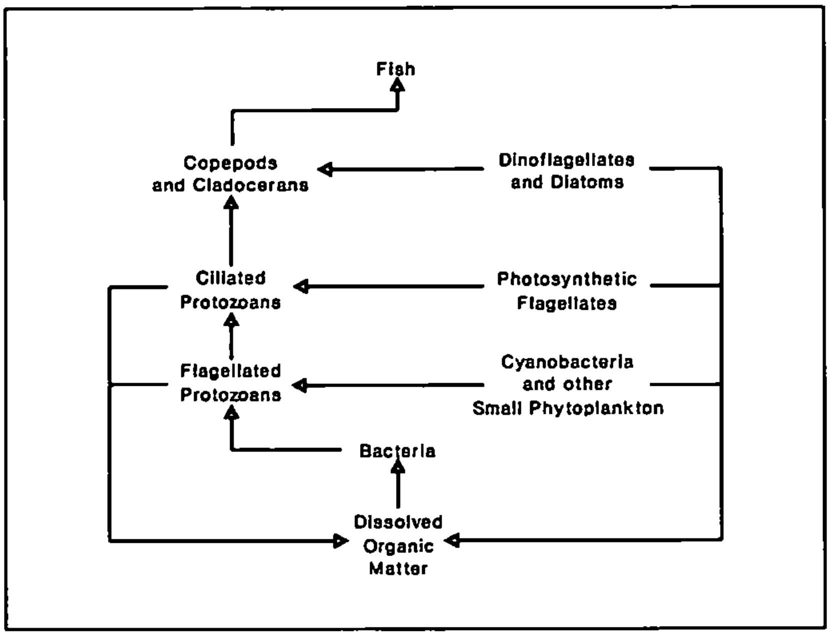

Fig.1.2: Feeding Transfers in the Pelagic Community

Adapted from Mann [1991]

Phytoplankton are an extremely important component of the pelagic community [Jeffries and Mills, 1990]. Their productivity is vital in sustaining the food web (Fig.1.2), and the effects of their abundance and diversity on the overall ecology of lakes are profound. Phytoplankton populations demonstrate seasonal dynamics, with a succession of species over the year. As day length, light intensities and temperatures increase in late winter, the abundance of various diatom species also increases, reaching a maximum in spring [Moss, 1988]. Diatom blooms utilise dissolved nitrate, phosphate and silicate until a shortage of one or more of these nutrients causes numbers to decline. Chrysophytes, which include species able to grow in nutrient depleted water, may overlap with and succeed spring diatom growth. During the summer, when nutrient levels in the water column are periodically low, but rapid cycling of nitrogen and phosphorus occurs, populations are generally composed of green algae and Cryptomonads. Low dissolved oxygen concentrations in late summer seem to favour the growth of blue-green algae. As dissolved inorganic nitrogen becomes scarce, nitrogen-fixing blue-greens proliferate.

Seasonal succession of algal populations is controlled by a number of factors, some of which are poorly understood [Moss, 1988]. However, the trophic structure of the pelagic zone (Fig.1.2) is known to exert a strong influence over algal populations. In spring, zooplankton numbers increase in response to rising temperatures and contribute to the decline of diatom populations through grazing. Young fish, spawned in large numbers in late spring, are often zooplanktivores. Consequently, the numbers of zooplankton are reduced by fish predation, thus lowering grazing pressure on phytoplankton populations. As the zooplanktivorous fish are consumed by piscivores, there may be a recovery of cladoceran populations, such as *Daphnia*, which, in turn, reduce the numbers of smaller algae.

These interactions, between nutrient levels and phytoplankton, zooplankton and fish populations, may be severely disrupted by eutrophication. However, another component of the ecology of lakes, the littoral macrophytes, also plays an important role in maintaining the balance of the system. The effects of eutrophication on submerged macrophytes, in particular, is integral to this overview.

The term macrophyte covers all the large aquatic plants, not only flowering species, but also algae, such as stoneworts and blanket weed, mosses and liverworts, and ferns [Jeffries and Mills, 1990]. This varied assortment of plants grows together, responding to the same environmental pressures and changes. Macrophytes are generally classified according to whether or not they are attached to the sediment [Wetzel, 1983]. Those which are rooted may be emergent, floating-leaved or submerged. Species which live unattached, within or upon the water, are referred to as free-floating macrophytes.

The productivity and distribution of submerged macrophytes are limited by a range of factors. The most important of these are light, nutrients, temperature, hydrostatic pressure, and wave action [Hutchinson, 1975]. Adaptation to light limitation in submerged macrophytes takes various forms, the degree of which, in particular species, influences the optimal water depth for biomass and production [Spence, 1975]. Diminution of radiant energy by attenuation and absorption increases with depth. Populations of plankton in the surface waters may further reduce the level of light received by submerged macrophytes [Wetzel, 1983]. The maximum depth at which autotrophs remain productive clearly depends therefore on the transparency of the water [Hutchinson, 1975]. In Europe, the maximum depth for angiosperms, in clear water, appears to be about 6.5m, whereas charophytes have been recorded at 12m, and bryophytes at 20m [Spence, 1975].

Fluctuations in the biomass of submerged macrophytes produce important consequences for lake ecosystems, because of their effects on the physical and chemical environment, and the littoral biota [Carpenter and Lodge, 1986]. As the largest sessile organisms in freshwater ecosystems, macrophytes exert important physical influences on light, temperature, water flow, and substrate. The growth and decomposition of submerged macrophytes produce biogeochemical changes in the levels of dissolved oxygen, inorganic and organic carbon, nutrients, and sediment accretion. Macrophytes remove inorganic carbon from the water by assimilation, so that their metabolism may strongly influence dissolved inorganic speciation, and pH. During active growth, macrophytes release between 1% and 10% of their photosynthetically-fixed carbon to the water as dissolved organic compounds, which contributes to the metabolism of bacteria and epiphytic organisms.

One of the important questions about the role of littoral zones in lake ecosystems is the extent to which macrophyte beds act as sources or sinks of nutrients. During periods of active growth, macrophyte stands act as a net sink for phosphorus [Howard-Williams, 1981], while during senescence they are a net source [Landers, 1982]. In general, they are sinks for particulate matter, and sources for dissolved phosphorus and organic carbon [Carpenter and Lodge, 1986].

The importance of submerged macrophytes in biotic interactions is proportional to their biomass and productivity [Carpenter and Lodge, 1986]. Macrophytes are colonised by a rich array of microbes, algae, and consumers. Productivity of the epiphyte complex ranges from 4% to 93% of that of their hosts. Epiphytes appear to be much more active than macrophytes in dissolved nutrient exchange with the water, gaining only a relatively small proportion from their hosts [Howard-Williams, 1981]. The release of extracelluar organic matter by macrophytes, including dissolved carbon compounds, is primarily utilised by epiphytes, but the possibility exists that the products may also affect the phytoplankton [Carignan and Kalff, 1982; Godmaire and Planas, 1983]. It has also been suggested that the effect may extend to an influence upon the species composition of phytoplankton [Godmaire and Planas, 1986]. High invertebrate densities associated with macrophytes may stem from the availability of epiphytes as food, and the use of live plants for shelter and nutrition [Carpenter and Lodge, 1986].

The impact on submerged macrophytes of avian, mammalian, fish, and invertebrate grazing may be significant, but has rarely been quantified. Waterfowl often consume large portions of peak standing crop of submerged plants [Jupp and Spence, 1977b]. For instance, during the winter months, the leaves of submerged macrophytes provide the bulk of the diet of mallard, coot, and pochard, and a smaller component during the summer [Cramp and Simmons, 1977]. The seeds, fruits, and turions of submerged macrophytes are important food sources for goldeneye, moorhen, and tufted duck. Mute swans consume mostly submerged vegetation all year round, and, as adults, need 3.6 kg to 4 kg of fresh plant matter a day. Nearly all of the littoral fish of Europe devour some plant material; submerged macrophytes comprise a significant proportion of the diet of roach, rudd, and ide [Carpenter and Lodge, 1986]. The diet of rudd, one of the most common littoral fish of eutrophic European lakes, includes 65% to 90% submerged macrophyte tissue.

Numerous studies of eutrophication have shown that submerged macrophytes often assume an increasingly greater importance to the total primary productivity of lakes until, under high nutrient loadings, and lake fertility, the whole system becomes subject to severe light attenuation. This limitation is usually associated with intense phytoplanktonic and epiphytic algal productivity [Wetzel, 1983]. Dense phytoplankton populations alone are sufficient to reduce light to a level where it is inadequate to support growth of submerged macrophytes [Jupp and Spence, 1977a]. Evidence exists, however, which shows that loss of submerged macrophytes is often the result of increased growth of, and shading by, epiphytes and filamentous algae, and that phytoplankton development is subsequent to, rather than causative of, this event [Phillips, Eminson and Moss, 1978]. With advanced eutrophication, numbers of submerged species decline and they may be eliminated. Phytoplankton and emergent macrophytes, with their attendant microflora, become the major contributors to lake productivity [Wetzel, 1983].

Since different species of submerged macrophytes are adapted to light limitation in varying degrees, eutrophication, and subsequent shading by epiphytes, filamentous algae, and phytoplankton, produces changes in the composition of the macrophyte community. This is well illustrated in the case of the Norfolk Broads, where in lakes possessing total phosphorus concentrations below $50 \mu g/l$, clear water and abundant submerged macrophytes are found [Moss, 1983]. In Broads where concentrations exceed 100μg/l, turbid water and dense phytoplankton populations occur. A transitional group of Broads with total phosphorus concentrations between 60μg/l and 100μg/l, support relatively dense phytoplankton populations and substantial stands of aquatic plants.

The type of vegetation found in all these Broads does, however, vary [Moss, 1983]. In the low phosphorus Broads, low-growing species of Charophyta and vascular plants exist, whereas in those with intermediate phosphorus levels, tall-growing submerged plants, such as *Myriophyllum spicatum* L., *Potamogeton pectinatus* L. and *Ceratophyllum demersum* L. are found. These species are able to grow rapidly to the water surface, there to form dense mats, often associated with filamentous algae, and floating-leaved, free-floating, or emergent plants. In the high phosphorus group of Broads, all aquatic macrophytes, except emergent taxa, have disappeared.

Community characteristics of the first and intermediate groups have been recognised, through sediment analysis, as those of former stages of the most eutrophic Broads [Moss, 1983]. These observations show that during eutrophication, plant communities change, so that those species more able to tolerate, or overcome, low light levels succeed those which cannot. A sub-phase has also been recognised, in which only free-floating or floating-leaved macrophytes exist.

These findings are confirmed by studies of other lake systems. A synthesis of results obtained through the study of a series of Swiss lakes belonging to differing trophic levels, led to a classification of the stages of colonisation by submerged plants [Lachavanne, 1985]:

1. <u>The Colonisation Stage</u> (ultra-oligotrophic to oligotrophic), when plants tend to occupy all the zone which may be colonised, restricted mainly by light and pressure, without forming dense stands.

2. <u>The Blooming Stage</u> (mesotrophic to eutrophic), when a marked enrichment of the flora, and an increase in density of vegetation occurs, but depth of colonisation decreases. Following this, tall hydrophytes become more common, and species diversity declines.

3. <u>The Regressive Stage</u> (highly eutrophic to hypereutrophic), when diversity and abundance of species decreases drastically. In extreme cases the submerged vegetation disappears completely, and local stands of emergents remain.

The cause of the demise of submerged and floating vegetation is linked directly and indirectly to eutrophication [Moss, 1983]. Dense phytoplankton populations not only account for the final demise of submerged species, but also, owing to their greater ability to take up inorganic carbon (as pH rises with photosynthetic activity) may inhibit the growth of submerged macrophytes. The disappearance of free-floating plants, such as *Lemna minor* L., which could not be affected by shading, is thought to be the result of nitrogen depletion by earlier-growing phytoplankton. Loss of the well-rooted, floating-leaved *Nymphaea* and *Nuphar* species again cannot be attributed to shading, nor to nitrogen depletion in the water. However, the early, submerged leaves of these species are particularly vulnerable to epiphyte growth and to consequent shading, and the absence of submerged plants as mechanical protection against wave action may render them more vulnerable. Where geese and swans are deprived of submerged plant food the less palatable lilies may be grazed.

With the demise of submerged plant communities, other changes occur [Moss, 1983]. Open water forms of zooplankton become more evident, in that those which find refuge from fish predation among plants decline. Benthic fauna becomes sparse, and is restricted mainly to chironimid larvae and oligochaetes. This reduced diversity is associated with the loss of habitat structure provided by plants. A change to increased numbers of roach and bream, and a scarcity of pike, perch and rudd, has also been linked to the demise of submerged plants. These last three species depend on plants for spawning, cover and as habitat for prey. Reduction in numbers of mute swans and coot may well be associated with a decrease in abundance of their preferred food, submerged plants.

The mechanisms by which the change to mainly phytoplankton productivity occurs during eutrophication therefore involves not only nutrients, plants and phytoplankton, but also a complex series of interactions with zooplankton, fish and other organisms. Balls, Moss and Irvine [1989] established that nutrient load alone did not displace submerged plant communities. Results by Irvine, Moss and Balls [1989] suggest that fish populations, in some circumstances, such as in the absence of suitable refuges, may exert major influences on the zooplankton community, independent of nutrient loading. A release from fish predation enables large and efficient-grazing zooplankton to proliferate. The consequent impact which these animals may produce in the phytoplankton community, during rising nutrient loading, forms the basis for an hypothesis on the reasons for the loss of submerged macrophytes with eutrophication.

The persistence of submerged plant beds at very high nutrient loadings, where phytoplankton are prevented from developing by cladoceran grazing, suggests that increased loadings *per se* may change the nature of the plant community from low to tall-growing forms, but that thereafter there exists a wide band of loadings (and consequent nutrient concentrations) through which either phytoplankton or higher plants may be maintained in alternative stable states [Irvine, Moss and Balls, 1989]. Each, once established, is maintained by a number of buffering mechanisms, breakdown of which is essential for the change from one to another. The authors state that their experiments and other evidence suggest that an aquatic plant community may be stabilised by, among other factors:

(a) luxury uptake of nutrients
(b) allelopathy
(c) shedding of leaves with heavy epiphyte burdens
(d) harbouring of large populations of grazers
(e) provision of refuges for such grazers against fish predation

In turn, a phytoplankton state, once established, may be maintained by factors such as:

(a) algal growth early in the season, compared with much later germination of aquatic seeds, or sprouting of turions and rhizomes, which then become shaded
(b) easier acquisition of carbon dioxide by phytoplankton, especially late in the season, when pH is high
(c) vulnerability of large cladoceran grazers to fish predation, in an open, unstructured environment
(d) production of large, inedible algae with low surface to volume ratios, at high nutrient concentrations

It may also be that lack of suitable habitat, or alternative food sources, reduces populations of larger fish, and encourages small zooplanktivores, such as roach. In the absence of predation by piscivores such as pike, large populations of roach can effectively maintain phytoplankton growth characterised by high biomass and persistence of cyanobacteria [Harper, 1992].

Reasons for the breakdown of the mechanisms which preserve plant or phytoplankton states are more speculative [Irvine, Moss and Balls, 1989]. Displacement of submerged macrophytes by phytoplankton seems to occur relatively rapidly, which supports the idea of a 'switch', rather than a linear effect of loading. Destruction or displacement of the plants by independent agents would be effective in causing this transition. In the Broads, such agents may include the effects of herbicides, mechanical damage by boats, or grazing by an introduced mammal, the coypu. Destruction of the cladoceran grazing community by pesticides is thought to be a further causative possibility. Once the phytoplankton state has been established, evidence suggests that a reversal simply by reduction of external nutrient loads may be difficult to effect. It has been established, again through work on the Broads, that a major factor in this resistance to change is release of phosphorus from the sediments [Phillips *et al.*, 1994]. Sediment resuspension, and consequent release of nutrients, caused by benthivorous fish, may enhance internal loadings [Breukelaar *et al.*, 1994], thus adding to the complexity of interactions among organisms subjected to the effects of advanced eutrophication.

From this overview of the effects of eutrophication on the ecology of lakes, it is apparent that, from the point of view of nature conservation, nutrient enrichment needs to be alleviated. Quite apart from any dangers to public health and to the quality of drinking water, produced by the development of toxic algal blooms in hypertrophic lakes and reservoirs, there is a real threat to many forms of wildlife in lakes across the trophic spectrum.

Chapter 2

Means of Achieving Reductions in External Phosphorus Loadings

Reductions in external phosphorus loadings are a prerequisite for the restoration of eutrophicated lakes and reservoirs [Holland *et al.*, 1989; Loehr, Ryding and Sonzogni, 1989; Sas, 1989; Cooke *et al.*, 1993], even in those cases where internal loadings are sufficient to maintain an elevated trophic state [Sas, 1989]. Several comprehensive publications have been produced [e.g. Dunst *et al.*, 1974; Ryding and Rast, 1989; Cooke *et al.*, 1993] which detail numerous ways in which such objectives may be achieved. It would be inappropriate, however, given the SSSI status of the lakes and reservoirs under consideration, to advise methods of load reduction which could be environmentally insensitive. Chemical methods, such as the direct addition of phosphorus-precipitating substances to inflowing waters [Clasen, Rast and Ryding, 1989], and substantial engineering projects, such as the construction of pre-dams [Benndorf and Pütz, 1987], are, therefore, not considered here.

Inputs of phosphorus to surface waters are derived from a variety of sources, mainly agricultural and urban drainage, and domestic and industrial wastewaters [Lee, Rast and Jones, 1978]. Estimates of the proportions contributed by these different sources, in the UK and selected European Community countries, are shown in Table 2.1. From this data, it may be concluded that domestic wastewater contributes a significant proportion of phosphorus to surface waters [Morse, Lester and Perry, 1993]. Agricultural inputs are also potentially high, especially from livestock. The figures for the latter, however, are based on an assumed loss rate of 10%, which, compared to the figure of 3% cited by Johnes, Moss and Phillips [1994], and other values referred to in Table 3.3, is extremely high and probably an overestimate.

The data suggest that, in the UK, about 43% of phosphorus losses to surface waters are contributed by agriculture as a whole. A previous estimate, cited in Lund and Moss [1990], amounted to 36%, of which 20% was thought to originate from livestock and 16% from fertiliser. The contribution of phosphorus from domestic wastewater was calculated to be 53%. It is interesting to note, however, that both surveys estimate the contribution from domestic sources in the UK to be greater than in most EC countries, and the EC as a whole. This difference may be attributable to the low level of tertiary treatment of wastewater in the UK [Morse, Lester and Perry, 1993].

Table 2.1: Estimates of Phosphorus Inputs (%) to Surface Waters in Selected EC Countries and in Total EC

Adapted from Morse, Lester and Perry [1993]

Source	UK	Holland	France	Italy	Total EC
Human Waste	24	23	18	35	24
Detergents	19	3	15	2	10
Livestock	29	57	31	26	34
Fertiliser	14	9	19	18	16
Industry	8	5	6	8	7
Background	6	3	11	11	9
Total Load (000t P)	82	24	106	56	610

Nutrient inputs to water bodies are often categorised as being either from point or non-point sources. Point sources are those which emanate *via* a discrete discharge. They include wastewater from sewage treatment and industrial processes, leakages from animal waste and silage storage, and sewered urban runoff, especially after heavy rainfall, when storm sewage overflows operate [Loehr, Ryding and Sonzogni, 1989]. Non-point sources are, by definition, diffuse in origin. They include atmospheric deposition, runoff containing agricultural fertilisers, from both organic and inorganic sources, and unsewered urban runoff.

Such distinctions are often useful in assessing strategies for nutrient reduction programmes, since point sources are generally easier to identify, quantify and control because of the discrete nature of their entry into aquatic ecosystems. In this study, however, sources of phosphorus will be categorised as those which are associated with the discharge of domestic and industrial wastewater and those which are attributable to agricultural production. The means by which phosphorus exports to eutrophicated water bodies may be reduced will also be considered in these terms.

2.1 Domestic and Industrial Sources of Phosphorus

Domestic and industrial sources of phosphorus include discharges from sewage treatment works (STWs) and sewered urban runoff, which may include storm sewage overflow [Loehr, Ryding and Sonzogni, 1989]. Substantial reductions in the phosphorus content of effluent from STWs may be achieved through the application of tertiary (or advanced) treatment. The costs involved in such technology are, in the UK at least, thought to be prohibitive for smaller STWs. In such cases, an alternative option may be adopted, in which effluent is routed through constructed wetlands prior to discharge into receiving waters. A further method of reducing phosphorus loadings from domestic sources, which has been adopted in parts of North America and Western Europe, is by the use of phosphate-free detergents. In certain cases the best practicable option may be the diversion of all phosphorus-rich effluents away from eutrophicated water bodies. With respect to urban runoff or stormwater, there exists a range of alternative management strategies which not only reduce the incidence of flooding but also improve the quality of water discharged. These strategies include various means of allowing infiltration of rainwater, of controlling flows entering the drainage system, and of attenuating currents within watercourses or sewers.

2.1.1 Tertiary Treatment of Wastewater

The wastewater which arrives at STWs includes domestic sewage, and some types of industrial and agricultural effluents [Wilson and Jones, 1994]. In the UK, the majority (78%) of wastewater produced by such sources receives secondary treatment (Table 2.2). Relatively high proportions of wastewater remain untreated or receive only primary treatment (13% and 8% respectively), while a negligible amount (1%) is passed through tertiary processes.

The main stages in the treatment of wastewater are outlined in Fig.2.1. The rates of phosphorus removal of each of these levels are shown in Table 2.2. In some coastal areas untreated or partially treated sewage is discharged directly into the sea [Wilson and Jones, 1994]. Primary treatment consists of the extraction of gross solids, such as rags, paper and plastic, through screening. Grit and sand are removed by passing the wastewater through a long channel, in which the heavier particles sink. During primary sedimentation, solid matter settles out and is removed as sludge. The processes of digestion and/or de-watering is applied to the sludge produced at this, and at later, stages. Anaerobic digestion produces methane and stable solids. De-watered or digested sludge may be disposed of to land or dumped at sea. The liquid part of the wastewater subjected to primary settlement may be discharged to watercourses or to sea, or, more commonly, it is given secondary treatment.

Table 2.2: Incidence and Average Phosphorus Removal Rates of Various Sewage Treatment Levels in the UK

Adapted from Wilson and Jones [1994]

Treatment	Incidence %	Sludge % P Removed	Discharge % P Remaining
None	13	0	100
Primary	8	15	85
Secondary	78	30	70
Tertiary	1	90	10

The main objective of secondary treatment is to lower the levels of such variables as total suspended solids, biochemical oxygen demand, and pathogenic bacteria [Welch, 1992]. This is achieved through two main methods, biological filtration, and the activated sludge process, both of which utilise bacterial action to decompose organic matter [Wilson and Jones, 1994]. Apart from ammoniacal nitrogen, nutrient concentrations are not normally targeted, but reductions do occur at a rate of 15% to 40% of influent phosphorus [Yeoman *et al.*, 1988].

Tertiary treatment of wastewater may be directed at the reduction of nutrient levels, and a phosphorus removal rate of 90% or more may be achieved [Wilson and Jones, 1994]. This may be accomplished through either chemical or enhanced biological treatment, or a combination of both [Yeoman *et al.*, 1988]. Chemical treatment involves the addition of calcium, iron or aluminium salts in order to achieve phosphorus precipitation. Biological removal relies upon enhanced phosphorus uptake by bacteria, in excess of normal metabolic requirements, within the activated sludge process used for secondary treatment.

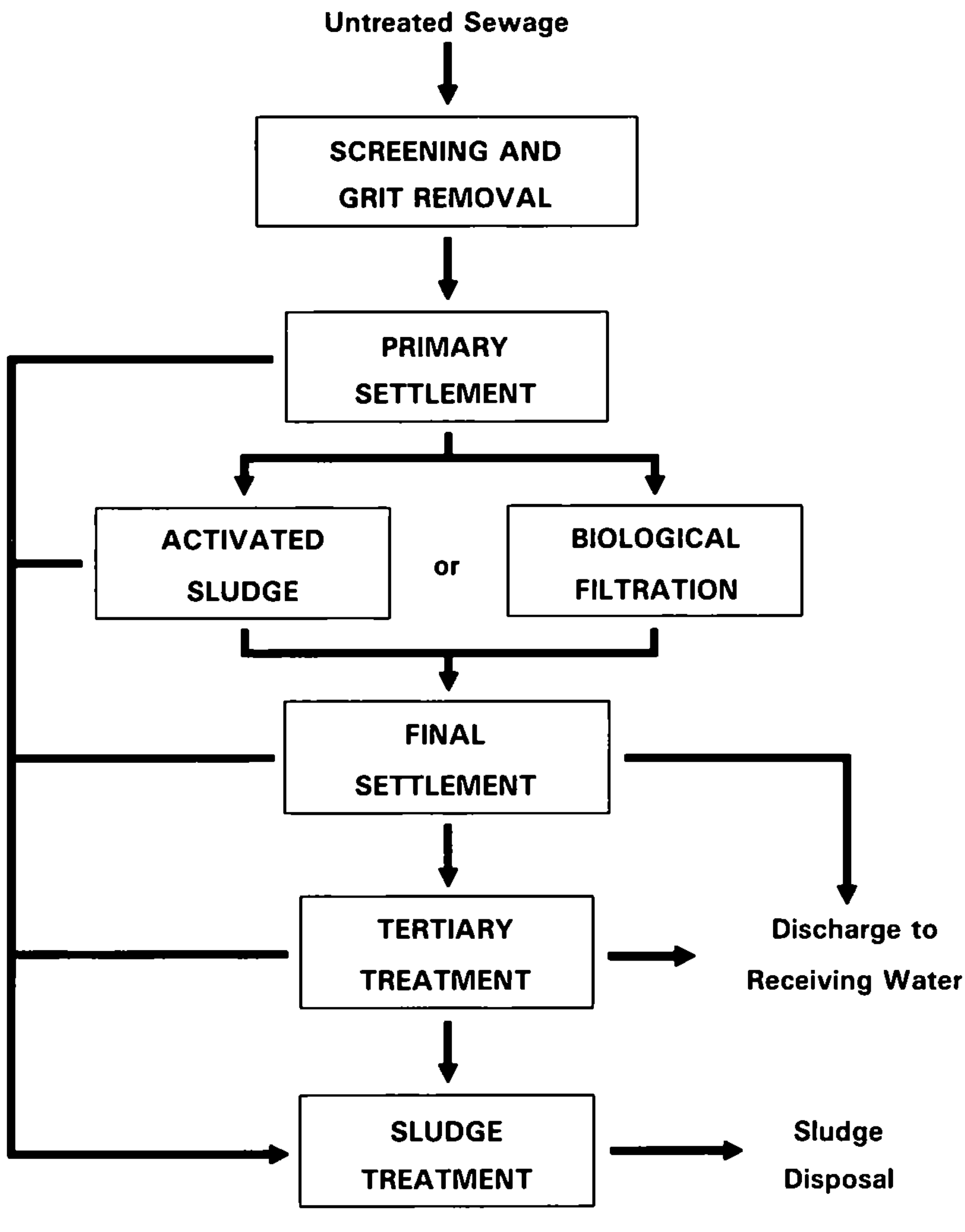

Fig.2.1: Treatment Routes in STWs at Primary, Secondary and Tertiary Level

Adapted from Wilson and Jones [1994]

Chemical phosphorus removal, usually with ferric sulphate as the precipitant, is the method most commonly employed in Britain [Wilson and Jones, 1994]. The main advantages of using ferric sulphate are the low cost, and the de-watering properties of the sludge produced [Yeoman *et al.*, 1988]. However, ferric phosphate, the precipitate, is relatively inaccessible to plants, which limits utilisation of the sludge as fertiliser. Additionally, iron is not the most efficient precipitant and may cause corrosion and staining. Chemical dosing levels are often controlled by time-clocks or flow rates, but both these methods may lead to overdosing, causing discoloration of effluent and increased operational costs [Thomas and Slaughter, 1992]. Consequently, larger STWs discharging into the Rivers Ant and Bure now utilise the redox potential of the secondary effluent to control dosing levels, producing cost and performance benefits. A disadvantage of chemical methods in general is the increased volume of sludge produced, by between 15% and 35%, which may exacerbate disposal problems [Yeoman *et al.*, 1988; Thomas and Slaughter, 1992].

Enhanced biological methods generally produce less sludge, of a quality more suitable for use as fertiliser [Yeoman *et al.*, 1988]. The rate of phosphorus removal is often higher, but loss of efficiency may occur with changes in operating conditions, such as the ratio between chemical oxygen demand and phosphorus concentration, such that it may be necessary to use chemical processes as a back-up. Other problems include the uncontrolled growth of bacteria, which may lead to bulking of the sludge, and the high degree of operator skill required by the process. One advantage of advanced biological treatment is the saving on operational costs (10% to 20%) of chemical precipitants. Additionally, initial investment costs may be reduced by retrofitting existing activated sludge installations, so saving on extensive construction work.

The sludge produced by all systems of sewage treatment must be disposed of in one way or another. At present, 53% of sewage sludge in the UK is used as agricultural fertiliser, 24% is dumped at sea, 16% is sent to landfill sites, and 7% is incinerated [Wilson and Jones, 1994]. The option of dumping at sea will cease by 1998, under the EC Directive on Urban Waste Water Treatment (Section 4.4.3.2), and alternative methods of disposal are being investigated. The government's view on the subject of sludge disposal subsequent to the implementation of this Directive is that there is no single right solution, but that incineration is likely to play an increasing role [Department of the

Environment, 1991b]. Any process which tends to increase the volume of sludge produced, will, inevitably, exacerbate the problem of disposal.

In terms of recycling the nutrient content of sewage sludge, its use as fertiliser is the most appropriate. However, the capacity of the farming industry to accommodate larger quantities of sewage sludge is debatable, since the disposal of animal waste is often a problem [North and Bell, 1990]. One solution, suggested by Wilson and Jones [1994], may be the installation of tertiary treatment systems which recover phosphate in a form which may be re-used in the production of artificial fertilisers, animal feed and industrial-grade phosphates. For instance, a magnetic water treatment system uses lime or metal salts to precipitate phosphates and other pollutants, such as arsenic, copper and cadmium. The precipitates become attached to a magnetic carrier material, magnetite (Fe_3O_4), which enables them to be separated by a magnet. Another system, the Crystalactor, is a fluidised-bed reactor which produces water-free calcium phosphate pellets. The costs of both systems are said to be comparable with other methods.

In March 1991, a report prepared by consultants on behalf of the Department of the Environment, entitled "Pollutants in Cleaning Agents", concluded that any eutrophication problems should be dealt with by phosphate stripping at STWs [Technical Committee on Detergents and the Environment, 1992]. In the light of the EC Directive on Urban Wastewater Treatment (Section 4.4.3.2), the UK will be obliged to implement phosphate stripping capability at STWs serving over 10,000 population equivalent, and which discharge into designated 'sensitive' areas. Since it is estimated that, at present, only 1% of sewage in the UK receives such tertiary treatment (Table 2.2) [Wilson and Jones, 1994], the capital investment involved in meeting the requirements of this Directive will be great. The most cost-effective and environmentally sensitive methods of phosphate stripping will, therefore, need to be investigated.

2.1.2 Constructed Wetlands for Tertiary Treatment

Constructed wetlands consist of soil or gravel filled beds, in which common reeds (*Phragmites australis* (Cav.) Trin. ex Steudel) are planted [Cooper, Hobson and Findlater, 1990]. They may be used for complete treatment of sewage after primary settlement. More commonly, however, constructed wetlands are employed to 'polish' wastewater effluents [Green and Upton, 1994] and it is this function which will be considered here. The advantages of using constructed wetlands for tertiary treatment, rather than the methods outlined above, are low capital and maintenance costs, simple construction, and an ability to withstand a wide range of operating conditions [Cooper, Hobson and Findlater, 1990].

Macrophytes perform a range of functions within constructed wetlands [Brix, 1994]. Various physical effects are the most important. These include stabilising the surface of the bed, providing conditions for physical filtration, and increasing the surface area available for microbial growth. Other important functions include oxygenation of the rhizosphere, which aids aerobic degradation of organic matter and the process of nitrification. The aesthetic and wildlife value of macrophytes may also be significant, especially in areas devoid of such habitat.

In the UK, a number of reed bed systems have been constructed, most of which are specifically designed to provide cost-effective tertiary treatment at smaller STWs [Green and Upton, 1994]. The capital costs of constructed wetlands are favourable when compared to other methods such as modular sandfilters, up to a design load of 10,000 population equivalent. Performance data from several constructed wetlands indicate that they are highly effective in reducing total suspended solids (TSS), biochemical oxygen demand (BOD) and ammoniacal nitrogen concentrations. The reed bed treatment does not, however, appear to reduce phosphorus levels substantially. Results from five sites demonstrate that, on average, total phosphorus is reduced by 21%, from 9.5 to 7.8mg/l. Orthophosphate levels are lowered by an average of 15%, from 8.9 to 7.8mg/l. These performance levels are enhanced by the inclusion of comparatively newly constructed wetlands, which seem to possess greater capacity for precipitation of phosphorus. The older systems monitored achieve only 3% to 14% total phosphorus and 0% to 11% orthophosphate removal rates.

The use of constructed wetlands for tertiary treatment seems to be gaining acceptance by the water companies, which are seeking cost-effective ways of meeting more stringent discharge consents on TSS and BOD levels at smaller STWs. In terms of phosphorus concentrations, however, this method does not appear to achieve sufficient reductions, especially over time, to affect significantly the loadings on eutrophicated water bodies.

2.1.3 The Use of Non-Phosphate Detergents

It has been estimated that detergents contribute about 19% of phosphorus exported to surface waters in the UK [Morse, Lester and Perry, 1993]. The elimination of phosphorus from detergents may, therefore, provide reductions in loadings from STWs. Sodium tripolyphosphate (STPP), or pentasodium triphosphate ($Na_5P_3O_{10}$), is added to detergents as a 'builder', which performs a number of functions [Wilson and Jones, 1994]. Its most important role is in softening the water by sequestering calcium and magnesium ions. STPP also maintains pH at required levels and prevents redeposition of dirt particles on clothes. Essentially, STPP allows surfactants in detergents to operate at maximum efficiency, thereby reducing the amount of powder needed.

As a result of concern about eutrophication, and consequent legislation in several countries, manufacturers have gradually replaced STPP with alternative builder systems [Wilson and Jones, 1994]. The majority of domestic detergents in Germany, Italy, Switzerland, Austria, Norway and the Netherlands, and a significant proportion in France, Belgium, Sweden and Denmark, no longer contain STPP (Fig.2.2).

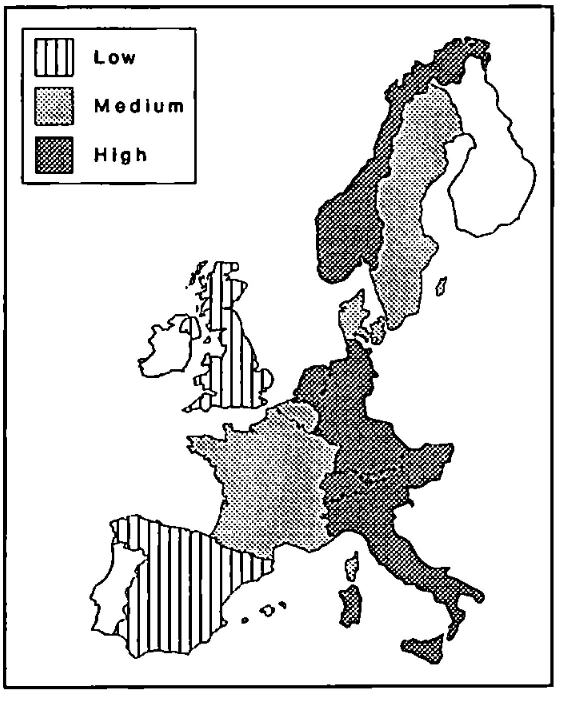

Fig.2.2: The Use of Non-Phosphate Detergents in Western Europe

Adapted from Coffey and Gudowicz [1990]
Updated from Wilson and Jones [1993]

In the USA, initiatives to curb the use of phosphate in detergents began during the early to mid 1970s [Jones and Hubbard, 1986]. By the mid 1980s, phosphate detergent bans were implemented in several states, particularly those around the Great Lakes. In Asia, Japan has enacted a full phosphate ban, and Korea implemented a voluntary exclusion as a result of strong consumer demands [Coffey and Gudowicz, 1990]. In the UK, the process of replacement has taken much longer, and it is only during the last few years that phosphate has steadily lost its market share [Wilson and Jones, 1994].

Several alternative builder systems have been developed, although some, such as EDTA (ethylenediaminetetraacetic acid: $(HO_2CCH_2)_2NCH_2CH_2N(CH_2CO_2H)_2$) and NTA (nitrilotriacetic acid: $N(CH_2CO_2H)_3$), have aroused reservations about their acceptability on environmental and health grounds [Wilson and Jones, 1994]. Through their ability to maintain essential metal ions in solution, these synthetic chelating agents possess a potential for great impact on metal solubility and transport in surface waters [Daum and Newland, 1982]. The most widely-accepted alternative to STPP is zeolite A, which is derived from aluminium oxide (Al_2O_3) and which is apparently inert [Wilson and Jones, 1994]. However, zeolite A does not fulfil all the functions of STPP and must be used in conjunction with co-builders, usually polycarboxylic acids (PCAs), which are petroleum-based compounds.

As mentioned above, detergent phosphate bans were implemented in the USA during the mid 1980s. Such bans, which often provoked controversy, were introduced within the catchments of lakes affected by eutrophication, in attempts to improve water quality. In a number of States in which legislation was introduced, limnological investigations were carried out to assess the impact of bans. These studies enabled analysis of whether detergent phosphate bans actually reduce external phosphorus loads sufficiently to improve water quality. In a summary of research covering five States in which bans were implemented, Maki, Porcella and Wendt [1984] came to the conclusion that the elimination of detergent phosphate did not measurably improve lake water quality. Most studies reported that any changes were too small to observe, compared to natural variations. The reason given for the lack of improvement was that the reductions in phosphorus loadings caused by the bans were generally insufficient to induce a change in trophic status.

In response to the conclusions drawn by Maki, Porcella and Wendt [1984], and subsequent criticism of detergent phosphate bans, Lee and Jones [1986] reported the development of a general relationship which could be used to evaluate the potential benefits of such bans. It was based on the finding that a 20% reduction in phosphorus loading must be achieved in order to produce a discernible effect on water quality, independent of the trophic status of a water body. Lee and Jones concluded that eutrophication-related water quality could rarely be improved by a ban, and only where a high proportion of the phosphorus entering a water body is derived from detergents.

A more recent contribution to the continuing debate over the use of phosphates in detergents is provided in a report by Wilson and Jones [1994]. These authors compared life cycle assessments (LCAs) of STPP and of zeolite-PCA. The relatively new, and still developing, discipline of LCA evaluates the environmental impact of products and processes from 'cradle to grave' (that is, from the extraction of raw materials through processing, manufacture and use to disposal). The conclusions drawn were then applied to the problem of eutrophication in a series of practical recommendations. Their report therefore brings a wider perspective to the phosphate detergent question.

For each stage in the production and manufacture of STPP and zeolite-PCA builders, polluting emissions to the atmosphere and to receiving waters, and solid waste to land, were listed [Wilson and Jones, 1994]. Each pollutant was then ascribed environmental impact points awarded by a panel of scientists. Tests were carried out to determine an equivalent performance ratio for the builders. This demonstrated that 1 kg of zeolite-PCA was needed to give a wash performance equivalent to that produced by 0.7 kg of STPP. A final weighted score for environmental impact was achieved by applying this ratio to the points for pollutants.

The results of this part of the study indicated that there are no appreciable differences in environmental impact between the two builders when assessed over their full life cycle [Wilson and Jones, 1994]. The authors then carried the debate further by arguing that STPP was "the builder of choice from an environmental perspective" for several reasons. As outlined above, experience in the USA and elsewhere indicates that the removal of STPP from detergent formulations does little to alleviate the problem of eutrophication. Additionally, STPP, unlike zeolite, may be recovered from wastewater and recycled. Technology is available which is able to recover phosphates in the form of calcium phosphate pellets, which can be used in the production of detergents and animal feed. Furthermore, the removal of phosphorus from *all* sources in wastewater, rather than simply that derived from detergents, would be far more likely to produce an improvement in the quality of receiving rivers and lakes.

2.1.4 Diversion of Sewage Effluent

Diversion of sewage effluent entails the construction of pipelines to reroute wastewater out of a lake's drainage basin [Cooke *et al.*, 1993]. The effluent may be diverted downstream of the lake, to another catchment, or directly to sea. Such an option may seem attractive, but there are several drawbacks:

(a) The diverted effluent may have a deleterious effect on recipient waters [Dunst *et al.*, 1974], so that the problem of enrichment is merely shifted elsewhere. Diversion to sea, rather than rivers, is more acceptable, since the dilution factor is much higher. This option is only feasible, however, for inland waters situated close to the coast.

(b) Depending on the distance and logistics involved, the cost of diversion can be high [Cooke *et al.*, 1993]. The benefits of diversion (based on estimates of the proportion of phosphorus contributed by sewage effluent) must, therefore, be weighed against the costs.

(c) As a result of diversion, the flushing rate of a lake may be reduced significantly, increasing the length of time nutrients are retained [Moss, Stansfield and Irvine, 1990]. This may produce undesirable changes in the phytoplankton, despite a reduction in phosphorus load.

For some lakes and reservoirs, however, diversion of sewage effluent may be a viable option, particularly for those lying close to the sea. In England, diversion has been implemented at Alderfen Broad, where phosphorus-rich stream water was re-routed around the lake, and at Barton Broad, where effluent from a STW was diverted to sea [Moss, Irvine and Stansfield, 1988].

The effects of sewage effluent diversion alone may be assessed for Alderfen Broad, where this measure has not yet been supplemented by other phosphorus reduction methods, as at Barton Broad [Moss, Irvine and Stansfield, 1988]. After diversion in 1979, Alderfen Broad initially experienced reduced total phosphorus concentrations and phytoplankton populations, clear water and the establishment of submerged macrophytes [Perrow, Moss and Stansfield, 1994]. However, diversion also led to a decrease in water level and flushing rate, which seems to have encouraged increases in internal phosphorus loading and concentration, so that the initial favourable effects were not maintained.

The best known and well-documented case of water quality improvement following diversion of treated sewage is that of Lake Washington, USA [Cooke *et al.*, 1993]. The main reason for the rapid recovery of this lake was its fast rate of water renewal (0.4 per year) in relation to its depth (mean of 37m). Additionally, the large hypolimnion had not reached an anoxic state, so that its internal loading, after diversion, was insignificant. The diverted water contained 88% of the lake's external phosphorus load, but, in terms of volume of inflow, was not sufficient to reduce significantly the flushing rate.

These cases serve to illustrate that, for diversion alone to elicit a marked improvement in water quality on a long-term basis, the effluent should represent a significant proportion of external phosphorus load, but not of inflow volume [Moss, 1988]. This maxim seems to apply particularly in the case of shallow lakes in which internal loadings are potentially high.

2.1.5 Stormwater Management

The hydrological consequences of urbanisation are numerous and diverse (Fig.2.3). Increase in the area of impervious surfaces leads to reduced infiltration, which lowers the rate of soil moisture recharge, to the extent that dry weather stream flow and groundwater yield are diminished [Tourbier, 1994]. It has been estimated that under natural ground cover about 50% of rainfall volume drains into the soil, whereas in urban areas with 75% to 100% paved surfaces this is reduced to about 15%. Conversely, impervious surfaces increase the volume of surface water runoff, from about 10% under natural ground cover to about 55% in paved areas. Pollution associated with runoff presents an additional consequence. Paved surfaces collect nutrient-rich dust, and vehicle related trace metals, which are likely to be washed away by high levels of runoff.

Urban sewerage systems are designed to remove surface water runoff as quickly and as efficiently as possible [Hall, 1984]. This function is usually served by combined sewer systems, in which both foul sewage and stormwater are accommodated within the same pipes. During times of dry weather the sewage is treated before discharge. Following a storm event overflow structures limit the volume of runoff which is carried to STWs and the excess is discharged untreated. Such combined sewer overflows (CSOs) are usually designed to operate when the current exceeds six times dry weather flow. Separate stormwater sewers, introduced in order to avoid the pollution problems associated with CSOs, are designed to discharge directly into local water courses.

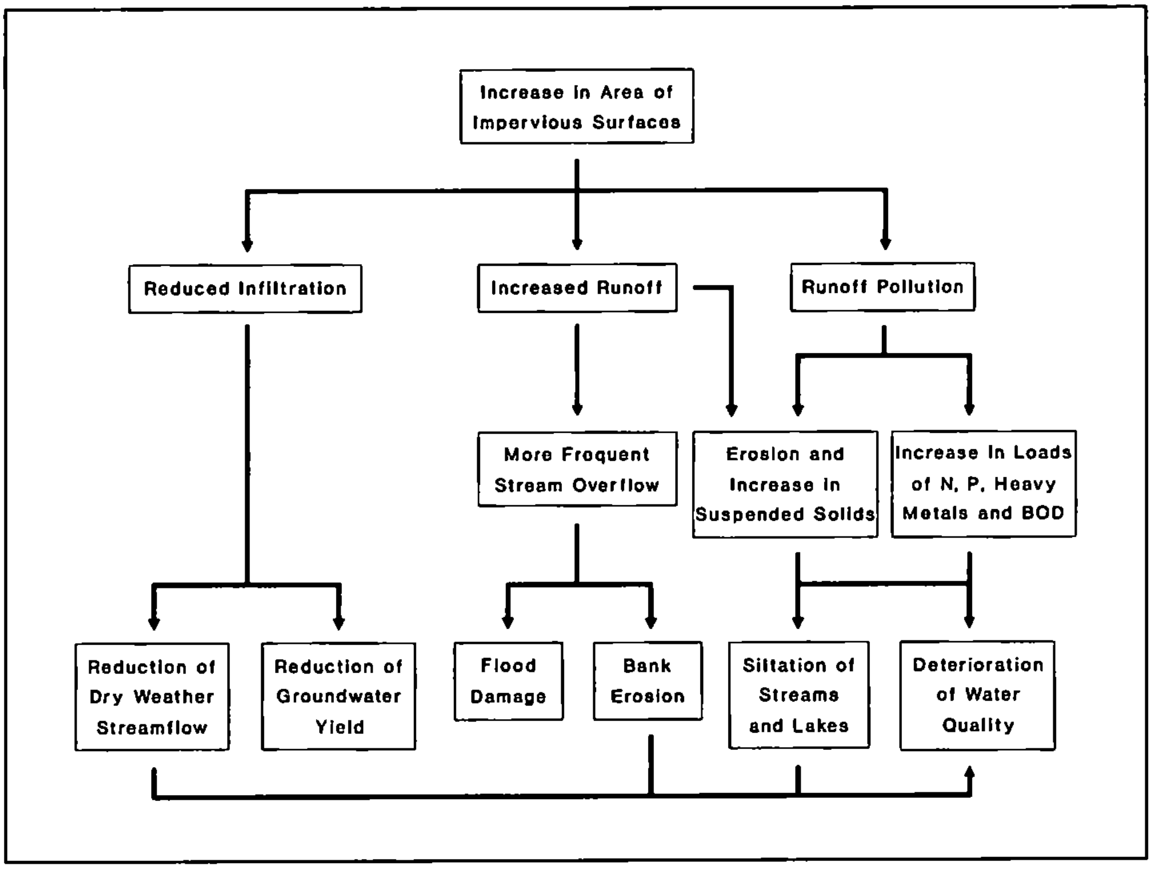

Fig.2.3: The Interaction of Stormwater Problems

Adapted from Tourbier [1994]

With either system the relatively rapid transport of large volumes of water to rivers may create downstream flooding, especially where the amount of overbank storage has been reduced by development. Consequently, capital expenditure is required to construct flood alleviation channels, embankments or storage ponds.

In the UK, 70% of urban drainage systems utilise CSOs [House *et al.*, 1993], with some 21,000 structures operating in England and Wales [Clifforde and Johnson, 1993]. The pollution problems associated with CSOs are well recognised, to the extent that substantial lengths of water courses are deemed to be seriously affected by overflow discharges. Storm sewage consists of a mixture of domestic waste and industrial effluent, diluted by surface runoff [Balmforth, 1990], which itself is often highly polluted with elevated levels of suspended solids and nutrients (Table 2.3). As the storm discharge advances down the sewer it drives the dry weather sewage before it. Since urban catchments respond much more quickly to rainfall events, CSOs often operate before flow in the water course has risen. These factors combine to produce a sharp rise in the concentrations of pollutants in the receiving water during the early part of a storm.

Table 2.3: Mean Pollutant Concentrations for Separate Stormwater Sewers and Combined Sewer Overflows in the UK

Adapted from Ellis [1989]

Pollutant	**Event Mean Concentration (mg/l)**	
	Separate Stormwater Sewers	**Combined Sewer Overflows**
Suspended Solids	190	425
Biochemical Oxygen Demand	11	90
Chemical Oxygen Demand	85	380
Ammoniacal Nitrogen	1.45	6.0
Total Phosphorus	0.34	10.0
Total Lead	0.21	0.25
Total Zinc	0.30	0.87

The 'first flush' of pollution may be succeeded by a delayed effect caused by degradation of adsorbed soluble, colloidal and fine particulate organic matter, and associated depression in dissolved oxygen levels [Hvitved-Jacobsen, 1982; Rees and White, 1993]. In some rivers receiving CSOs, deterioration of water quality may be the result of high flow *per se*, owing to the release of pollutants from resuspended sediments [Rees and White, 1993].

Stormwater discharges, whether from separate systems or from CSOs, vary in magnitude, duration and frequency [House *et al.*, 1993]. From an ecological perspective, such discharges produce not only chemical toxicity, but also habitat instability. The composition of aquatic communities will shift towards those species capable of withstanding continuous changes in habitat. Such episodic events have been shown to produce short- and medium-term effects on phytoplankton and zooplankton communities [Gast, Suykerbuyk and Roijackers, 1990], and long-term effects on sessile diatoms and macroinvertebrates [Willemsen *et al.*, 1990].

In terms of eutrophication, stormwater discharges represent intermittent influxes of high concentrations of nutrients. As a proportion of total annual nutrient loading on a water body, the contribution of stormwater is generally low [House *et al.*, 1993]. Such loading may, however, lead to significant, short-term disturbances of planktonic communities, medium-term effects such as changes in benthic communities, and long-term accumulation of nutrients and organic matter in the sediments (Fig.2.4).

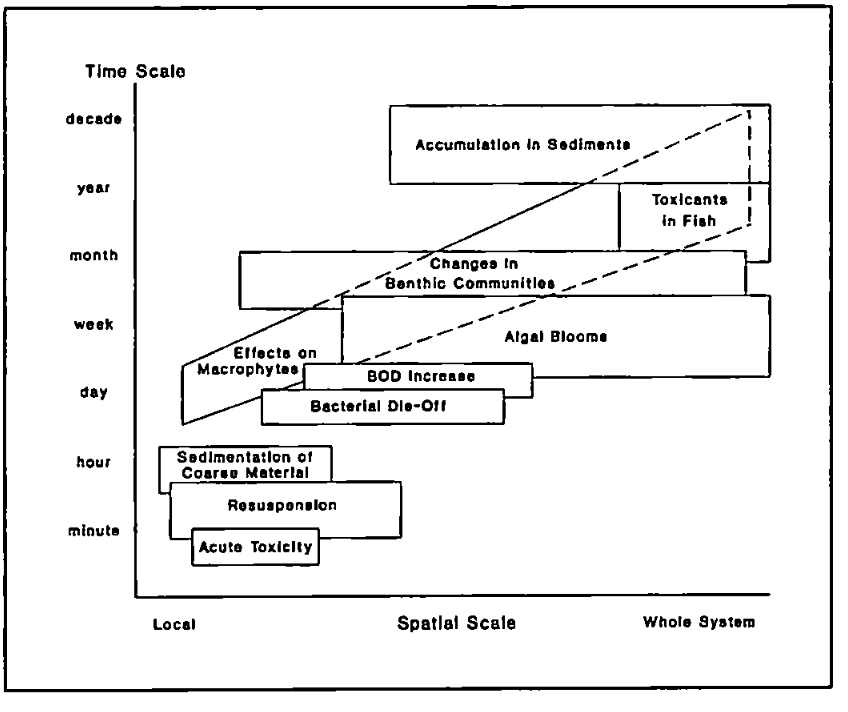

Fig.2.4: Temporal and Spatial Impacts of Stormwater on Receiving Waters

Adapted from House *et al.* [1993]

The management of stormwater, in order to minimise pollution and flooding problems, from both separate and CSO systems, may be achieved in three ways [Beale, 1992]:

(a) increase the capacity of the drainage system

(b) reduce flows entering the drainage system

(c) attenuate flows within the drainage system

The option of enlarging the drainage system capacity is one which has, until recently, seemed an acceptable way of dealing with increasing stormwater loads. However, high economic costs are involved in upgrading existing sewers, and it has become unacceptable merely to shift the problem downstream. Moreover, there exists a wide range of methods for diminishing the surge impact of stormwater, either through reducing flows before they enter the drainage system or by attenuating them within watercourses or sewers (Fig.2.5). The utilisation of one or more of these methods may serve to disperse or to avoid the impacts of large volumes of stormwater, thereby reducing the risks of flooding and the operation of CSOs.

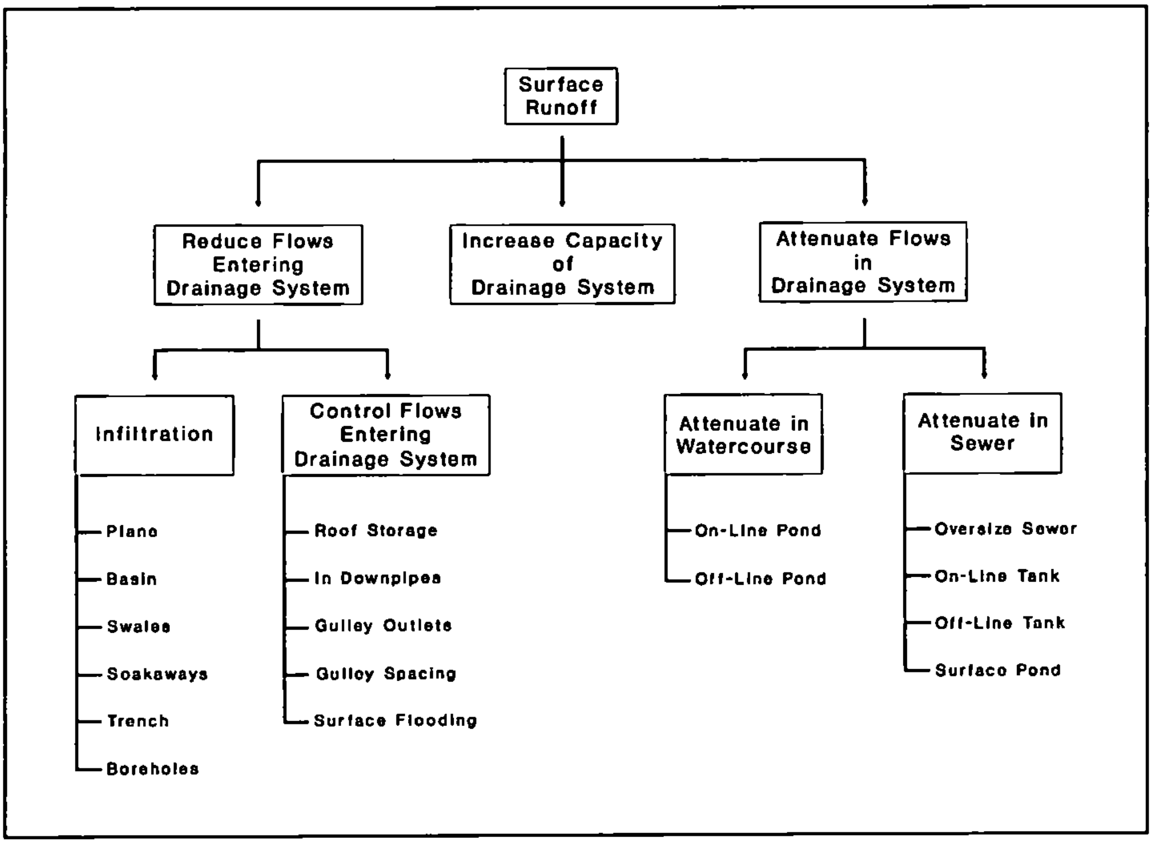

Fig.2.5: Options for Urban Stormwater Management

Adapted from Beale [1992]

The method of stormwater management which addresses the problems of flooding and pollution in a fundamental manner is that of infiltration. Conveyance of stormwater to natural subsurface drainage systems may be achieved through the use of permeable planes, such as pavements, through seepage within basins, swales, soakaways and trenches, or by direct transfer to aquifers *via* boreholes [Beale, 1992]. Infiltration not only assists in avoiding surges of stormwater and therefore discharges from CSOs, but also in replenishing groundwater, maintaining stream base flow and improving water quality [Ferguson, 1990]. Further benefits of this practice are a significant decrease in the volume of water entering STWs and a reduction in total phosphorus in the discharge [Geldof, Jacobsen and Fujita, 1994]. Added to this, the costs of infiltration trenches are considerably less than those of the detention systems described below.

There is, however, the risk of soil and groundwater pollution associated with infiltration of stormwater from certain sources [Mikkelsen *et al.*, 1994]. In order to avoid contamination, infiltration systems which retain the pollutants need to be employed. Designs may include the use of appropriate sub-base materials or vegetation which is capable of uptake and storage of pollutants. Stormwater infiltration has been practised in some urban areas of the USA for over fifty years [Ferguson, 1991], in Japan for more than a decade [Fujita, 1993] and, more recently, in Germany [Stotz and Krauth, 1994]. A number of infiltration methods are generally used in combination on a site-specific basis. In the UK, research is being carried out on the construction of permeable pavements, but work is still at an experimental stage [Pratt, Mantle and Schofield, 1989].

Management of stormwater may also employ methods of controlling the flow prior to its entry into the main drainage system (Fig.2.5). These include detention in roof and downpipe tanks, restricting the runoff rate from drainage gulleys and providing above-ground storage when the capacity is exceeded, and reducing the spacing between gulleys [Beale, 1992]. Such methods have not, however, been utilised to any great extent in the UK.

A method of stormwater management often used in new developments in the UK is the attenuation of flows in watercourses, which involves the construction of detention or balancing ponds (Fig.2.5). Such facilities act as temporary storage for stormwater, which is allowed to flow into the drainage system at a reduced rate [Ferguson, 1991]. Some detention ponds may incorporate a permeable lining, so that infiltration also occurs. The ponds may be permanently inundated and act as wetland features for amenity and wildlife purposes, or they may contain standing water only following storm events. The scale of such constructions ranges from small ponds serving housing estates, to large flood storage reservoirs, such as Willen Lake, Milton Keynes, which may accommodate a wide range of water-based leisure activities [Hall, Hockin and Ellis, 1993]. Detention ponds have traditionally been used purely for flood control, but an increasing number are being designed as a means of improving water quality through sedimentation, adsorption and biological uptake [Toet, Hvitved-Jacobsen and Yousef, 1990; Breen, Mag and Seymour, 1994; Startin and Lansdown, 1994]. Fig.2.6 illustrates the layout of a detention pond which incorporates a vegetative treatment system.

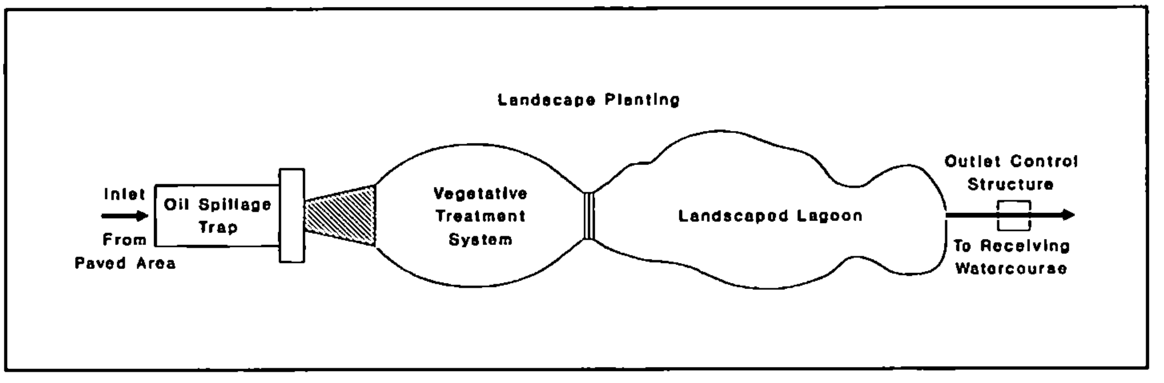

Fig.2.6: Detention Pond with Vegetative Treatment System

Adapted from Startin and Lansdown [1994]

The attenuation of stormwater within the sewerage system may also be used to reduce peak flows (Fig.2.5). These methods again rely on excess flow volumes being stored and then released at a controlled rate [Beale, 1992]. Such storage may be provided by oversize sewers, or on-line or off-line tanks. Flow controls for oversize sewers and on-line tanks may consist of a range of devices, the most effective of which appear to be vortex regulators [Parsian and Butler, 1993]. These are relatively recent innovations, but an estimated 1500 are in service in the UK. The advantages of vortex regulators are that they are less prone to blocking and require lower storage volume (up to 30% less). Off-line tanks serve as storage for flows diverted when the downstream system has reached capacity, requiring less volume than on-line tanks [Beale, 1992]. Both forms of sewer storage tanks do, however, require emergency overflow systems. In circumstances where stormwater can be isolated prior to entering the sewerage system, the use of surface storage ponds may be implemented, with flows from the drains entering and leaving *via* screened pipes. This method has been used successfully for motorway drainage. Such ponds may also be transformed into constructed wetlands, planted with species which are capable of uptake and storage of petroleum hydrocarbons, lead and zinc [Ellis *et al.*, 1994].

Although the problems associated with urban drainage are several and varied, the pollution aspects necessarily include transport of excessive nutrients to receiving waters. In the context of this study, methods of stormwater management which reduce substantially the level of sewage-related pollution, and which also benefit the ecology of aquatic ecosystems are preferable. The information presented above suggests that infiltration methods perform these functions better than detention systems, not least because they deal directly with the fundamental problem of urban drainage, rather than applying control technologies to the symptoms.

2.2 Agricultural Sources of Phosphorus

The intensification of agriculture after the second World War was intended to provide food security and improve self-sufficiency for the UK [Harvey, 1991]. The realisation of these objectives entailed, however, not only substantial increases in fertiliser application rates [Chalmers, Kershaw and Leech, 1990], but also the adoption of practices which exacerbate soil erosion [Arden-Clarke, 1988]. As mechanical and chemical advances replaced traditional practices, types of terrain once perceived as poor agricultural land were taken into production [Harvey, 1991], thereby increasing the area subjected to intensive cultivation. The effects of agricultural 'improvement' on nature conservation has been considerable, particularly in terms of habitat destruction [NCC, 1990]. The repercussions of fertilisation practices for soil fertility, soil ecology, nutrient cycling, and consequently, nutrient losses, have been equally deleterious [Arden-Clarke, 1988], to the extent that runoff from agricultural land is the major source of nonpoint losses to surface waters [Loehr, Ryding and Sonzogni, 1989].

In order to assess methods for the control of agricultural nonpoint sources, the soil processes involved in phosphorus losses need to be understood. The use of inorganic fertilisers, animal manure and, in some areas, sewage sludge, generally increases the nutrient loads in runoff, compared to that from unfertilised soils. Rate and timing of application are important factors in determining the extent of such losses. However, the nutrient content of the soil, as well as its texture, chemistry, type and physiography, also affect the degree of runoff and its phosphorus content.

The natural phosphorus content of soil ranges from 100 to 2,500 mg/kg, depending on the parent material and texture, as well as soil management factors [Daniel *et al.*, 1994]. The amount of phosphorus in organic and inorganic forms is also influenced by these factors. In most soils, 10% to 50% of the phosphorus is organic, consisting largely of stable fulvic and humic compounds. Mineralisation of the more labile forms usually occurs at the rate of 5% of soil organic phosphorus per year. Inorganic phosphorus consists of iron and aluminium phosphates in acid soils and calcium phosphates in alkaline soils. Less than 10% of inorganic phosphorus is available for plant uptake at any one time, since most forms are highly insoluble. As the labile form in soil solution is depleted by plant uptake, it is slowly replenished by mineralisation of organic phosphorus, desorption of labile phosphates and dissolution of soluble forms of phosphorus applied as fertiliser. Adsorption of phosphorus by soil tends to occur more rapidly than desorption, so that a general decrease in bioavailable phosphorus occurs after fertiliser application.

The loss of soil phosphorus in runoff occurs in dissolved and particulate or sediment-bound forms (Fig.2.7). Dissolved phosphorus is largely composed of orthophosphate, with small amounts of organic forms, so that 90% to 95% of this fraction is bioavailable [Daniel *et al.*, 1994]. As rainfall interacts with a thin layer (< 2cm) of soil, the processes of desorption, dissolution, extraction and mineralization of soil phosphorus facilitate movement of the dissolved fraction in runoff. Rainfall which percolates through the soil profile transfers dissolved phosphorus to subsoils, where most of it is adsorbed, resulting in low concentrations of this fraction in subsurface flow. Sediment phosphorus comprises particulate forms adsorbed by soil particles and organic matter greater in size than $0.45\mu m$. Movement of sediment phosphorus is, therefore, determined by erosion. As a result, phosphorus losses from cultivated land are composed of 75% to 95% sediment phosphorus, whereas runoff from grass and woodland from where there is less erosion, contains largely the dissolved form.

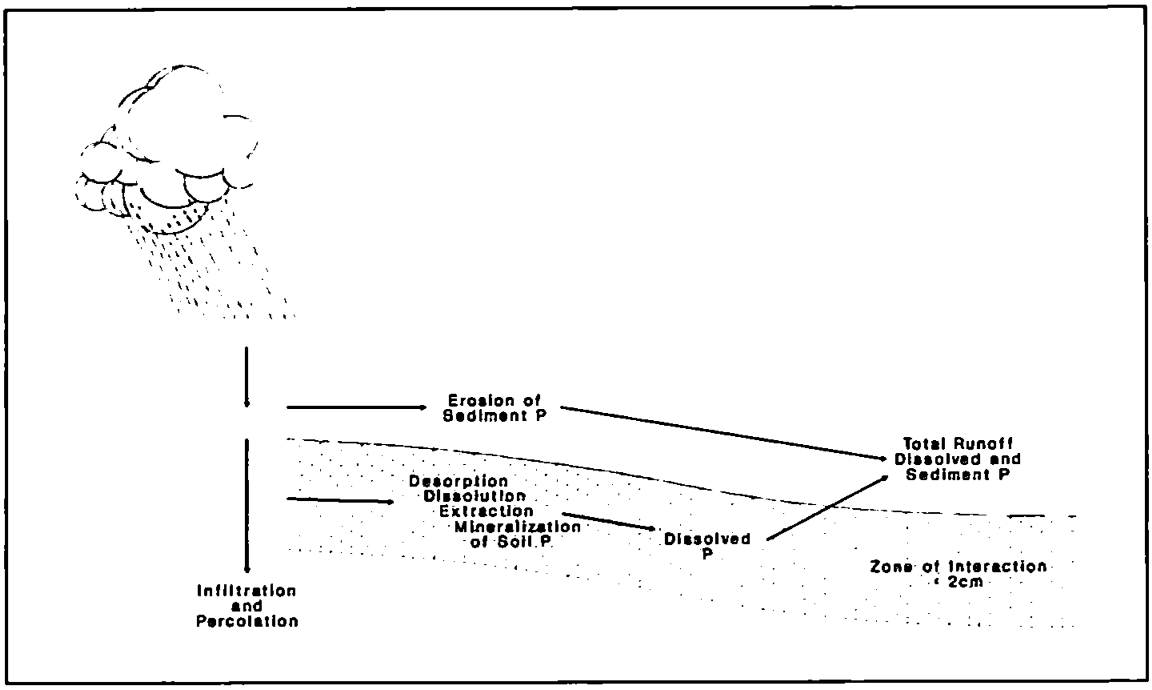

Fig.2.7: The Transport of Phosphorus (P) in Agricultural Runoff

Adapted from Daniel *et al.* [1994]

Given the lack of mobility of phosphorus in most types of soil, only a small proportion is generally lost in subsurface flow [Daniel *et al.*, 1994]. In deep sandy soils, however, subsurface transport can be significant, owing to low surface area or a lack of phosphorus-retaining components [Harris, Wang and Reddy, 1994]. The main mechanisms by which phosphorus is lost from agricultural land is by runoff and erosion [Sharpley and Smith, 1990]. Therefore control of sediment and associated phosphorus transport can effectively reduce phosphorus losses from nonpoint agricultural sources.

Intensive farming systems, especially those concerned with livestock production, are likely to elicit point sources of nutrient-rich effluent. These consist largely of leaks and spillages from the storage and disposal of slurry and silage liquor [NCC, 1991a]. Incidents of pollution from silage arise through poor silo construction and ineffective effluent management systems. Slurry-related losses are often associated with prolonged wet weather and inadequate separation of clean from dirty water, leading to storage overloading. Fig.2.8 shows the percentage of reported pollution incidents attributable to different types of agricultural waste.

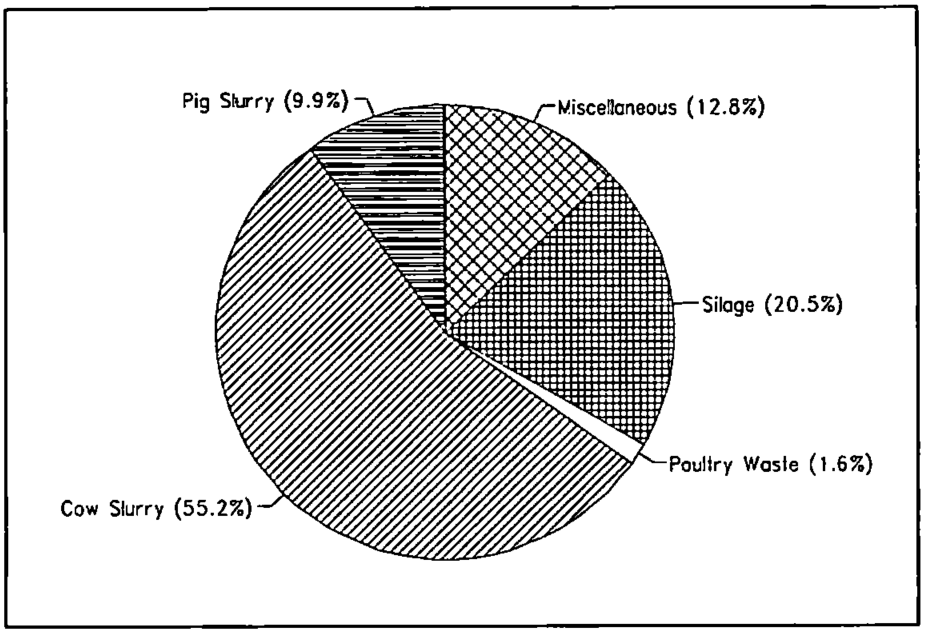

Fig.2.8: Farm Pollution Incidents in England and Wales 1985-1989

Adapted from NRA [1992a]

Acute pollution may cause significant toxic effects, including fish kills [NRA, 1992a]. The biochemical oxygen demand of materials such as slurry and silage effluent is very high, even compared to raw human sewage. Chronic pollution from point sources, which is much more difficult to detect, often produces more subtle, but significant, impacts on aquatic habitats and communities. The application of animal waste to the land may also cause acute or chronic pollution of water courses. Problems occur when too much is applied, where fields are underdrained, or when conditions during or after application are such that waste is lost in runoff.

It is clear that the contribution of agriculture to the eutrophication of surface waters is significant and that nutrient losses can occur at most stages of production. Yet eutrophication represents but one aspect of the way in which intensive farming has led to the deterioration of the countryside in terms of natural habitats and species conservation [NCC, 1991a]. In recent years, the problem of agricultural surpluses within the European Community, caused by the Common Agricultural Policy (CAP), has served as a focus for the debate surrounding modern farming and nature conservation [Potter, 1991]. Two basic strategies may be adopted in order to reduce the level of agricultural production, and so provide external benefits in terms of nature conservation.

First, farmers can be paid to remove some of their land from intensive farming and to manage it in such a manner as to enhance wildlife interests (the land diversion or set aside strategy). Second, farmers can be induced to adopt methods and systems of production which result in an overall reduction in the intensity of farming and the integration of nature conservation with land use (the extensification strategy). Several alternatives to intensive agriculture have been proposed, including modified forms of 'traditional' mixed farming and organic farming. Among conservationists there are supporters of both approaches, although the majority, including the nature conservation agencies, advocate the second. Those who favour land diversion argue that discrete and permanent changes in land use provide better opportunities for planned conservation objectives. The proponents of extensification, on the other hand, claim that what is required is to alter the way in which all land is farmed in order to foster conservation of species, habitats and landscapes in the wider countryside. These two strategies are not, however, mutually exclusive. A dual approach in which special areas are designated, while nature conservation is integrated into the wider countryside, would serve all interests. It is merely the availability of conservation options for EC land diversion schemes which has forced a choice between one strategy and the other.

With respect to the contribution of agriculture to eutrophication, the same debate might be envisaged. Indeed, the means by which nutrient losses from agriculture can be reduced may be categorised along the same lines. One strategy is to accept modern intensive farming, and to reduce its effect on water quality as much as possible. This might simply involve adherence to good agricultural practice, but it may also include a reduction in production levels through land diversion. The latter option would allow the use of agricultural land along water courses for other purposes, which could include sites for nature conservation and/or the establishment of riparian vegetation to act as a buffer for nutrients. Another strategy is one which advocates other, less intensive methods of agricultural production. This alternative attempts to reverse the trends of the last fifty years through the extensification of agriculture, adopting 'traditional' mixed farming or organic farming methods. In practice, reducing agricultural nutrient losses may well entail utilising aspects of both strategies. For the purposes of this study, the approaches discussed will be adherence to good agricultural practice, the use of land diversion schemes for the benefit of water quality, and the extensification of agriculture through the adoption of alternative farming practices.

2.2.1 Adherence to Good Agricultural Practice

In July 1991, the Ministry of Agriculture, Fisheries and Food (MAFF) and the Welsh Office Agricultural Office (WOAD) produced a Code of Good Agricultural Practice for the Protection of Water. It is intended as a practical guide to help farmers avoid causing water pollution [MAFF and WOAD, 1991]. The Code includes advice on safe means of storage and application to the land of slurries, dirty water, solid manures, other organic wastes, silage effluent and fertilisers, as well as guidance on the safe use of fuel oil, sheep dip and pesticides, and on disposal of animal carcasses. A section is devoted to advice on means of reducing nitrate losses whilst maintaining production levels.

The Code recommends that all storage facilities for farm waste should be designed and constructed according to standards laid down by BS 5502: Part 50: 1989 [MAFF and WOAD, 1991]. New or substantially enlarged installations must comply with the Control of Pollution (Silage, Slurry and Agricultural Fuel Oil) Regulations 1991 (Section 4.4.2). The Code outlines the stipulations for storage of each type of agricultural waste, as set out in the Regulations. Stores constructed prior to March 1991 do not have to meet these standards, although the NRA can require improvements to such facilities if it is thought that there is a significant risk of pollution. In order to identify such storage inadequacies, farm campaigns are undertaken by the NRA in catchments where water quality problems are known to exist [NRA, 1992a]. Visits to farms may result in prosecutions, but advice on remedial action and availability of grant aid is also given [Barker, 1991]. The number of farms polluting or at high risk represented 41.5% of the total visited up to 1992, although it needs to be taken into account that problem catchments were targeted initially [NRA, 1992a]. Over time, the majority of stores will be replaced or extended, and will therefore comply with the Regulations. Existing facilities at high risk of pollution will, eventually, be detected by the NRA through farm campaigns. It may, however, take many years for improvements in storage facilities to affect water quality.

The Code also provides guidelines for the application of animal wastes to land, an aspect which is not covered by the Control of Pollution (Silage, Slurry and Agricultural Fuel Oil) Regulations 1991 [MAFF and WOAD, 1991]. Areas of land which should not receive waste at any time are those within 10m of watercourses and 50m of a spring, well or borehole which is used for human consumption. Very high risk areas include fields which are likely to flood within a month of application, are frozen hard, or located next to a watercourse, spring or borehole, under conditions such as severe compaction, waterlogging, or on steep slopes. Organic wastes should not be applied at times when these conditions prevail. High risk areas, such as moderately sloping fields next to a watercourse, where the soil is at field capacity, should not receive more than the recommended amounts of waste. In all other situations, waste needs to be applied with care, and watercourses should be checked frequently during and after spreading.

With respect to inorganic fertilisers, the Code stipulates that bags of solid material should not be stored within 10m of a watercourse and that liquid forms require a suitably designed storage tank, located as far away as possible from drainage systems [MAFF and WOAD, 1991]. The Code states that farmers should take special care when applying any inorganic fertilisers on fields where there is a risk of runoff to surface water, for instance when the field is waterlogged or frozen hard, and that nitrogen fertiliser should not be spread between 1 September and 1 February unless needed by a crop during this time. Farmers are also advised to "avoid applying fertiliser to a watercourse" [*ibid*, p.46]. The section on nitrates contains general information on various practices aimed at reducing leaching. These include avoiding ploughing up permanent grassland for arable production, sowing crops in early autumn, and careful fertiliser application rate and timing. It is stated that the risk of nitrate leaching from organic manures is higher than that from inorganic fertilisers, but the benefits of added soil organic matter through the incorporation of animal wastes are not mentioned.

In response to the Consultation Paper on the Code, the Institution of Water and Environmental Management (IWEM) made numerous suggestions, most of which were incorporated into the final draft [IWEM, 1991b]. Other comments were disregarded, some of which are of relevance to this study. The IWEM considered that, in merely stating that "a suitably qualified person" and "experienced persons" [MAFF and WOAD, 1991, p.12] should be employed for the design and construction of storage facilities, MAFF does not fully treat farming as a business activity. The Institution was of the opinion that in industry an engineer would be specified. The IWEM also questioned the minimum distance of 50m radius from a borehole for the application of organic wastes. It regarded this distance as inadequate, particularly when compared to the 250m radius stipulated by the NRA. In the light of this statement, the 10m minimum distance from watercourses also appears inadequate. Additionally, the advice to "avoid" applying fertiliser to a watercourse seems rather weak and ineffectual.

The Code of Good Agricultural Practice for the Protection of Water appears to be aimed primarily at reducing acute and chronic pollution from the storage of animal waste, and at lowering the level of nitrate leaching from soils. Moves towards these objectives are to be encouraged, but the Code does not mention the process of eutrophication or losses of phosphorus, nor does it provide advice on methods of reducing soil erosion from agricultural land. These aspects are dealt with in The Code of Good Agricultural Practice for the Protection of Soil, published two years later in 1993. The section on soil erosion, though brief, recommends various management strategies, including contour cultivation, early planting of winter crops, introduction of grass into arable rotations, and the use of buffer areas of rough vegetation and hedges to encourage deposition of sediment. It is advised that land which is regularly and severely affected by erosion should be put down to permanent grassland or woodland under one of the various government-supported schemes. The Code also advises on the maintenance of soil organic matter through the addition of manures or by returning crop residues to the land.

With respect to phosphorus losses, the Code states:

> Do not apply more phosphorus (in fertilisers or manures) than the crops in the rotation need. If soil particles with a high phosphorus content are eroded and deposited in rivers, lakes or the sea, the phosphorus content of the water will rise. This may cause eutrophication, making algae grow too fast, which uses up the oxygen in the water and can kill fish. Leaching (washing out) of soluble phosphorus compounds in drainage water can occur if a soil is nearly saturated with phosphorus. This is only likely to occur on sandy soils after many years of applying very large amounts of animal manures. [MAFF and WOAD, 1993, p.10]

The effectiveness of the Code on Water would be improved considerably if it were rewritten with appropriate cross references to the Code on Soil, particularly in relation to phosphorus losses. On the whole, the recommendations contained in these two Codes of Good Agricultural Practice, if adhered to, would probably produce reductions in the nutrient content of water draining agricultural land. The crucial factor in the effectiveness of the Codes appears to be the extent to which farmers are prepared to heed such advice, particularly where protection practices might entail expenditure or loss of income.

2.2.2 Agricultural Land Diversion

Land diversion, or 'set aside' as it is often termed, entails the removal of areas of land from agricultural production, either temporarily or permanently [Potter, 1991]. Such land may be devoted to other non-agricultural uses, allowed to lie fallow, or abandoned to revert to natural vegetation. Within the EC, the introduction of short-term set aside schemes was designed primarily to reduce the level of agricultural surpluses, whereas in the USA long-range land diversion programmes have been in existence since the 1930s, not only to lessen over-production, but also as a means of reducing soil erosion, water pollution and groundwater depletion. Although EC, and therefore UK, agricultural legislation increasingly attempts to encompass environmental concerns [Robinson, 1991], the importance of building conservation goals into land diversion schemes, and targeting those areas most likely to benefit environmental objectives is not fully recognised [Potter, 1991]. In the absence of a land diversion programme aimed directly at reducing nutrient losses from agriculture, this study examines the various ways in which current schemes may be utilised to improve water quality and aquatic ecosystem health in general. The legislation governing land diversion in the UK will be outlined in Section 4.3.3.2.

Estimates of the hectarage likely to be involved in agricultural land diversion in the UK vary considerably. A conservative figure is probably 3 to 4 million hectares, between the baseline year of 1985 and the year 2000 [NCC, 1987g]. With such large areas potentially available, the targeting of land diversion programmes would not only ensure efficiency in economic terms, but also maximise the environmental benefits. Three main objectives have been proposed in order to produce criteria for the targeting of land diversion [Potter *et al.*, 1991]:

(a) To match cropping patterns to environmental conditions best suited to them

High cereal prices and improved technology have enabled arable production on land of poorer quality, which may be more suitable for livestock rearing. Research has demonstrated that while yields on poorer areas may be comparable with those on better quality land, they can only be achieved through much higher applications of fertiliser and agrochemicals. Additionally, the use of very sandy or clayey soils for arable production may lead to soil compaction and erosion.

(b) To protect environmentally vulnerable land from degrading uses

Particular types of soil, topography and land use are more likely to be affected by water and wind erosion. For instance, areas vulnerable to periodic flooding generally consist of floodplains and marshes brought into arable production through costly protection and drainage systems, and may be more suited to low-intensity grazing.

(c) To restore natural and semi-natural ecosystems for wildlife and amenity

Agricultural land diversion could provide unique opportunities for the restoration and creation of habitats lost through agricultural intensification. Herb-rich grassland, heathland, wetland and woodland may be restored, depending on local conditions of climate, topography and soils. The selection of areas for restoration and creation of habitats could be partly determined by the location of existing designated nature conservation areas, which would provide reservoirs of species for recolonisation.

The application of these objectives to land diversion schemes would produce substantial benefits for water quality, as well as for nature conservation in general. The targeting of land vulnerable to soil erosion would directly reduce nonpoint phosphorus losses to surface waters. The removal of floodplains from arable production and the restoration of wetland ecosystems would not only create valuable habitats, but would also improve general water quality.

Unmanaged river corridors, complete with aquatic-terrestrial ecotones, possess a diverse physical structure composed of pools, riffles, secondary channels, backwaters, fringing marshes and floodplain woodland [RSPB, NRA and RSNC, 1994]. Engineering works on rivers carried out to alleviate flooding or improve drainage for agricultural purposes may seriously degrade the ecological value of both the main channel and the adjacent floodplain habitats (Fig.2.9). Such operations tend to isolate the river from its floodplain, which results not only in loss of habitat, but also reduces the ability of the floodplain to carry out many of its natural functions, including nutrient and sediment storage. The diversion of agricultural land adjacent to water courses could, therefore, provide opportunities for the restoration of river corridors, for the benefit of both nature conservation and water quality.

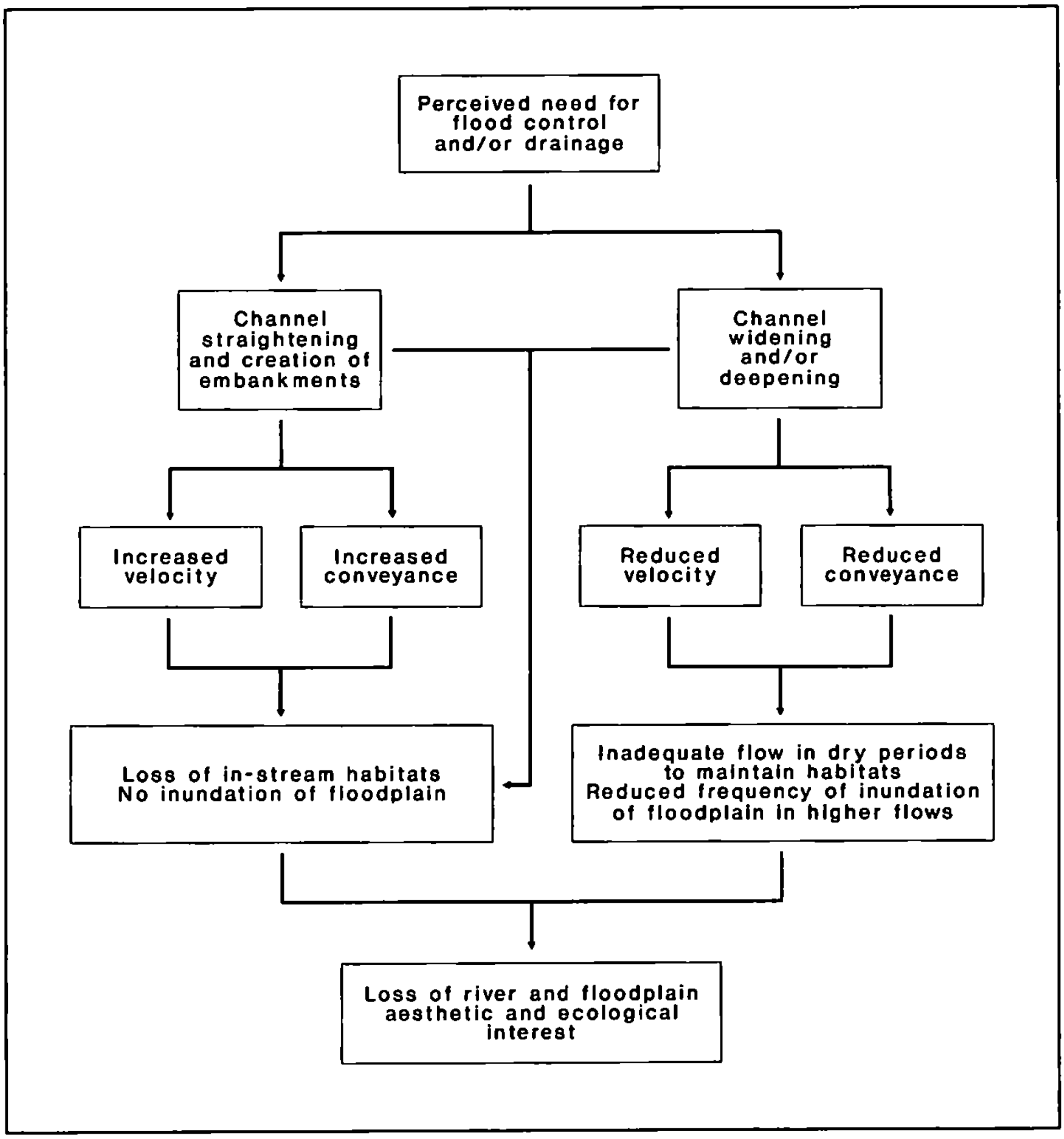

Fig.2.9: The Impact of Engineering Practices on River and Floodplain Ecology

Based on RSPB, NRA and RNSC [1994]

The design and implementation of river management schemes increasingly incorporate habitat conservation or improvement objectives [RSPB, NRA and RSNC, 1994]. Such schemes do not, however, necessarily involve restoration of the entire river corridor, even in cases where enhancement of nature conservation interest is the principal objective. A prime example of this piecemeal approach to the issue of river corridor restoration is the establishment of riparian vegetation specifically to retain sediment and nutrients, in order to improve water quality. The use of such 'buffer zones' represents a limited, more pragmatic approach to the wider concept of river corridor restoration, utilising one of the several functions attributable to aquatic-terrestrial ecotones. Nevertheless, it has been shown that buffer zones can contribute significantly to the reduction of nutrient losses from agricultural land [Muscutt *et al.*, 1993; Vought *et al.*, 1995], and may, therefore, provide a practical function for diverted land.

A buffer zone may be defined as "a permanently vegetated area of land... adjacent to a watercourse and managed separately from the rest of a field or catchment" [Muscutt *et al.*, 1993, p.60]. The vegetation may consist of trees, grasses, or aquatic plants. The efficacy of nutrient buffer zones appears to depend on various factors including width, slope, soil type, vegetation cover, and the extent of artificial drainage [Phillips, 1989a]. Where grazing occurs, buffer zones may be created by the removal of livestock a certain distance away from the river. Ungrazed grass buffer zones have been shown to be reasonably efficient in reducing nitrate loss, especially in undrained floodplains, through the process of denitrification [Haycock and Burt, 1990]. However, in most free-draining agricultural soils phosphorus is mainly transported to floodplains through surface run-off, so that for this element woodland buffer zones appear to be more effective in controlling loss to surface waters. Woodland soils generally possess a greater hydraulic conductivity than those of grasslands [Gregory and Walling, 1973], thus reducing surface runoff, and allowing the deposition of particulate material. A dense, well-developed canopy reduces the intensity of rainfall, lowering rates of erosion. In general, dissolved phosphorus forms are retained through infiltration, uptake and adsorption, while particulate forms are physically intercepted and deposited [Lee, Dillaha and Sherrard, 1989].

However, "all riparian forests are not created equal" [Phillips, 1989b, p.221]. Afforestation of pasture catchments alone, may actually increase sediment loads and nitrogen and phosphorus concentrations in streams [Smith, 1992]. Artificial drainage and the lack of close riparian ground cover within plantations increases erosion and therefore nutrient export. For the development of efficient buffer zones with high nutrient filtering and retention capacity, entire woodland ecosystems are therefore necessary.

Whilst the potential of buffer strips for reducing nutrient loads is clear, several critical questions have been raised [Muscutt, *et al.*, 1993]. First, investigations into their efficacy have been carried out in other countries (mainly the United States), where soils, climate and drainage systems may be different from those of the UK. Second, the long-term performance of constructed buffer strips has yet to be evaluated. It has been suggested that riparian areas may reach an equilibrium, such that retained nutrients would subsequently be released [Ormenik, Abernathy and Male, 1981], or that conversion to more mobile nutrient forms may occur [Dillaha *et al.*, 1986]. Additionally, from the evidence presented above [Smith, 1992], it would appear that during initial planting and establishment, nutrient loads to adjacent streams may actually be increased, and that this may continue until adequate woodland ground cover develops.

It is probable that buffer zones will, increasingly, be designed with the dual aim of enhancing both water quality and nature conservation interest. However, in certain respects these two objectives may not be reconcilable. The trapping and assimilation of excess nutrients in the riparian zone may increase the fertility of the soil to the extent that a diverse, species-rich plant community will not develop [Bakker, 1987]. The effects of nutrient buffer zone vegetation on the aquatic ecology may not necessarily be beneficial, especially in terms of light levels and organic matter inputs [Vannote *et al.*, 1980; Delong and Brusven, 1994]. The type of vegetation used for buffer zones needs to be appropriate to particular reaches of a river, from headwaters to mouth.

> "[The] quantity and quality of allochthonous inputs from riparian and other terrestrial vegetation has far-reaching implications, potentially affecting dynamics of litter decomposition and communities associated with litter processing, fish community structure and production, and invertebrate community structure". [Delong and Brusven, 1994, p.59]

It follows that, in order to benefit both water quality and nature conservation interests, the design and management of buffer zones need to be based on a sound understanding of river ecology.

The concept of using buffer zones has been applied not only to nutrient retention, but also to a number of other functions. The widths required for buffer zones to achieve prescribed functions are illustrated in Fig.2.10. For nutrients, a width of between 15m and 30m is thought to be adequate to reduce loadings significantly, whereas for bankside habitat conservation a zone of 20m to 170m is necessary [Haycock and Muscutt, 1995]. This comparison serves to illustrate further the apparent disparity between criteria for nutrient reduction and those for nature conservation. From an economic perspective, the costs involved in establishing a buffer zone sufficiently wide to conserve or enhance nature conservation interests would be considerably more than for nutrient retention. In summary, targeted land diversion schemes which aim to enhance both water quality and nature conservation interests are unlikely to succeed unless the more exacting criteria necessary for the latter objective are adopted.

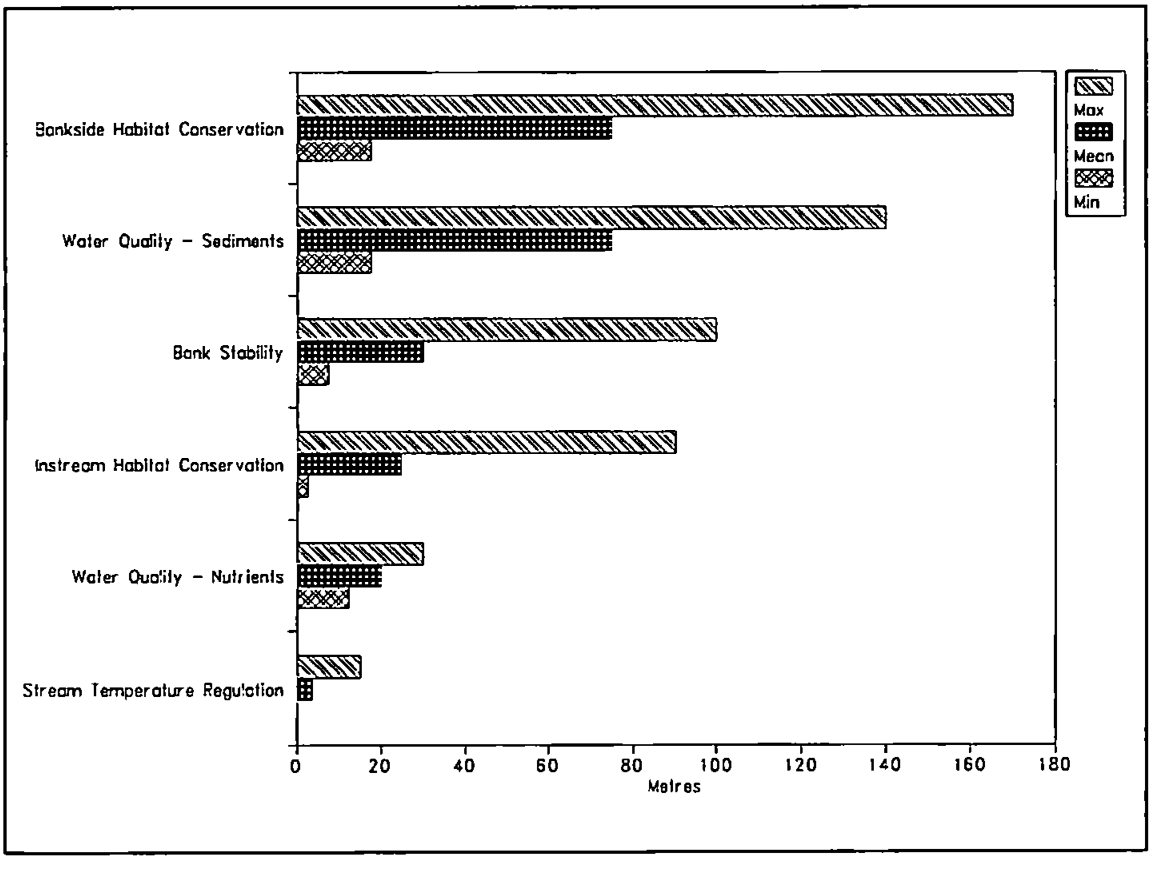

Fig.2.10: Widths of Buffer Zones to Achieve Prescribed Functions

Adapted from Haycock and Muscutt [1995]

With respect to land diversion schemes in general, it must be borne in mind that the primary objective is to reduce agricultural output through supply control. The environmental benefits to be gained as a side-effect are bound to be less than might be achieved through policies specifically aimed at improving the countryside [Hodge, 1992].

> "The value of output reduction in improving the environment varies considerably, depending where it takes place... On the other hand,... the value of a tonne of wheat not produced in one location does not vary much from that not produced in another" [*ibid.*, p.68].

In other words, the apparent complementarity of agricultural and environmental objectives in land diversion schemes may be rather superficial. It may, therefore, be necessary to develop separate land diversion schemes which address the specific problems of environmental degradation in the wider countryside.

2.2.3 Agricultural Extensification

Extensification of agriculture is defined as the adoption of farming practices which reduce production per unit area [NCC, 1987g]. The considerable post-war decline in the extent and quality of semi-natural and good wildlife habitats is largely the result of the intensification of agriculture [NCC, 1990]. The system of safeguarding *sites* of nature conservation interest, while crucial in maintaining examples of nationally and internationally important habitats, has not prevented massive declines in wildlife populations and habitats in the wider countryside. The survival of a network of good wildlife habitats in the wider countryside is not only important in its own right, but is also necessary to support reservoir populations and provide corridors of dispersal for species protected by site designation. Intensive land use can also directly affect adjacent designated areas. The problem of eutrophication of freshwater SSSIs is a prime example of the way in which intensification of agriculture and consequent deterioration of the wider countryside can impinge upon the quality of designated areas. Extensification of agriculture may, therefore, contribute significantly to the conservation of both the wider countryside and eutrophicated freshwater SSSIs.

It may be thought, especially in the context of this study, that a lessening in the intensity of agriculture could be achieved simply through reductions in fertiliser use. Application rates of inorganic nitrogen in England and Wales increased five-fold between 1953 and 1974, and doubled again in the following decade [Chalmers, Kershaw and Leech, 1990]. Use of inorganic phosphorus, on the other hand, increased by about 50% during the former period, and has changed little since then. The higher rate of increase in the application of inorganic nitrogen fertiliser is attributable to its high solubility, to elevated crop requirements induced by changes in the type of cropping, and to the higher yield potential of new varieties. Overall increases in the use of inorganic fertilisers may be explained partly by the fact that they possess certain advantages over organic manures, particularly in terms of convenience and economy [Arden-Clarke, 1988], but government subsidies up to 1974 [Chalmers, Kershaw and Leech, 1990], and the trend towards specialisation and the use of new varieties and cropping patterns has also engendered a reliance on inorganic forms.

It has been proposed by several commentators that a general decrease in the level of inorganic fertiliser application rates, exercised through the levy of taxes or by a quota system, would produce a beneficial effect on the environment. Such studies tend to concentrate on the problem of nitrogen, to the point of using the terms 'fertiliser' and 'nitrogen' as synonyms. Some of the results of these studies are given in Table 2.4.

Table 2.4: Predictions of Reductions in Fertiliser Use According to Different Levels of Tax Imposition

Study	Tax (%)	Reduction in Use (%)
England [1986]	100	10 - 24
Burrell [1989]	10	5
McCorriston and Sheldon [1989]	10 25	6.5 16.25

It is difficult, however, to extrapolate from these results the possible outcome of a tax on inorganic phosphorus fertilisers. Furthermore, it is debatable whether a reduction in application rates *per se* would achieve a significant lowering of export levels, for two reasons. First, as noted above, application rates of phosphorus fertilisers have not risen at the same pace as those of nitrogen. Second, since phosphorus is normally transported largely in the particulate form, through soil erosion, a reduction in the application rate may not necessarily lessen levels of export. It follows, therefore, that a reduction in the intensity of farming *practices* may be more beneficial.

Many of the habitats and rural landscapes which we value have been formed and sustained by land management practices associated with traditional forms of agriculture [NCC, 1990]. In general, most natural and semi-natural habitats have survived because they are still subject to low intensity forms of management. For instance, the Culm grassland of North Devon is a complex of species-rich, wet acidic grassland, heath, fen and mire communities, the diversity of which is maintained by summer grazing, winter burning, topping and scrub coppicing [Wolton, undated]. This low intensity management programme has evolved over centuries in order to enable the use of seasonally waterlogged, poorly-drained land for grazing. Without such management, succession would transform the grassland into scrubby areas of lower conservation value. Through agricultural 'improvements', such as drainage, fertilisation and reseeding, much of the Culm grassland has been lost. The remaining areas survive as a result of the continuation of traditional management practices.

Extensification of agriculture might, therefore, entail a reversal of the trend towards intensification, incorporating aspects of traditional farming. This may be achieved by identifying the features of modern agriculture which are detrimental to nature conservation in general, and water quality in particular, and applying traditional practices to produce the benefits shown in Table 2.5. Traditional mixed farming, which was once widespread in lowland Britain, may be less detrimental to water quality than intensive agriculture in several respects. The combination of tillage and livestock rearing ensures full use of animal wastes, reducing the need for additional fertiliser. Mixed farming also requires the maintenance of hedges and other boundaries [NCC, 1990], which provide linear barriers to soil erosion. The use of crop rotations and spring-sown, rather than winter-sown, cereal varieties reduces fertiliser requirements and soil erosion.

Floodplains used as grazing marshes are maintained by low intensity management involving summer grazing and a single cut for hay, rather than being drained. Livestock production under traditional mixed farming generally involves lower intensity grazing, with cattle being left outside during winter or bedded on straw [NCC, 1991a]. Fodder is usually produced in the form of hay, rather than silage. These practices mean that smaller amounts of liquid waste are produced, which reduces the risk of point source losses of nutrients.

Table 2.5: The Nature Conservation and Water Quality Benefits of Traditional Farming Practices

Derived from Arden-Clarke [1988], NCC [1990] and NCC [1991a]

Intensive Practices	Traditional Practices	Nature Conservation Benefits	Water Quality Benefits
Specialisation	Mixed farming	Greater diversity of habitats across farm, including hedgerows; less need for pesticides	Less inorganic fertiliser required; retention of hedgerows, providing barriers to soil erosion
Continuous cropping	Rotational cropping	Greater in-crop species diversity; larger biomass and diversity of soil fauna	Lower fertiliser requirements; less soil erosion; maintenance of soil organic matter
Greater reliance on machinery	More labour intensive	Retention of hedges and smaller size fields increasing diversity of species and habitats	Less soil compaction; contour cultivation possible, reducing soil erosion
Cultivation of winter-sown cereals	Cultivation of spring-sown cereals	Food supply for birds during breeding season	Less fertiliser needed; less soil erosion as a result of winter cover
Drainage of wet grassland	Maintenance of grazing marshes	Habitat for breeding waders and other wildlife retained	Retention of floodplains for sediment and nutrient storage
Livestock reared in intensive units	Livestock reared less intensively on open pasture	Unimproved grassland retained	Less concentration of animal waste, leading to lower risk of point source nutrient loss
Silage produced for fodder	Hay produced for fodder	Hay meadows retained	No risk of pollution from silage liquor

Under traditional farming practices it is less likely for land to be cultivated, mown or grazed right up to the edges of rivers and lakes, thereby lessening the extent of soil and nutrient losses, and the destruction of natural wetland ecotones. The economies of scale necessary for the success of intensive agriculture include the use of large machinery for the cultivation of sizable fields. Reliance on such equipment is not compatible with the practice of contour or cross-slope cultivation, which is capable of reducing average soil loss by as much as 50% [Clasen, Rast and Ryding, 1989].

It would appear that a widespread reversal of the trend towards intensification of agriculture is likely to elicit a positive effect on water quality, and to benefit nature conservation in general. Clearly, there is a need for the development of some form of low-input, extensive agriculture. Although traditional mixed farming practices are more beneficial to nature conservation, they do not necessarily represent a comprehensive, recognised system with which to replace intensive agriculture. Additionally, the positive effects of traditional farming on the environment are generally incidental, rather than intentional. There are, however, a number of farming systems which not only provide a means of achieving low-input, extensive agriculture, but which also aim to contribute direct benefits for wildlife. The most well-known and accepted of these is the organic system, which is now recognised and defined by EC legislation.

Organic farming does not represent merely a return to traditional agriculture, since it utilises techniques developed from an understanding of and research into crop and livestock breeding, soil science and ecology [NCC, 1990]. A basic principle of organic farming is to use ecological processes within the food production system. It does not, therefore, employ artificial fertilisers or pesticides, but instead relies on integrated systems of livestock management, crop rotation, and maintenance of natural predator populations to control pests. Standards for organic agriculture are regulated by the Soil Association (SA) through its Symbol Scheme. The SA Standards require that all existing areas of semi-natural vegetation are retained and include management prescriptions for hedgerows and unimproved pastures which are aimed at maintaining nature conservation value. Conversion to organic farming takes place over a minimum two year period, after which, if the holding meets the requirements, the SA issues a Certificate of Registration. An inspection and review of each holding is carried out every year.

At present, organic farming occupies about 50,000 hectares [SA, 1994], but the SA is campaigning for 20% of British farmland to be converted by the end of the century [NCC, 1990]. This aim may be encouraged by the introduction of grant-aid for organic conversion, under the EC Agri-Environment programme, which will be discussed in

Section 4.3.3.3. Elsewhere, organic farming is experiencing a marked expansion, with consumer demand outstripping supply in both Canada [Hill and MacRae, 1992] and Australia [Conacher and Conacher, 1991]. In Finland, organic farming has become a major topic in public discussion about the future of agriculture [Korva and Varis, 1990].

The SA claims that organic farming yields are about 20% less than those of intensive agriculture [SA, undated], but independent studies have shown that, for some crops, as much as 50% less is obtained [Korva and Varis, 1990]. However, taking into account the surcharges which certified produce can attract, gross margins can be higher for organic farmers [Poutala *et al.*, 1994]. A study in the USA concluded that widespread adoption of organic farming methods would increase net farm income and fulfil domestic demand for agricultural products, but would increase consumer food costs and reduce export levels [Olson, Langley and Heady, 1982].

The benefits of organic farming to nature conservation, in comparison to intensive agriculture, are, according to the NCC [1990]:

(a) the prohibition of artificial pesticides and fertilisers reduces danger to wildlife and ecosystems (see comments below)
(b) the utilisation of natural predators to control agricultural pests requires the maintenance of diverse habitats in and around crops
(c) lower livestock grazing densities reduce pollution risks
(d) maintenance of soil structure and fertility by encouraging an abundant and diverse soil fauna provides food for birds and other wildlife
(e) the combination of tillage and livestock increases diversity of habitat and requires the maintenance of hedgerows

On the whole, the recognition of these benefits is based on scientific evaluations which indicate that invertebrates, plants and birds are present in greater numbers and diversity on organically farmed land [Arden-Clarke, 1988; Fuller, Hill and Tucker, 1991; Moreby *et al.*, 1994]. Additionally, maintenance of soil organic matter, the use of practices which reduce soil erosion, and effective recycling of animal wastes [Pimentel, 1993] lower susceptibility to losses of phosphorus. However, the popular literature produced by the SA belies any understanding of soil processes, as illustrated by the following extract:

"**Q. What about artificial fertilisers?**
Artificial fertiliser applied to the soil leaches into groundwater and rivers. Excessive nitrates are a health hazard in drinking water because they can be converted into carcinogens within our bodies. Fertiliser leaches into rivers and lakes causing algal blooms, killing aquatic life by starving them of oxygen". [SA, undated, p.2]

The implication of this statement is that organically-derived fertilisers cannot produce high nitrate levels in drinking water or cause eutrophication of aquatic systems. It is not the use of organically-derived manures *per se* which reduces the level of nutrient losses from organic farming systems, but the lower overall intensity of such practices and the attention paid to soil health and conservation.

2.3 Summary

This overview highlights the numerous sources of phosphorus in the environment and the means by which losses to surface waters may be reduced or eliminated. Phosphorus in effluent from STWs is an important and easily controlled source. It is clear that the use of phosphate-free detergents would, in most cases, be insufficient to improve the water quality of eutrophicated water bodies. The installation of tertiary treatment to remove phosphorus at larger STWs should, therefore, be regarded as a cost-effective investment in water quality and nature conservation. However, there is a need for alternative measures at the many smaller STWs which discharge into streams feeding freshwater SSSIs. The use of constructed reed beds for tertiary treatment may seem an attractive option, but advances need to be made in terms of their ability to remove phosphorus, consistently and over time, before such facilities can be regarded as effective means of reducing loadings. Diversion of phosphorus-rich effluent may be a viable measure for some sites, providing the flushing rate is not altered significantly.

Urban stormwater can be a significant source of phosphorus, particularly in drainage systems utilising CSOs. There exists a wide range of options for urban stormwater management. These include allowing stormwater to infiltrate as much as possible, thus avoiding many of the problems associated with surges and sewage contamination. Surface water detention ponds, incorporating vegetative treatment, can also be effective in reducing the level of nutrients and contaminants entering sensitive aquatic systems.

With respect to agricultural sources of phosphorus, a significant reduction in the intensity of farming practices is required, particularly among specialist enterprises. This may be achieved either through land diversion schemes or through extensification, but is unlikely to occur as a result merely of adherence to good agricultural practice. The adoption of a low-input, extensive agricultural system, such as organic farming, would not only improve water quality, but also benefit nature conservation in the wider countryside. However, it needs to be recognised that opportunities for land diversion and extensification are only available because of agricultural surpluses, yet solutions to the problems caused by intensification are unlikely to lie in changes in agricultural policy alone. The success of land diversion and extensification schemes, in terms of nature conservation, may require the targeting of land uses and types.

The individual methods of reducing phosphorus export to eutrophicated water bodies have been described in isolation from each other. In practice it is likely that a variety of approaches are necessary, in order to achieve appropriate remediation strategies for given sites.

Chapter 3

Prediction and Evaluation of External Phosphorus Loadings

In order to determine the most effective means of reducing external phosphorus loadings on eutrophicated water bodies, it is necessary to identify the major sources within their catchments, and the relative contribution of each source to the total load [Loehr, Ryding and Sonzogni, 1989]. The most accurate method for estimating total loadings is direct measurement at the mouth of inflowing rivers, but this produces little information on the relative contribution of different sources. An alternative method, which is particularly useful for estimating loadings from nonpoint sources, is to employ unit area loads or export coefficients. This approach assumes that, under average hydrological conditions, a given land use will export a reasonably constant nutrient load per unit land area to the receiving waters draining the catchment. The area of each land use category in a catchment is multiplied by the appropriate export coefficient and the results summed to give the total nonpoint load. Inputs from atmospheric deposition, and point sources, particularly effluent from sewage treatment, may be calculated in the same general way. The total load from all sources is taken to be the sum of these calculations. Direct measurement of concentration levels at the inflows is also required for calibration purposes. While the use of export coefficients to predict nutrient losses is an attractive, simple tool, requiring minimal catchment data input compared to more complex models, there are a number of limitations to the process, which are examined in this study.

3.1 The Determination and Use of Export Coefficients

A large number of studies have been carried out in order to quantify nutrient losses from different land use types. Several authors [Vollenweider, 1968; Uttormark, Chapin and Green, 1974; Reckhow, Beaulac and Simpson, 1980] have attempted to collate such information, converting the data into uniform units, and tabulating them in a format convenient (or otherwise) for the selection of relevant export coefficients. Very few of the investigations included in these reports were carried out specifically to estimate losses to standing waters, the objective being to measure nutrient and often other material losses from distinct areas of land. Changes induced by subsequent transport are therefore not included in these results. This point is particularly relevant for nonpoint losses of phosphorus, since this element is largely exported in particulate forms, which may not be transported sufficiently far to enter a lake.

Uttormark, Chapin and Green [1974] compiled a large amount of data from work conducted almost entirely in the USA. From this they produced an average phosphorus export coefficient for agricultural land of 0.3 kg/ha/a, within a 'typical' range of 0.1 to 1.0 kg/ha/a. Reckhow, Beaulac and Simpson [1980] also analysed the results of a large number of experimental studies, again mainly undertaken in the USA. A mean value for phosphorus export from agricultural land was calculated as 1.134 kg/ha/a, within a range of 0.08 to 3.25 kg/ha/a. These syntheses of results serve to illustrate the degree of variation possible, even where a large database is available.

As well as a compilation of export coefficients, Reckhow, Beaulac and Simpson [1980] provide a manual for the selection and application of export coefficients for a particular watershed. The factors which need to be taken into account are summarised in Fig.3.1. Reckhow and Simpson [1980] present a phosphorus model for lake quality management which uses generalised export coefficients to predict loading. The coefficients are based on data from Uttormark, Chapin and Green [1974] and Reckhow, Beaulac and Simpson [1980] and are shown in the table below.

Table 3.1: General Phosphorus Export Coefficients

Reckhow and Simpson [1980]

	Agriculture	**Forest**	**Precipitation**	**Urban**	**Input to septic tank**
	kg/ha/a				kg/ca/a
High	3	0.45	0.6	5	1.8
Mid	0.4 - 1.7	0.15 - 0.3	0.2 - 0.5	0.8 - 3	0.4-0.9
Low	0.1	0.02	0.15	0.5	0.3

The land use categories defined by Reckhow and Simpson [1980] are very broad, and do not differentiate between, for example, arable and grassland. Consequently, the range of values cited are extremely wide, since "the broadness of the 'source categories' leaves room for much variation, due to basin geology, erosional patterns, and intensity and types of use" [*ibid.*, p.1442].

Forest

- Species Type
- Soil Type
- Vegetation Age
- Climate
- Disturbance
- Fertilisation

Agricultural

Crops
- Soil Type
- Fertilisation
- Tillage
- Type of Crops

Pasture
- Grazing Regime
- Fertilisation
- Animal Density

Waste Storage
- Impervious Areas
- Animal Density

Urban

Residential
- Housing Density
- Vegetation Cover
- Fertilisation
- Pet Density

Public Parks

Commercial and Industrial

Sewage

Septic Tanks
- P Adsorption
- Soil Drainage
- Soil Permeability
- Groundwater
- Slope
- System Age
- Plant Uptake
- Cleaning Frequency

Treatment Works
- Type of Plant
- Combined Sewers
- P Detergent Use

Atmosphere

Agricultural

Urban

Fig.3.1: Factors for Consideration in the Selection of Export Coefficients

Based on Reckhow, Beaulac and Simpson [1980]

From this range of values the authors selected what were thought to be appropriate export coefficients for a particular example, namely Higgins Lake, Michigan, USA. In selecting these values, the factors listed in Fig.3.1 were taken into consideration. The final figures for Higgins Lake catchment are reproduced in the table below.

Table 3.2: Phosphorus Export Coefficients for Higgins Lake Catchment

Reckhow and Simpson [1980]

	Agriculture	**Forest**	**Precipitation**	**Urban**	**Input to septic tank**
	kg/ha/a				kg/ca/a
High	0.8	0.4	0.5	1.5	1.0
Mid	0.3	0.2	0.3	0.9	0.6
Low	0.1	0.02	0.15	0.5	0.3

Reckhow and Simpson [1980] explain the procedure involved in using their phosphorus model. They state:

> "The selection of appropriate phosphorus export coefficients is a difficult task... In this example, however, we have not explained our analyses or thought processes involved in the choice of export coefficients... In the interests of a concise presentation of this technique, it was not included. However, when this procedure is used... we strongly urge that the analyst present... all the information used in the selection of export coefficients" [*ibid.*, p.1443].

The authors emphasise that the values selected by them "describe conditions within the Higgins Lake watershed [and that] proper choice of export coefficients is a function of knowledge of... the watersheds of candidate export coefficients" [*ibid.*, p.1443]. Reckhow, Beaulac and Simpson [1980] concede that, "Despite the existence of pertinent literature, the selection of application export coefficients is still an unavoidably subjective task" [*ibid.*, p.21]. They recommend consulting the original experimental work for details on the conditions under which literature values were generated. This example of the selection of export coefficients serves to underline the subjective nature of such processes. It follows that without an explanation of the analysis involved, it is very difficult to assess the validity of values chosen.

Export coefficients not only present difficulties in their selection, but there are also limitations in their application, in that certain types of catchments and lakes are less suited to their use. First, highly urbanised catchments, in which natural drainage patterns are often severely disrupted [House *et al.*, 1993], present difficulties. Although export coefficients are available for urban areas, they should be applied with caution and, preferably, with knowledge of the sewerage system. Second, since export coefficients represent the transport of nutrients *via* surface and subsurface flow, they are less suitable for application to lakes which are predominately groundwater-fed. Third, the use of export coefficients is problematical where a series of water bodies are involved, since the sediments of lakes upstream of a site may retain a significant proportion of the loading [Canfield *et al.*, 1989].

From the information outlined above, the main limitations of the use of export coefficients may be summarised as:

(a) Export coefficients express losses from one land use type to another, or to surface streams, and do not necessarily take account of subsequent transport processes, nor the proximity of particular land uses to the water body.

(b) Experimental work on nutrient export levels have been carried out mostly in the USA, where hydrological, climatic and agricultural conditions may be different from those of the UK.

(c) A wide range of figures are available for the same land use type, to the extent that a detailed knowledge of catchment characteristics and farming patterns is necessary before appropriate export coefficients may be selected.

(d) The selection of export coefficients is necessarily a subjective exercise, in which choices need to be justified and assumptions made explicit.

(e) For particular types of catchments and lakes, the use of export coefficients may be inadvisable.

3.2 Application of Export Coefficients in the UK

The use of export coefficients to predict lake nutrient loadings has been widely used in North America and, to some extent, in continental Europe. In the UK, this approach has received limited attention, until recently. Johnes and O'Sullivan [1989] and Johnes [1990] employed export coefficients to quantify the nutrient loadings on Slapton Ley, Devon and the River Windrush, Oxfordshire, respectively. Johnes, Moss and Phillips [1994] developed the approach further and applied it to a wide range of lakes in Britain. The basic model described in the above papers has been adopted for use in this study, but certain aspects merit evaluation.

Phosphorus exports within catchments are commonly categorised as originating from either point or nonpoint sources. A further division may be made within the nonpoint category, in that phosphorus from organic sources (i.e. animal wastes) are distinct in origin from inorganic sources (i.e. artificial fertilisers). Vollenweider [1968] made this distinction, and calculated nonpoint organic and inorganic losses separately. However, the vast majority of subsequent work has employed export coefficients based on land use type only. This change in approach may have occurred for the following reasons:

(a) The need to reduce the amount of catchment data necessary to predict phosphorus loadings.

(b) The recognition that land use and management practices are often more important in determining the level of phosphorus export than application rates of either organic or inorganic fertiliser [e.g. Lambert *et al.*, 1985; Thomas *et al.*, 1992; Mostaghimi, Younos and Tim, 1992; McIntyre, 1993].

(c) The difficulty involved in measuring phosphorus export from organic and inorganic fertilisers separately, owing to the complex nature of soil phosphorus processes [White, 1981; Sharpley and Smith, 1990].

The main advantage of using a model which distinguishes between organic and inorganic sources is in the consequent ability to produce more specific recommendations for reducing loadings. For instance, if inorganic fertilisers are identified as the main source, then different management strategies would be recommended than if organic sources were predominant. The problem with this approach, however, is in obtaining appropriate, scientifically-based data on rates of losses for organic and inorganic fertilisers. In over forty research and review papers which cite export coefficients, very few specifically state the source of the phosphorus, that is, whether it is of organic or inorganic origin. The lack of relevant data has led to the use of assumed values, particularly for organic sources, in the work of Johnes and O'Sullivan [1989] and Johnes [1990].

The model used by these authors assumes a standard loss of 3% of phosphorus in animal wastes, applied as manure and directly voided. The references cited by Johnes and O'Sullivan [1989] for production and losses of phosphorus from livestock are Vollenweider [1968], Cooke [1976] and Gostick [1982], while Johnes [1990] specifically cites Cooke [1976] as the source for the 3% figure, who gives no such detail. It seems likely that this 3% was derived from Vollenweider [1968] who envisaged a 'best-case' scenario of 1% loss of phosphorus from organic manures and a 'worst-case' one of 5% loss. The mean of these two figures is 3%. This hypothesis is strengthened by the use of 17% as the loss rate for nitrogen, which is roughly the mean of 10% and 25%, the 'best-case' and 'worst-case' figures presented by Vollenweider for this element. Phosphorus losses are, therefore, based on an average of values stated to be "... of the order of 1-5% at the most" [*ibid.*, p.106]. In contrast, work in Northern Ireland and the USA has produced generally lower figures for losses of phosphorus from animal waste, as shown in Table 3.3.

Table 3.3: Phosphorus Loss Rates for Animal Wastes

Type of Waste	Loss (%)	Land Use	Country	Source
Slurry	0.1 - 0.5	Grassland and Fallow	NI	McAllister & Stevens [1981]
Farmyard Manure	0.6 - 1.2			
Dairy Manure	1.3	Grassland	USA	McLeod & Hegg [1984]
Poultry Manure	2.4			
Poultry Litter	1.2	Grassland	USA	Edwards & Daniel [1994]
Poultry Manure	0.5			
Swine Manure	2.6			

A further assumption is made in relation to animal wastes, in that 10% to 15% is allowed for losses during storage for both nitrogen and phosphorus. In the case of nitrogen it is very likely that losses of ammonia, dinitrogen and nitrous oxides will occur [Arden-Clarke, 1988]. However, the same loss rates are assumed for phosphorus, but since, during its natural cycle, this element does not enter a gaseous phase [O'Neill, 1985], such losses appear to be unlikely. Furthermore, Johnes and O'Sullivan [1989] cite Gostick [1982], who specifically discusses losses of *nitrogen* during storage, while Johnes [1990] refers to Royal Society [1983] *The Nitrogen Cycle of the United Kingdom*.

The model devised by Johnes [1990] uses parish records as the source of information for the calculation of land use and livestock levels within catchments. Only in very few cases do parish boundaries coincide to any extent with catchments. In order to overcome this problem the percentage area of parish within a catchment boundary was calculated and then applied to the data for that parish. This adjustment necessarily assumes that land use and livestock levels are homogeneously distributed throughout each parish [Mitchell, 1990]. The degree of error produced by this assumption will increase as the area of catchment within the parish decreases. Small catchments impinging on several large parishes will, of course, produce the least reliable results.

An assumption implicit in the reliance on parish data is that all animal waste produced within the parish is disposed of within the same area. In the case of small to medium size mixed farms this is probably a reasonable assumption. For other types of farming, however, the situation may be quite different. For instance, intensive poultry and pig units produce large quantities of waste which they cannot utilise [North and Bell, 1990]. Disposal of this waste to other farmers, to manufacturers of garden fertilisers, or by dumping, may well involve transportation out of the parish.

The parish records consist of summaries of agricultural census returns, presented as totals for whole parishes. In order to safeguard the confidentiality of information for individual farms, where there are fewer than three holdings in a parish, the data are amalgamated at the county level [Clark, Knowles and Phillips, 1983]. This may mean that data for important parishes within a catchment are not available. Furthermore, many farms do not lie wholly within a single parish. Such farms are normally included in the parish containing the greater part. It follows that the area of agricultural land shown in the parish summary is only approximately comparable with the geographical boundaries of the corresponding parish of the same name. This lack of comparability is generally greater for smaller parishes.

An important limitation in the use of parish records for data on catchment land use is that they provide information on *agricultural* uses only. The extent of settlements and roads is not provided, so that the area of urban land in a catchment cannot be calculated. This lack of data necessarily means that potentially high nutrient export from urban areas are not included in the model.

The lack of available data on phosphorus losses from organic sources proved to be a limitation in the necessary adjustment of the model during development. Johnes [1990] carried out a sensitivity analysis of the model by changing each export coefficient by an initial 10% and then by 25%, whilst the remaining parameters were held constant. The results of this analysis were then used to adjust sensitive parameters within the observed range of literature values to optimise the model, "... although the coefficients available for livestock did not provide the necessary range for adjustment" [*ibid.*, p.310]. So, for phosphorus, the 3% loss rate for organic sources remained unamended, while some of the coefficients for inorganic sources were adjusted. The final model calibration for the River Windrush produced an error for phosphorus of only 0.15%, but bearing in mind the assumptions outlined above, it must be concluded that the coefficients for inorganic sources were adjusted to accommodate the lack of reliable data for organic sources.

As already discussed, there is a wealth of literature on export coefficients for different land use types, but very few of them specifically state whether the sources are organic or inorganic. Johnes [1990] selects a range of values for different land uses, citing Reckhow and Simpson [1980] as the reference. Contrary to the advice given in Reckhow and Simpson [1980] and quoted above, Johnes [1990] does not present sufficient information to explain the processes involved in selecting these export coefficients. Table 3.4 details the phosphorus loss rates from inorganic sources used by Johnes [1990]. No value is selected for urban inputs, since, as noted above, data for this type of land use are not available from parish records.

Table 3.4: Phosphorus Loss Rates Selected by Johnes [1990]

Parameter	Optimized Value (kg/ha/a)
Temporary grass loss	0.3
Permanent grass loss	0.1
Cereals loss	0.8(a) 0.65(b)
Root crops loss	0.8
Field vegetable loss	0.8(a) 0.65(b)
Oilseed rape loss	0.8(a) 0.65(b)
Woodland loss	0.02
Rough grazing and Set-Aside loss	0.02
Rainfall input loss	0.2
(a) First choice parameter for initial model calibration (b) Second choice parameter from final model calibration	

Direct comparison of the losses rates chosen by Johnes [1990] for organic and inorganic sources is hindered by the fact that for the former a percentage loss is given, whereas for the latter a rate of loss per hectare is used. However, the export coefficients for inorganic sources can be converted to a percentage loss, using data on fertiliser application rates given by Johnes [1990]. The figures produced are shown in Table 3.5.

Table 3.5: Export Coefficients for Inorganic Sources [Johnes, 1990] Converted to Percentage Loss of Fertiliser Applied

Land Use	P Export (kg/ha/a)	P Applied (kg/ha/a)	P Export (%)
Temporary Grassland	0.3	20.3	1.5
Permanent Grassland	0.1	6.8	1.5
Cereals	0.8 0.65	26.2	3.1 2.5
Root Crops	0.8	8.7	9.2
Field Vegetables	0.8 0.65	20.8	3.8 3.1
Oil Seed Rape	0.8 0.65	28.8	2.8 2.3

Comparisons may now be made between the assumed losses from inorganic and organic sources on different types of land use. Temporary and permanent grasslands receive applications of inorganic fertiliser as well as livestock waste directly voided onto the land. The fertiliser supply may be supplemented by additions of slurry and farm yard manure (FYM). According to Johnes [1990] losses on grasslands from inorganic sources are about 1.5%, compared with 3% from organic sources. There is no scientific data to support the idea that twice as much phosphorus is lost from livestock waste than from inorganic fertilisers on the same type of land. Indeed, a study in the Netherlands [Gerritse, 1981] demonstrated that pig slurry consists of about 80% *inorganic* phosphorus, and that nearly all organic phosphorus compounds are strongly retained in the soil. In sheep faeces deposited onto New Zealand pastures, 56% to 82% of phosphorus is present in the *inorganic* fraction [Nguyen and Goh, 1992]. Such high proportions of inorganic phosphorus in organic wastes would suggest that similar loss values are applicable to both organic and inorganic sources of phosphorus on grassland. For other types of land use losses of phosphorus from inorganic sources vary from 2.3% to as much as 9.2%. Yet it is assumed that a standard 3% of phosphorus from organic sources would be exported, regardless of land use.

The more general limitations on the use of export coefficients were outlined at the end of Section 3.1. It has been shown that there are a number of additional questions specifically related to the export coefficient approach adopted by Johnes [1990], which may be summarised thus:

(a) Despite the benefits of distinguishing between organic and inorganic sources of phosphorus, the validity of this approach is questionable in several respects.

(b) The lack of relevant data on losses from organic sources has led to the use of an assumed value (3%), the scientific basis for which is debatable.

(c) With respect to losses from inorganic sources, little explanation of the processes involved in the selection of values is provided.

(d) The use of parish records as the source for land use and livestock data introduces varying degrees of error into the calculations.

(e) Reliance on parish records for land use data means that urban areas are not accounted for in the model.

The methodology used by Johnes [1990] also forms the basis for further work by Johnes, Moss and Phillips [1994]. This study, under contract to the National Rivers Authority (NRA), developed a classification system for lakes. The work merits evaluation in that the reservations expressed above may have some bearing on apparent anomalies in the initial results.

Within the study, contemporary values of a range of hydrological, chemical and biological variables were used to 'hindcast' those of a reference baseline state (the 1930s), and compared. Percentage differences in these variables were then used to indicate changes in levels of eutrophication, acidification and infilling. Contemporary and baseline nutrient variables were determined using the export coefficient approach adopted by Johnes [1990]. The coefficients chosen for the prediction of loadings on the River Windrush, and subsequently Slapton Ley, were applied to ten other sites, selected to reflect a wide range of environments. The results of this exercise were compared to actual data for the ten sites and the coefficients adjusted in order to reflect regional variations.

In order to reduce the amount of data necessary to run the model for a far greater number of sites, a regional approach was then adopted [Johnes, Moss and Phillips, 1994]. Land use regions in England and Wales were identified, using the Land Utilisation Survey of the 1930s. Average export coefficients were devised for these regions, of which there are typically five to ten per county. Results obtained using the coefficients based on Land Use Regions appeared to correlate well with those from the catchment specific approach. The model was then tested on a further 94 British lakes and "shown to give reasonable results" [*ibid.* p.VII], although available contemporary data was not usually adequate for verification.

Comparisons were made between the values obtained for baseline and contemporary variables, and the percentage change calculated. The average percentage change for a total of eight variables, including lake total phosphorus and total nitrogen, were determined in order to provide an index of the degree of eutrophication over time. This overall percentage change was then categorised:

Class 1 - Negative changes and positive changes < 25%

Class 2 - 26% to 50% change

Class 3 - 51% to 100% change

Class 4 - 101% to 200% change

Class 5 - > 200% change

Examples of results obtained for individual lakes by the processes outlined above are shown in the Table 3.6.

Table 3.6: Provisional Eutrophication Classification of a Range of Lakes

Johnes, Moss and Phillips [1994]

Lake	**Lake Total P** (% change)	**Lake Total N** (% change)	**Degree of Eutrophication** (% change)	**Provisional Eutrophication Class**
Loe Pool	168	66	21	1
Slapton Ley	43	102	74	3
Burrator Reservoir	648	- 3	96	3
Barton Broad	16	451	105	4
Malham Tarn	138	- 47	120	4
Rutland Water	454	1867	775	5

With reference to Table 3.6, anomalies seem to occur in the classification of some sites. For instance, Loe Pool, which evidence suggests has undergone a high level of nutrient enrichment (Section 6.1.1), is assigned to Class 1. The opposite situation occurs with Malham Tarn. The majority of literature on this site (Section 6.7.1) suggests that little change in trophic status has occurred over the past forty years, and that this site would, more reasonably, appear in Class 1.

Another possible misclassification is that of Burrator Reservoir. This moorland water body, for which there is a lack of published data, receives no sewage effluent and lies within a catchment in which only low intensity farming and forestry take place. Yet, according to Johnes, Moss and Phillips [1994] the reservoir has experienced a 648%

increase in lake total phosphorus since the 1930s, a value which far exceeds those for lakes acknowledged as highly eutrophicated such as Slapton Ley (Section 6.1.2), Barton Broad (Section 6.3.1) and Rutland Water (Section 6.4.3). This extremely high figure contributed to the overall change of 96%, which places Burrator Reservoir in Class 3, alongside Slapton Ley.

The lack of adequate data for verification purposes alluded to by Johnes, Moss and Phillips [1994] is both a drawback inherent in the approach adopted, and a common problem in water quality modelling in general. The results generated by the use of export coefficients require for verification data on nutrient concentrations in inflows, which are seldom available. The alternative is to utilise data on in-lake concentrations. However, such values reflect not only inputs *via* surface and subsurface flow, but also groundwater and internal sources, and, importantly, the level of biological uptake. There is not necessarily therefore any correlation between actual in-lake concentrations and predicted values. Of course, if water quality monitoring were assigned a higher priority, and inflow concentration data available, this problem would not arise. However, under the current situation verification of the results is problematical.

The provisional results of the scheme are distilled into a simple classification index. Differing lake progressions may, however, be masked by such an index. For example, a lake categorised as being in Class 1, that is, assessed as being subject to a small degree of change, may be representative of two different scenarios. It may be a productive lake which has undergone a slight increase in nutrient status of no real consequence. Or it may be an unproductive lake which has experienced the same degree of enrichment, but with damaging repercussions for the ecosystem. This lack of distinction between sites in the same Class may lead to a situation in which those at the lower end of the trophic spectrum, in danger of losing their high conservation interest, appear to require no remedial action.

The results obtained by Johnes, Moss and Phillips [1994] are, at present, provisional, and a second phase of work is planned. This review highlights the need to address a number of issues:

(a) Limitations on the use of export coefficients for the calculation of contemporary nutrient loads must also apply to 'hindcasted' values.

(b) Some sites do appear to be misclassified, which may be a function of the export coefficient approach.

(c) The lack of suitable data for verification of results is a serious handicap which may only be partially solved by the commencement of short-term monitoring.

(d) The eutrophication classification index does not appear to distinguish between lakes undergoing differing trophic progressions.

3.3 The Vollenweider-OECD Model

The work of Vollenweider [1968] was an important step in terms of the classification of lakes according to their trophic status and the recognition of phosphorus as the key element which controls eutrophication in most lakes. Although his earliest model was based on limnological evidence inferred from a small data set, it represented a first approximation of the relationship between phosphorus loading and trophic response [OECD, 1982]. It was an attempt to provide a basis for the development of a trophic status index with respect to nutrient loading, which could be examined and developed.

Vollenweider [1968] was able to demonstrate that if annual external nutrient inputs to lakes were expressed as loadings per unit area, lakes of different sizes could be compared. He recognised, however, that the relationship between nutrient loading and in-lake concentration is dependent on a number of hydrological and morphological factors, not least of which is depth. Vollenweider therefore plotted data on phosphorus loadings against mean depth, adding boundaries between oligotrophic and eutrophic lakes, and introducing the notion of 'dangerous' and 'permissible' levels of loading. In the original context, these terms were used to denote the effect of eutrophication on historically oligotrophic lakes [OECD, 1982]. The numbers attached to the boundary lines may become targets for eutrophication control, whereas phosphorus loadings of naturally productive lakes may far exceed the 'permissible' boundary.

The work of Vollenweider was further developed through his involvement in the Cooperative Programme on Eutrophication of the Organisation for Economic Cooperation and Development (OECD) which was designed to quantify the relationship between nutrient load and trophic reaction [OECD, 1982]. The Programme, begun in 1973, represented a unique and comprehensive international cooperative effort to generate an extensive and reliable limnological database for the development of sound management principles for the control of eutrophication [Sas, 1989].

Data from the Programme were used to achieve new criteria for phosphorus loading. Improvements to the loading relationships were obtained through the inclusion of terms for hydraulic residence time and for rates of sedimentation [Vollenweider, 1975], producing Vollenweider's third model [Vollenweider, 1976]. Evaluating the data in 1978, Vollenweider and Kerekes were able to demonstrate a significant relationship between external phosphorus load and in-lake phosphorus concentration, as well as algal biomass (expressed as chlorophyll *a* concentration) [Sas, 1989]. Expert opinion was then employed to allocate a trophic category (oligotrophic, mesotrophic or eutrophic) to the lakes investigated. It was found that the trophic state of each lake lay within a definite range of values for phosphorus concentration and algal biomass (Table 3.7), so that the relationship could be ascertained statistically. Contemporaneous and subsequent work, including that of Rast and Lee [1978], Janus and Vollenweider [1981] and Jones and Lee [1982], has confirmed that, with few exceptions, lakes generally conform to the relationships established by the Vollenweider-OECD model [Jones and Lee, 1986].

Table 3.7: OECD Boundary Values for Trophic Categories

OECD [1982]

Trophic Category	Phosphorus (μg/l)	Chlorophyll a (μg/l)	Maximum Chlorophyll a (μg/l)	Mean SDT (m)	Minimum SDT (m)
Ultra-Oligotrophic	≤ 4	≤ 1	≤ 2.5	≥ 12	≥ 6
Oligotrophic	≤ 10	≤ 2.5	≤ 8	≥ 6	≤ 3
Mesotrophic	10 - 35	2.5 - 8	8 - 25	6 - 3	3 - 1.5
Eutrophic	35 - 100	8 - 25	25 - 75	3 - 1.5	1.5 - 0.7
Hypertrophic	≥ 100	≥ 25	≥ 75	≤ 1.5	≤ 0.7

The objective of the OECD Programme was to provide guidelines for eutrophication control, restoring water quality "to a level of lower and more acceptable trophic conditions" [OECD, 1982 p.96]. The standard OECD equations for the relationships between mean inflow phosphorus, expected mean lake phosphorus and expected mean chlorophyll *a* concentrations, as a function of average water residence time, were synthesised to produce a diagram for calculating phosphorus loads necessary to achieve preset conditions (Fig.3.2). The diagram was designed as a rapid and crude means by which objectives may be assessed, by predicting the resultant trophic category.

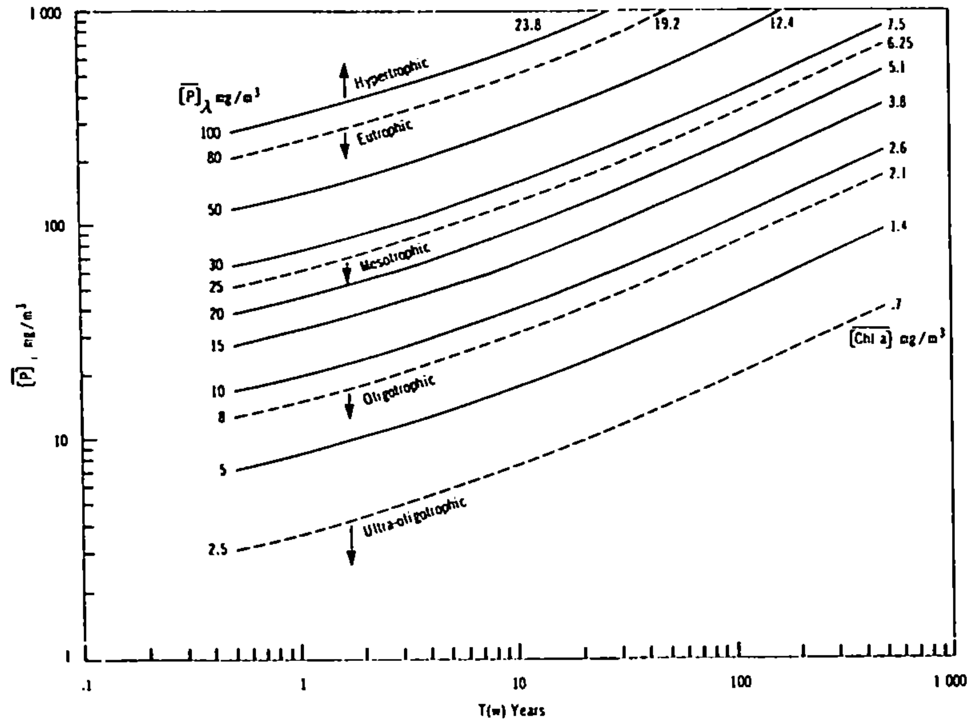

Fig.3.2: Synthesis of Standard OECD Equations for the Assessment of Lake Management Objectives

OECD [1982]

Verification of the idea that eutrophication was reversible led to the implementation of phosphorus control programmes in the majority of countries in Europe and North America. In qualitative terms, many of the lakes receiving reduced phosphorus input have responded in accordance with the predictions of the model by exhibiting improved water quality [Sas, 1989]. However, in quantitative terms, accordance has been verified for very few sites. Additionally, in several cases, reductions in phosphorus inputs have led to delayed or only slight improvements in water quality.

The lack of response to phosphorus load reduction observed in some restoration programmes led to an evaluation of recovery for eighteen eutrophicated lakes located in Western Europe [Sas, 1989]. The Vollenweider-OECD model provided the reference for recovery evaluation, but despite its advantages, it was felt that there were some critical points in its application:

(a) The model is based upon the assumption that regression analyses from a large number of lakes in different steady state conditions may be used to predict the effect of phosphorus load reduction on a particular lake. It was felt that this assumption was in need of validation

(b) The ease with which the model can be applied without sufficient knowledge of factors peculiar to a particular lake may lead to the instigation of management programmes which may not achieve the desired objectives

(c) Significant doubts about the practical application of the model have been raised because of the relatively wide confidence limits for predictions

(d) The assumption of a steady state suggests that the model will not predict the transient phase after a restoration programme begins, in which a delay in response may occur

(e) Changes in the composition of the algal community are not accounted for in the model

With respect to the reasons for delayed or disappointing recovery following restoration programmes, the investigation concluded that the following mechanisms operate [Sas, 1989]:

(a) The flushing rate of a lake determines the degree to which the 'old stock' of phosphorus is removed from the system

(b) In shallow lakes (those in which most of the epilimnion is regularly in contact with the sediment) net annual release of phosphorus from the sediment often occurs in the first years after restoration

(c) In lakes receiving very high phosphorus loadings algal biomass may not be limited by this element, so that an initial reduction in in-lake concentrations will have no effect

(d) When in-lake phosphorus concentrations are reduced to the point where this element becomes limiting, algal response is characterised either by overall biomass reduction or by migration to deeper water

In relating these observations to the ability of the Vollenweider-OECD model to predict them, Sas [1989] concluded that the first mechanism, associated with flushing, was easily predicted. The second, that of 'sediment memory', was outside the scope of the model and therefore not predicted by it. With respect to the effects of phosphorus loading on algal biomass, the model appeared to predict response reasonably well.

A more recent critique of the Vollenweider-OECD model is presented by Johnes, Moss and Phillips [1994]. The authors emphasise the degree of variation exhibited by lakes and the continuous nature of the trophic spectrum. They decry the persistence of the notion that distinct lake types exist. It is true that spatial state schemes, such as that conceived by the OECD, are limited by the fact that they do not reflect changes over time, and that naturally nutrient-rich lakes are placed in the same category as eutrophicated ones. In these respects, state-changed schemes, such as that devised by the authors, possess a distinct advantage.

The OECD results and, in particular, the diagnostic diagrams produced by Vollenweider, do provide the impression that the boundary lines between 'dangerous' and 'permissible' loadings may be used as targets for nutrient control. This approach was employed in the work of O'Sullivan [1992; 1993] and Wilson, Gibson and O'Sullivan [1993], but only in the absence of alternative objectives based on scientific knowledge of previous trophic status. The OECD report [1982] advises managers to "establish a water quality objective (expected trophic response) taking into consideration the intended use of the water and the natural trophic conditions of the area" [*ibid.* p.98].

Following this advice, water quality objectives may be established merely by deciding upon the appropriate trophic category for a particular lake. Such an approach may be adequate in cases where the priority is to achieve water quality suitable for various human uses. In order to define realistic and appropriate objectives for ecological purposes it may be necessary to obtain more specific information, such as data on pre-eutrophication levels of phosphorus inputs. Such data is, however, largely unavailable for the majority of lakes in this country, hence adoption of the export coefficient approach by Johnes, Moss and Phillips [1994] in assessing degrees of eutrophication. As outlined above, this approach is not without inherent difficulties, and it may be that more direct, and more expensive, methods of ascertaining lake productivity changes are necessary, namely the use of palaeoecological interpretation.

In conclusion, the Vollenweider-OECD model possesses various advantages and limitations which may be summarised thus:

(a) As a steady state model, it cannot be used to assess past nutrient enrichment

(b) The model is capable of predicting trophic and algal biomass responses to loading reductions reasonably well, except in cases where internal loadings become significant during the transient phase after restoration.

(c) In order to take full advantage of the model's predictive ability, knowledge of a lake's natural trophic condition is required.

(d) In the absence of an alternative tool, it remains the most readily available and widely applicable model of its kind.

Chapter 4

Legislative Framework for Conservation and Restoration

There is no UK legislation which directly addresses the causes and effects of eutrophication. From a regulatory viewpoint, the main problems involved in dealing with eutrophication lie in the often diffuse and multiple nature of sources of nutrients, and in the distances, which may be considerable, between such sources and the surface waters they affect. The current regulatory system carries the potential to work comparatively well for point sources of nutrients, but it appears ineffective where losses are the cumulative effect of a large number of activities.

In the development of strategies to control eutrophication, and to conserve the wildlife of affected lakes and reservoirs, it is necessary to draw upon a range of laws which deal with different aspects of the problem. Legislation most relevant to eutrophication of freshwater Sites of Special Scientific Interest (SSSIs) may be categorised as that relating to nature conservation, to agriculture and to water quality.

The development of nature conservation objectives is inextricably linked to the history of agricultural intensification, but in legislative terms, until quite recently, these topics have been approached separately. The relevance of water quality legislation to the problem of eutrophication is reasonably obvious, but a less direct relationship with nature conservation in general has also been established. A number of other areas of legislation are also of interest, most notably laws concerned with the planning system, but these will be discussed within the three main subject areas. The information on legislation in this Chapter is current as of 31st March 1995.

4.1 Sources of Legislation

Legislation related to nature conservation, to agriculture and to water quality, operate within a framework which includes British, European Community (EC) and international laws and policies. Each of these sources of legislation tend to operate in different ways and some explanation is necessary, prior to assessment of individual laws.

4.1.1 British Legislation

British legislation, in environmental areas, tends to consist of statutory (as opposed to judge-made) law [Ball and Bell, 1994]. Statutes are often framework Acts, which require delegated legislation, such as regulations, in order to be effective. Delegated legislation, in most cases, is made by the Secretary of State or another member of the government, who is often granted very wide powers to interpret an Act. Statutes may require different regulations (Statutory Instruments) to be laid before parliament prior to implementation of separate parts. This sometimes means that legislation is implemented in a piecemeal and gradual manner.

Definitions in statutes, including those of fundamental concepts such as 'development', are often left deliberately unclear, in order to preserve flexibility in the application of the law [Ball and Bell, 1994]. Clarification is usually provided in the form of central government Guidance Notes and Circulars to the appropriate enforcing body. Such documents possess no formal legal standing, yet may be used to effect far-reaching changes in the application of the law. Wide powers of discretion are often available to enforcement agencies, as well as central government. Indeed, discretionary decision-making is a common feature of most British environmental law. The combined effect of delegated legislation and the use of policy documents is that the definition of what may be considered as law is decidedly wider than the original statute.

Given the discretionary nature of the application and enforcement of so much of the law, central government policy is crucial in determining the balance between environmental concerns and other conflicting interests [Reid, 1994]. In England, the Department of the Environment (DoE) is the office responsible for nature conservation, planning and pollution control. Apart from its considerable influence in terms of policy implementation, the DoE (with the Treasury) is also accountable for the budgets of a number of regulatory agencies, and may exercise its power through financial controls.

4.1.2 EC Legislation

The EC is an increasingly important source of environmental law and policy enforceable in Britain, much of which has been welcomed by those concerned with environmental protection. It is also responsible, however, for many of the economic policies, especially in relation to agriculture, which themselves operate counter to the realisation of certain environmental objectives. An awareness of the influence of the EC on British law is therefore integral to an appreciation of the complexities involved in addressing the problem of eutrophication through legislation. The following overview is by no means complete in its coverage of EC legislation, but rather provides a preface to particular laws relevant to this study.

The European Communities Act 1972 gave constitutional force to Britain's membership of the then EEC, under the Treaty of Rome, and provided for the recognition of EEC legislation as domestic law [Ball and Bell, 1994]. The signing of the Treaty created not simply a free trade agreement, but a form of supranational state, with the power to establish its own laws. It was not, however, until 1987, when the Single European Act came into force, that the Community was given full authority to legislate in the environmental field. In 1992, the Maastricht Agreement altered the formal title of the EC to the European Union (EU), but the name of the Treaty itself, which is effectively the constitution, was changed from EEC to EC. Laws created by the EU are therefore EC legislation.

The majority of EC legislation comes in the form of Regulations or Directives [Reid, 1994]. Regulations take direct effect, without the need for implementation of national laws. Directives, on the other hand, require Member States to introduce legislation which meets the objectives of the Directive, within a specified period. The introduction of relevant national legislation constitutes formal compliance, but increasingly, actual compliance is monitored [Ball and Bell, 1994]. There are a number of ways in which a Member State may be in breach of a Directive:

(a) failure to implement within the stated time
(b) failure to implement all the requirements
(c) by adopting an incorrect interpretation
(d) by implementing through changes in practice, rather than by law
(e) inadequate enforcement

If full or proper implementation is not achieved by a Member State, the European Commission may initiate infringement proceedings, which can lead eventually to an action before the European Court of Justice.

Although the original aims of the EC were economic, the need for policy on the protection of the environment was accepted by the early 1970s [Ball and Bell, 1994]. The first Action Programme on the Environment in 1972 has been followed by a further four, until the Maastricht Agreement in 1992, which recognised for the first time that the development of a policy on the environment was one of the main functions of the EC. The four central principles of EC environmental policy, which are taken into account in formulating legislation, are:

(a) preventative action should be preferred to remedial action
(b) environmental damage should be rectified at source
(c) the polluter should pay for the cost of measures to protect the environment
(d) environmental policies should be integrated into other EC policies

As part of the fourth principle, the EC aims to examine the environmental implications of various sectors of the economy, including agriculture and industry. In relation to eutrophication, the development of these principles into legislation is essential, in order to address the problem in a comprehensive manner.

4.1.3 International Legislation

International environmental law generally consists of Conventions recognised by signatory States [Ball and Bell, 1994]. The treaty-making procedure involves the negotiation of a text agreed, increasingly, by consensus. This must then be ratified by the participants, in order to ensure that the legal, financial and administrative mechanisms are in place prior to implementation [Freestone, 1994]. Such laws are not strictly legally binding, because of the lack of sanctions available for non-compliance. However, the failure of a signatory to implement a Convention may well result in loss of credibility and political embarrassment. International Conventions are normally implemented through domestic legislation.

4.2 Nature Conservation

Within legislation relating to nature conservation many different aspects of human interaction with the countryside are addressed, and only those most relevant to this study will be examined. An understanding of the development of this legislation is necessary in order to appreciate the historical framework within which the current system operates, and the consequent constraints placed upon it in dealing with such a multi-faceted problem as the eutrophication of freshwater SSSIs.

4.2.1 The Development of Legislation for Nature Conservation

4.2.1.1 The Post War Era

In Britain, voluntary movements for the conservation of nature began in the Victorian era, but it was not until after the second world war that political commitment towards wildlife conservation was implemented [MacEwen and MacEwen, 1982]. The demand for 'a better Britain' and the election in 1945 of a Labour government engendered what was then the most comprehensive planning and social welfare system in the world. An integral part of the vision for a new Britain was the introduction of the National Parks and Access to the Countryside Act 1949. As might be expected, such innovative legislation possessed shortcomings, many of which have survived subsequent attempts to amend the law in this area. In the context of this study, several issues are of particular interest and have their roots in the history of nature conservation in this country.

During the 1920s and 1930s rural preservation became part of the political debate, with lobbying of parliament by groups representing apparently conflicting interests [Lowe *et al.*, 1986]. An influential body of opinion maintained that a secure and revitalised farming industry was the means by which both the economic viability and the natural beauty of the countryside could be conserved. Other lobbyists were not convinced by this argument, but were themselves rather disparate in their attitudes to the countryside. Two main lobby groups emerged: naturalists who, influenced by the new science of ecology, saw the need to protect habitats in order to preserve species, and those whose prime interest was access to the open countryside. This divergence between the scientific and the recreational approach to the countryside was strongly reflected in the 1949 Act, with the conservation of 'flora and fauna' being assigned to the Nature Conservancy, while responsibility for 'landscape' became the remit of the National Parks (later Countryside) Commission [MacEwen and MacEwen, 1982]. Thus was created "the great divide" [*ibid.* p.16], the political and scientific validity of which is highly questionable.

Under the Act, the role of the Nature Conservancy (NC) was to provide advice to government on the conservation of natural fauna and flora, based on scientific research [MacEwen and MacEwen, 1982]. It was given the ability to create National Nature Reserves, strengthened by powers to enter into management agreements with landowners, and to purchase land (compulsorily if necessary) for this purpose. The NC could also designate SSSIs, but at that time there was no statutory protection for such areas. The Commission, in contrast, was given no landowning or land management functions. Its role was simply to designate and supervise National Parks and Areas of Outstanding Natural Beauty. It is this artificial division between landscape and flora/fauna which many consider to be a severe limitation to an integrated approach to nature conservation.

Further aspects of the 1949 Act set precedents for the future. One of these was its emphasis upon designated areas of nature or landscape conservation, outside of which destructive practices could continue [Lowe *et al.*, 1986]. It was assumed that the effects of agricultural development would be benign. Urban and industrial pressures, perceived as the main threats to landscape and wildlife, were controlled under the Town and Country Planning Act 1947. Agriculture and forestry, on the other hand, were generally exempt from planning restrictions, and, under the complimentary Agriculture Act 1947, a secure environment for investing in farming was provided.

4.2.1.2 Developments During the 1960s and 1970s

Few people anticipated that agricultural intensification would transform the rural environment [Lowe *et al.*, 1986]. Intensification and mechanisation during the post-war era led to widespread loss of hedgerows, planting of ancient woodlands with conifers, ploughing-up of moorland, lowland heath, downland and old pasture, and drainage of lowland fens, mires and valleys. In terms of designated areas, loss of, and damage to, SSSIs become a serious problem, largely as a result of the lack of planning controls on agricultural development. Yet so strong was the romantic view of farming that reactions to changes in the countryside were slow and piecemeal.

During the early 1960s concern grew among conservationists, to the extent that demands were made for a much stronger body to oversee not only national parks, but also the rest of the countryside [Lowe *et al.*, 1986]. New legislation, in the form of the Countryside

Act 1968, reorganised the National Parks Commission as the Countryside Commission (CC), with new powers to promote research and to recommend grant aid throughout the whole of the countryside [Green, 1981]. Local authorities were given powers, including compulsory purchase, to establish Country Parks, a new category of protected area. These two aspects of the 1968 Act were positive steps, but, in essence, this piece of legislation is notable for what it did *not* achieve [Lowe *et al.*, 1986].

Although the NC was empowered to make management agreements and payments to the owners of SSSIs, only small numbers of sites were protected in this way [Green, 1981]. An attempt to include provision in the Act to bring SSSIs under planning control was unsuccessful. Instead, Ministers, government departments and public bodies were instructed to "have regard to the desirability of conserving the natural beauty and amenity of the countryside" [*ibid.* p.208]. Conversely, the NC was to "have *due* regard to the needs of agriculture and forestry" [*ibid.*; author's emphasis]. This reciprocal duty cancelled out any gain to nature conservation, with the result that agriculture and forestry continued to cause extensive damage to the countryside.

During this period, the status and credibility of the NC declined [Evans, 1992]. It had been subsumed into the Natural Environment Research Council (NERC) in 1965 and, although its duties remained the same, its funds were inadequate to buy reserves or to pay compensation to owners of SSSIs. The government was eventually forced to restore some integrity to its official conservation agency and, in 1973, passed the Nature Conservancy Council Act. This separated the Nature Conservancy Council (NCC), under the DoE, from the Institute of Terrestrial Ecology, which remained with NERC.

Revitalisation of the NCC following reorganisation was evident in the publication, in 1977, of *A Nature Conservation Review: the Selection of Biological Sites of National Importance to Nature Conservation in Britain*, under the editorship of Derek Ratcliffe [Evans, 1992]. The *Review* listed 735 key sites as the basis for a series of nature reserves, of which 99 were open water. Sites were selected according to the criteria of size, diversity, naturalness, rarity, fragility and typicalness, as well as their intrinsic appeal and potential value for nature conservation. The sites were graded on a scale of 1 to 4, with Grade 1 intended for NNR designation, but all were to be afforded SSSI status.

A further sign of assertiveness within the NCC appeared in the publication of *Nature Conservation and Agriculture* in 1977, which concluded that "all changes due to modernisation are harmful to wildlife except for a few species that are able to adapt to the new simplified habitats" [Lowe *et al.*, 1986 p.28]. As well as a growing awareness among conservationists of the nature and extent of damage to the countryside by agricultural intensification, the strategy of designating areas for conservation was questioned, not only because of its ineffectiveness, but also for ignoring the wider countryside. Throughout the 1970s the demands of conservation groups for powers to regulate the impact of agricultural and forestry development repeatedly confronted a powerful set of interests in the farming and landowning lobby. The National Farmers Union (NFU) and the Country Landowners Association (CLA) together represented a powerful force in British politics, and exerted an influence disproportionate to the size of their membership.

4.2.1.3 The Wildlife and Countryside Act 1981

The one episode in British politics which served to highlight and polarise the conflict between agriculture and conservation was the preparation and passage of the Wildlife and Countryside Act 1981 [Lowe *et al.*, 1986]. The passage of the Bill through its public consultation period and parliamentary phase took well over two years, and generated considerable debate. A total of 2300 amendments were tabled, which reflected the intensity of lobbying. This long and complicated process has been well documented by several authors [e.g. Cox and Lowe, 1983; Adams, 1984] and will not be described in detail here.

The NFU and CLA emphasised their commitment to 'goodwill and voluntary means', rather than formal constraints. Most of the conservation lobby adopted a 'cooperation plus safeguards' position, accepting the 'voluntary goodwill' approach but arguing the need for reserve powers to be used against those farmers unwilling to cooperate. They also supported the idea of conservation incentives and reform of the agricultural support system. Others, however, highlighted what they saw as a fundamental conflict of interest between agriculture and conservation, and advocated that farming operations should be brought completely under the control of planning regulations.

During the later stages of the parliamentary phase several factors combined to pressure the government into introducing concessions to the conservation lobby, including the need to comply with the EC Directive on the Conservation of Wild Birds [Lowe *et al.*, 1986]. By far the most important gain came in an amendment which required owners or occupiers of SSSIs to give three month's notice to the NCC of any intention to carry out proscribed potentially damaging operations (PDOs). For the first time, the NCC were to be given a say in agricultural land use planning and a much-needed mechanism to monitor SSSI loss. However, during the three month consultation period following renotification, owners and occupiers, although urged by the DoE to desist voluntarily from PDOs, were legally free to damage the site before the process was complete [Evans, 1992]. The scale of damage to SSSIs in the years following the 1981 Act were unprecedented, to the extent that by March 1985 about 10% of existing sites were expected to be denotified. The Act also required that all 3,800 SSSIs which had been designated up to that point were to be renotified to the owners and occupiers of the land. This was by no means a minor undertaking and was not completed until the end of 1991 [Francis, 1994].

Although the 1981 Act provided delaying powers and the scope for negotiation over the issue of PDOs, it became clear that a revision of the Act was necessary [Lowe *et al.*, 1986]. The resultant Wildlife and Countryside (Amendment) Act 1985 protected SSSIs from PDOs during the three month consultation period following (re)notification. It also extended from three to four months the period for negotiation following notification by an owner or occupier of their intention to carry out a PDO. In this way, two important 'loopholes' in the original Act were rectified.

In 1990, the government introduced the Environmental Protection Act (EPA), which addressed itself mainly to issues of pollution control and waste disposal [Evans, 1992]. One measure of interest to this discussion was the decentralisation of the NCC into separate bodies for England, Scotland and Wales. Three new organisations were created: the Nature Conservancy Council for England (English Nature: EN), the Nature Conservancy Council for Scotland (NCCS) and the Countryside Council for Wales (CCW). From its inception, the CCW acquired an amalgamation of NCC and CC duties, whereas it took another year before Scottish Natural Heritage (SNH) was created through the merger of NCCS and the Countryside Commission for Scotland [Boon, 1994]. In

England, however, the NCC and CC remain separate for the present. Although this belated integration of scientific and amenity interests in Wales and Scotland might be perceived as a positive measure, it received vehement opposition from various quarters [Ball and Bell, 1994]. The main objection was based on the perception that it weakened the effectiveness of the original bodies:

> "The clear impression that is left is that the main motivation behind the splitting up of the NCC was the political desirability of reducing the NCC's power in Scotland, where it had been active in opposing such things as afforestation of the unique flow country of Caithness and Sutherland." [*ibid.*, p.407]

Other objections raised were that the division of responsibility for nature conservation along national boundaries was illogical, and that it would inevitably lead to a lack of uniformity in the application of the law. The government's response to these criticisms was to establish the Joint Nature Conservation Committee (JNCC), with responsibility for advising on international laws and agreements, and for retaining common standards.

A further aspect of the 1990 Act which may produce benefits for vulnerable sites was an amendment to the Countryside Act 1968 [Ball and Bell, 1994]. This empowered EN to enter into management agreements on land adjacent to SSSIs for purposes connected with conserving the special interest of the SSSI. Such agreements may be particularly useful for wetland sites, in order to control drainage or nutrient levels.

4.2.1.4 The Present System

The legislation discussed above represents the development of the current system of nature conservation. The designation of SSSIs remains the key element upon which the system is based. It is necessary, therefore, to explain the means by which SSSIs are selected and designated, and to relate these to standing waters.

An SSSI is an area of land (or inland water) of special interest by reason of any of its flora, fauna, geological or physiographical features [Ball and Bell, 1994]. Biological SSSIs are selected according to criteria and standards set out in guidelines published by the NCC [1989b]. These build on those developed in *Nature Conservation Review*

[Ratcliffe, 1977], and provide a degree of standardisation to the process. A prime objective of the guidelines is to enable the identification of sufficient habitats to support viable populations of most wildlife species present in Britain [NCC, 1989b]. Within each Area of Search (AOS), usually a county or subdivision of it, all the different habitats and species present will be represented by at least one example or population. Generally, as rarity increases, so does the proportion of the area or population qualifying for selection. The primary criteria by which individual sites are assessed are those of size, diversity, naturalness, rarity, fragility and typicalness. Recorded history, position in an ecological/geographical unit, potential value and intrinsic appeal serve as secondary criteria.

For standing waters, site selection first requires classification according both to open water and to emergent vegetation, so that each type may be represented within the AOS [NCC, 1989b]. Once classified, selection is based on further criteria which include species richness, presence of a range of *Potamogeton* species, diversity of physical features, rarity of site type, paleolimnological and geological attributes, presence of an ecological series, and naturalness of the catchment.

The question of SSSI boundary definition often poses problems, particularly for vulnerable sites which may require surrounding buffer land in order to ensure full protection. The guidelines state:

> "Clearly, in relation to SSSIs, there can be no other category of protected land with legal status. Either surrounding land is regarded... as being of sufficient interest to be included within the site, or it is not" [NCC, 1989 p.34]

In the case of standing water SSSIs, it is well recognised that land use changes within the catchment may affect the special interest of the site. Consequently, where a catchment is still natural or semi-natural, and especially where it is small and discrete, the guidelines recommend that the whole of it should be included within the SSSI. Alternatively, a buffer zone around the water must be incorporated, which may consist of a grass field's width or the area defined by the first break of slope. A minimum width of 5m is recommended, or 50m where there is a threat of afforestation. In practice, land of lesser intrinsic scientific interest may be designated if it is part of the same environmental unit as that which is of interest, but the legal status of surrounding buffer zones is dubious [Ball and Bell, 1994]. A possible resolution to the problem of buffer zones around SSSIs, which will be particularly applicable to standing waters, is in the process of being adopted, and will be discussed below.

Once a site qualifies as an SSSI under the above criteria, EN must formally notify its existence and interest to the owners and occupiers of the land, to the Secretary of State, and to the local planning authority [Ball and Bell, 1994]. EN must also specify to landowners a list of PDOs to the SSSI. These are procedures which, in the opinion of EN, are likely to damage those features for which the land is of special interest. EN is required to consider any representation or objection during a three month consultation period. Once notified, owners and occupiers are placed under a reciprocal duty to give four months notice to EN of their intention to carry out a PDO, enabling consideration and consultation of the proposed operation. In those cases where EN objects to the carrying out of PDOs, it may (and in certain cases, must) offer the landowner a management agreement which could contain financial recompense in exchange for restrictions to safeguard the interests of the SSSI. Agreements may also include arrangements for positive management of the site, since its value may be lost through neglect, without any PDO being infringed. The mechanisms involved in the PDO process are illustrated in Fig.4.1.

Stronger powers are available through the use of Nature Conservation Orders (NCOs) to ensure the survival in Britain of any kind of animal or plant, to comply with an international agreement, or where the land is of national importance [Ball and Bell, 1994]. NCOs come into effect immediately, and can be used for urgent protection of an important area pending the formal notification as an SSSI. They may also be employed to provide EN with more time to negotiate a management agreement, since notification of this process imposes a twelve month ban on PDOs. The Secretary of State is authorised to confirm, amend or revoke an NCO, but these powers have been sparingly used, with only about twenty sites in England protected in this way since 1981 [EN, 1994]. As a last resort, EN possesses powers to seek approval for a compulsory purchase order from the Secretary of State [Ball and Bell, 1994]. If EN is unable to conclude an acceptable management agreement, and the owner refuses to sell the land, compulsory purchase is the only recourse available. These powers are, however, very rarely used.

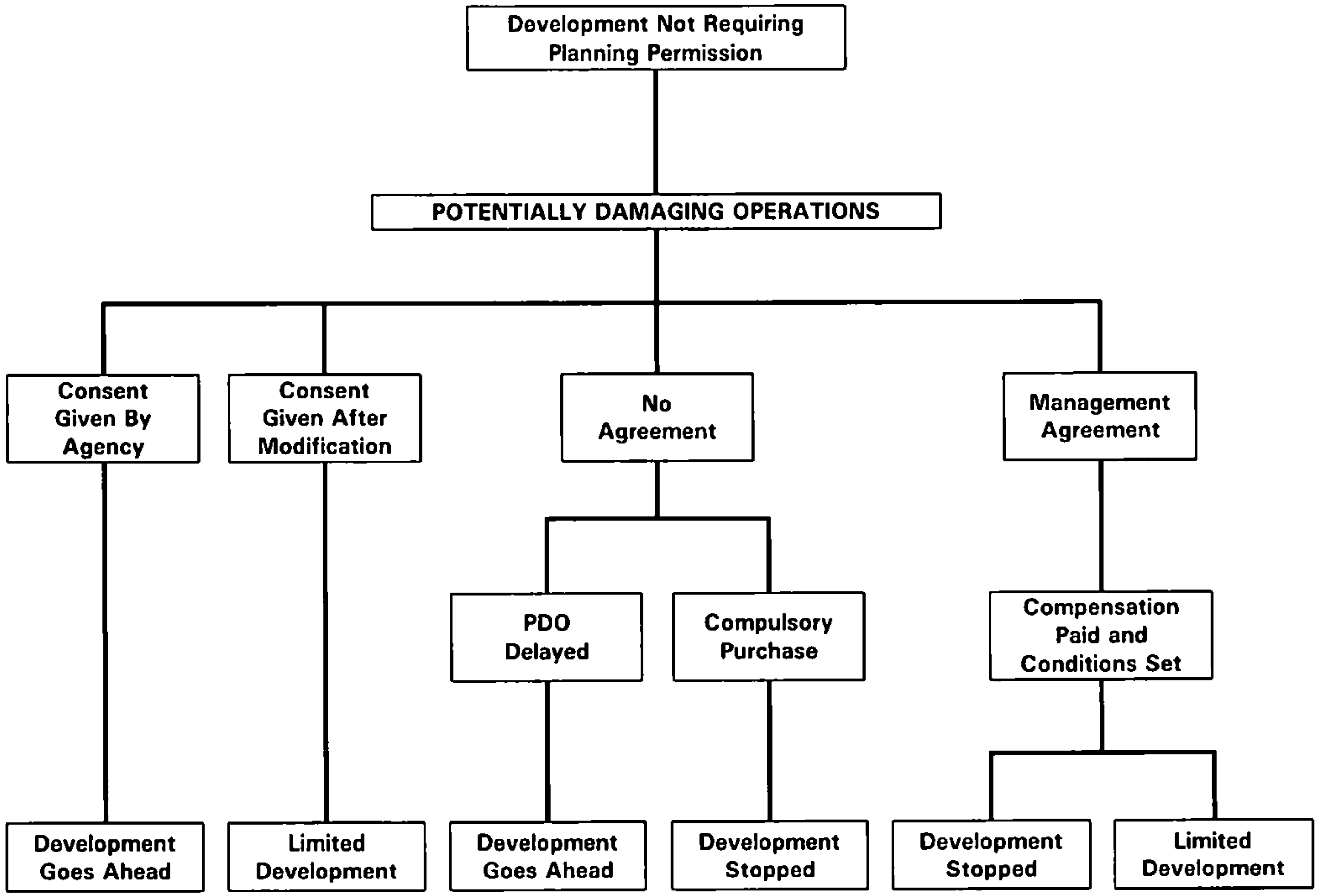

Fig.4.1: The PDO-Management Agreement Mechanisms

Based on Spash and Simpson [1994]

A number of other pieces of legislation impose specific duties in relation to SSSIs. Under the Water Resources Act 1991 and the Water Industry Act 1991, EN must notify such bodies as the NRA, water and sewage undertakers, and internal drainage boards of SSSIs which may be affected by their operations [Ball and Bell, 1994]. These bodies, once notified, must consult EN over any activity they intend to carry out which it thinks is likely to damage or destroy the site. The NRA is obliged to confer with EN before granting abstraction licences, and discharge or land drainage consents which it thinks are likely to damage an SSSI. These duties relate to all operations, not simply to PDOs, and apply to activities in the vicinity of an SSSI. In the case of freshwater SSSIs, this aspect is particularly pertinent, since activities to which it is applicable include upstream discharges, such as those from sewage treatment. If the relevant body fails to consult EN there is, however, no recourse unless EN brings a successful action for judicial review.

Under the Town and Country Planning (General Development) Order 1988, where an application for planning permission relates to, or may affect, an SSSI, the local planning authority (LPA) is required to consult EN before making a decision [Ball and Bell, 1994]. If planning permission is granted against their advice, EN may consider whether to ask the Secretary of State to call-in the application. In general, cases are only called in if, in the opinion of the Secretary of State, issues of more than local importance are involved. It may be asserted that all SSSIs are of national importance, but this is not the view normally taken by the Secretary of State.

Many activities which may damage SSSIs do not, however, require planning permission, either because they are not considered development or because they are exempt [Ball and Bell, 1994]. Automatic planning permission, or 'permitted development right' (PDR), is granted for a wide range of activities. These include developments of a minor or temporary nature, those carried out by a wide range of public services, such as for water, gas and electricity, and those of favoured activities, especially agriculture and forestry. The Order does contain some general restrictions. Additionally, LPAs are able withdraw PDRs for certain types of development or for specific sites, although government policy is that such powers should be used sparingly.

Outside of activities granted PDRs, the extent to which LPAs are able to protect nature conservation interests depends on policies adopted in their development plans [Payne, 1994]. LPAs are required, under the Planning and Compensation Act 1991, to include policies on the conservation of the natural beauty and amenity of the land, and are advised by the DoE that plans should endeavour to sustain the character and diversity of the countryside, including wildlife habitats. Both structure and local plans employ a range of policies for this purpose.

> "Most development plans will include a presumption against development which might adversely affect designated sites. Some development plans will make clear that the presumption against development may arise both where the effect is direct and indirect". [*ibid.* p.981]

Planning applications for certain types of development may also be the subject of an environmental assessment (EA), under the Town and Country Planning (Assessment of Environmental Effects) Regulations 1988 [Ball and Bell, 1994]. These Regulations implement EC Directive 85/337, and apply to two categories of development. 'Schedule one projects' require an EA in every case, whereas for 'Schedule two projects' an EA is necessary only if they are judged likely to give rise to significant environmental effects. The criteria used to assess the necessity for an EA include proximity to a sensitive location, such as an SSSI. An environmental statement, prepared by the developer, must contain certain specified information, including a description of the likely effects, direct or indirect, on flora and fauna, soil, water, and the landscape. In compiling such details developers are required to consult statutory bodies, including EN.

In March 1991, as a result of concern expressed over damage to SSSIs, the DoE issued a consultation paper on proposals for additional planning safeguards [DoE, 1991a]. A number of changes were proposed including withdrawal of PDRs for certain temporary recreational activities such as war games, motorsports and clay pigeon shooting. Monitoring of SSSIs had shown that such uses caused more damage than any other type of PDR. In September 1991, the Government decided to implement this safeguard, by amending the General Development Order [DoE, 1991c]. It was emphasised, however, that the withdrawal of PDRs was not to be taken as a presumption against these activities, but to enable nature conservation interests to be taken properly into account.

The consultation paper contained other proposals related to changes in the planning system. These are necessary for compliance with the EC Birds Directive 79/409 and the

EC Habitats Directive 92/43, which will be outlined in the next Sections. It is necessary at this point to emphasise that, in order to qualify for national or international designation, a site must already warrant SSSI status [Reid, 1994]. The degree of protection afforded to SSSIs is not, however, sufficient to conform with that required by the above Directives. Certain alterations to planning legislation were therefore necessary, in order to ensure greater protection. The measures introduced are detailed in the DoE's *Planning Policy Guidance: Nature Conservation* [1994], known as PPG9. Those aspects which affect only sites of international status will be discussed with the appropriate legislation.

As part of the revision of planning policy in relation to nature conservation, the DoE also introduced a new requirement for consultation on planning applications for development adjacent to SSSIs [DoE, 1994]. Previously, LPAs were advised to confer with EN about applications which may affect SSSIs indirectly, but the decision to consult lay with the LPA. In the light of the above Directives, it became clear that this situation required clarification. EN has therefore been requested by government to define consultation areas around all SSSIs, but giving priority to those for sites of international importance. A consultation area would not normally extend beyond about 500m from the boundary of an SSSI, although for areas such as wetlands it may extend as far as 2km upstream. The boundaries of such areas must be notified to LPAs by EN, who may also request consultation about certain types of development beyond the 2km maximum.

In the consultation paper, the government's position on the issue of presumption against development in SSSIs was stated:

> "The Government has considered the possibility of introducing a general presumption against development in SSSIs,... but it does not propose to do so. It considers that the measures proposed... will ensure that nature conservation interests are taken into account. It is also mindful that notification as an SSSI is a purely scientific statement of the special wildlife and geological interest of the site and is not the only factor to be taken into account in considering planning applications. A presumption against development in SSSIs might suggest that notification as an SSSI interferes with the rights of use of the landowner/occupier to the point where the land is devalued. This could imply radical changes in the procedures for designating SSSIs, including formal rights of objection for owners and occupiers". [DoE, 1991a p.9].

The legal and political ramifications of a presumption against development in SSSIs would be significant. Nevertheless, the implications of EC legislation to protect internationally important sites may necessitate the adoption of a policy against granting planning permission in these areas [Ball and Bell, 1994].

Despite periodic modifications in legislation relating to SSSIs, damage and loss still occur at a significant level. Although agricultural activities still account for a large proportion of incidents, these seldom lead to complete loss of a site. Damage caused by agriculture is often the result of poor management, such as overgrazing or neglect, which is difficult to control constructively [Ball and Bell, 1994]. Long-term damage and loss are more usually attributable to operations which are effectively outside the remit of the 1981 Act. These include activities granted planning permission, often against the advice of EN, major government developments, such as road-building, operations by statutory undertakers, and land rights existing prior to notification, such as for peat-cutting. The extent of the problem in England for 1993/94 is shown in Table 4.1.

Table 4.1: Damage to Sites of Special Scientific Interest in England: 1st April 1993 to 31st March 1994

Source: English Nature [1994]

Activities Causing Damage to Special Interest	Loss		Damage	
	Complete	**Partial**	**Long-Term**	**Short-Term**
Agriculture	0	1	9	24
Forestry	0	0	1	1
Planning Permission	0	5	0	2
Statutory Undertakers	0	2	7	5
Recreation	0	0	4	14
Insufficient Management	0	0	6	19
Miscellaneous	0	0	5	31
Total Cases	0	8	29	87

A further limitation on the level of protection afforded SSSIs is that only the activities of owners and occupiers are restricted, and not those of other individuals or bodies

[Reid, 1994]. A recent, notorious case which illustrates this point is that of *Southern Water Authority* v *Nature Conservancy Council* [1992] 3 All ER 481. A successful case was brought against the Authority for dredging part of Alverstone Marshes SSSI without EN's consent. On appeal, it was ruled that, even though the Authority knew its action constituted a PDO which would cause significant harm, it did not commit an offence, since it was neither an owner nor an occupier of the land.

The role of biological SSSIs is primarily for the protection and enhancement of habitats for the preservation of species diversity. Conservation of landscape and amenity interests is served by a number of designations, most notably those of National Parks (NPs), Areas of Outstanding Natural Beauty (AONBs) and Environmentally Sensitive Areas (ESAs). The effect of such designations on nature conservation is variable, but, in general, controls on development in the interests of preserving the landscape are likely to enhance the quality of habitat [Reid, 1994]. Agricultural and forestry developments are, however, mostly exempt from planning controls in NPs and AONBs. The aim of ESAs, on the other hand, is to support agricultural practices which are conducive to the conservation of natural habitats, through the availability of a payment scheme. Designation of ESAs is, therefore, a potentially useful tool for encouraging environmentally beneficial farming, and will be discussed in Section 4.3.3.1.

Although many of the features of the 1949 Act remain as the basis for protection of designated sites, the growth of environmental awareness in latter years has produced a shift in emphasis towards a concern for the wider countryside [Ball and Bell, 1994]. Nature conservation interests are increasingly integrated into legislation and policy related not only to rural issues, but also to urban. As noted above, for developments which require planning permission, nature conservation is now a 'material consideration'. LPAs have also been granted considerable scope for including specific conservation objectives in their development plans. These aspects of the planning system offer some potentially significant benefits for nature conservation in the wider countryside. Certain aspects of the 1949 Act remain unaltered, however, despite subsequent attempts to amend the law in this area.

A central tenet underlying the implementation of nature conservation legislation is the voluntary principle, under which incentives are offered to owners and occupiers in order to encourage them to become willing partners in the conservation of designated land [Reid, 1994]. The voluntary principle reflects the reluctance of the law to restrict the rights of landowners to use their property as they see fit. Critics maintain that the property rights of individuals are given undue weight at the expense of public benefits. Supporters insist that successful conservation requires the continuing cooperation of owners and occupiers, which can only be achieved by voluntary agreement. In other words:

> "The voluntary principle... is two-sided: whereas, negatively, it is an ideological defence of the autonomy of farmers and landowners; positively, it is about encouraging a social ethic concerning stewardship of the countryside" [Cox, Lowe and Winter, 1990 p.176]

Adherence to the voluntary principle is also an indication of the position of nature conservation in political thinking, in that it is seen as but one interest among many which must be considered in disputes over land use [Reid, 1994]. Admittedly, nature conservation is currently granted more recognition as one of these interests than ever before, but in very few circumstances is its status sufficient to warrant absolute priority. This is reflected in the assertion of the DoE, cited above, that "notification as an SSSI is a *purely scientific* statement" [DoE, 1991a p.9; author's emphasis]. In essence, this implies that nature conservation merits no political, social or economic weighting in being assessed alongside other interests, and that only 'science' will benefit from the protection of SSSIs. On the basis of such statements, it is likely that the voluntary principle will remain central to the government's approach to nature conservation. However, the EC Directive on Habitats will provide a rigorous standard against which to test the effectiveness of British law and may require stronger protection for designated sites than is possible under the voluntary principle [Reid, 1994].

4.2.2 EC Directives

Given the status of EC legislation in British law (Section 4.1.2), the legal impact of EC designation of important nature conservation sites may be considerable [Reid, 1994]. Legal obligations at the European level have, in the past, helped to form domestic law, and it is likely that such influence will increase. So far, two measures have been introduced which aim to provide protection for the habitats of wild plants and animals.

4.2.2.1 The Birds Directive

The Directive on the Conservation of Wild Birds 79/409 was adopted in 1979 and took full effect in 1981 [Reid, 1994]. Under the Directive, Member States are required to take measures to maintain a sufficient diversity of habitats for all bird species occurring in their territory. They are also required to designate special protection areas (SPAs) to conserve the habitats of certain listed rare or vulnerable species listed in an Annex. Areas so designated must then be protected so as to avoid pollution, habitat deterioration and disturbance. In Britain, protection of SPAs is provided mainly through the town planning and SSSI systems. Designation of SPAs is carried out by the government in consultation with the conservation agencies.

The provisions of the Wildlife and Countryside Act 1981, which was introduced partly as a result of the Birds Directive, do achieve broad conformation, but some problems remain [Reid, 1994]. Only a small number of SPAs have been designated: in England, fifty sites classified to date, and a further 35 potential sites [DoE, 1994]. Of those designated, many are protected only as SSSIs, without management agreements to ensure maintenance of habitats [Reid, 1994]. It has been established through the European Court of Justice that damage or disturbance to SPAs can only be justified on grounds of public health and safety (*Commission* v *Federal Republic of Germany*, known as the Leybucht Dyke case), and not on those such as economic or recreational pressure. The requirements of the Directive are therefore incompatible with the wide discretion normally available to LPAs to allow damaging operations in SSSIs, and further protection is deemed necessary. Several issues are addressed in PPG9, which will be discussed below. Britain is not, however, the only Member State to encounter problems in the implementation of the Birds Directive. A total of fifteen cases have so far come before the European Court of Justice, the greatest number for any piece of EC legislation [Wils, 1994].

Although the Birds Directive has undergone adaptations in terms of the species listed in the Annexes, the only amendment to the actual wording comes through the EC Directive on Habitats, which replaces the Article concerning measures to be taken with regard to SPAs [Wils, 1994]. The Habitats Directive may be seen as a broadening of the Birds Directive to encompass the full range of wild fauna and flora and natural habitats.

4.2.2.2 The Habitats Directive

The Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43) was adopted in May 1992 and took effect in May 1994 [Reid, 1994]. The Directive aims to conserve habitats which are of ecological significance, either in themselves or as the host of threatened species. Member States are required to establish Special Areas of Conservation (SACs), with the objective of creating by the year 2000 a network (Natura 2000) within Europe. Although a separate timetable applies to the designation of SACs, legislation was required to be in place by June 1994. In Britain, implementation was through the Conservation (Natural Habitats &c.) Regulations 1994.

Annexes of the Directive contain lists of the habitats and species for which SACs should be designated. The process of designation occurs in two stages. First, a list of sites is proposed by each Member State, selected according to criteria set out in the Directive, must be completed by June 1996 [Reid, 1994]. During the second stage, the Commission will prepare a draft list of sites of EC importance, again according to criteria contained in the Directive. The final list is to be adopted by May 1998 by the Council of Ministers, on the basis of a qualified majority vote. A major innovation in the Habitats Directive is that, in exceptional circumstances, the Commission has the power to initiate designation of a site which the Member State has not listed. The final decision on whether the site is included requires the Council to vote unanimously, allowing the power of veto to the Member State concerned. The potential remains, however, for the EC to intervene in the selection of designated sites.

Once a site has been designated, a Member State must take appropriate measures to avoid deterioration of the habitat, or disturbance of the species, for which the site was notified [Reid, 1994]. In the British government's opinion, "Existing legislation... already provides a sound basis for implementation of the Directive" [DoE, 1993 p.137], but additional measures are necessary "in areas where conservation and protection from threats or damage was previously achieved through the application... of policy" [*ibid.* p.140]. Accordingly, the Habitats Regulations contain amendments to planning legislation and the Wildlife and Countryside Act. Government policy relating to the Habitats Regulations is contained in PPG9, which sets out the procedure for consideration of development proposals which may affect SPAs and SACs (Fig.4.2).

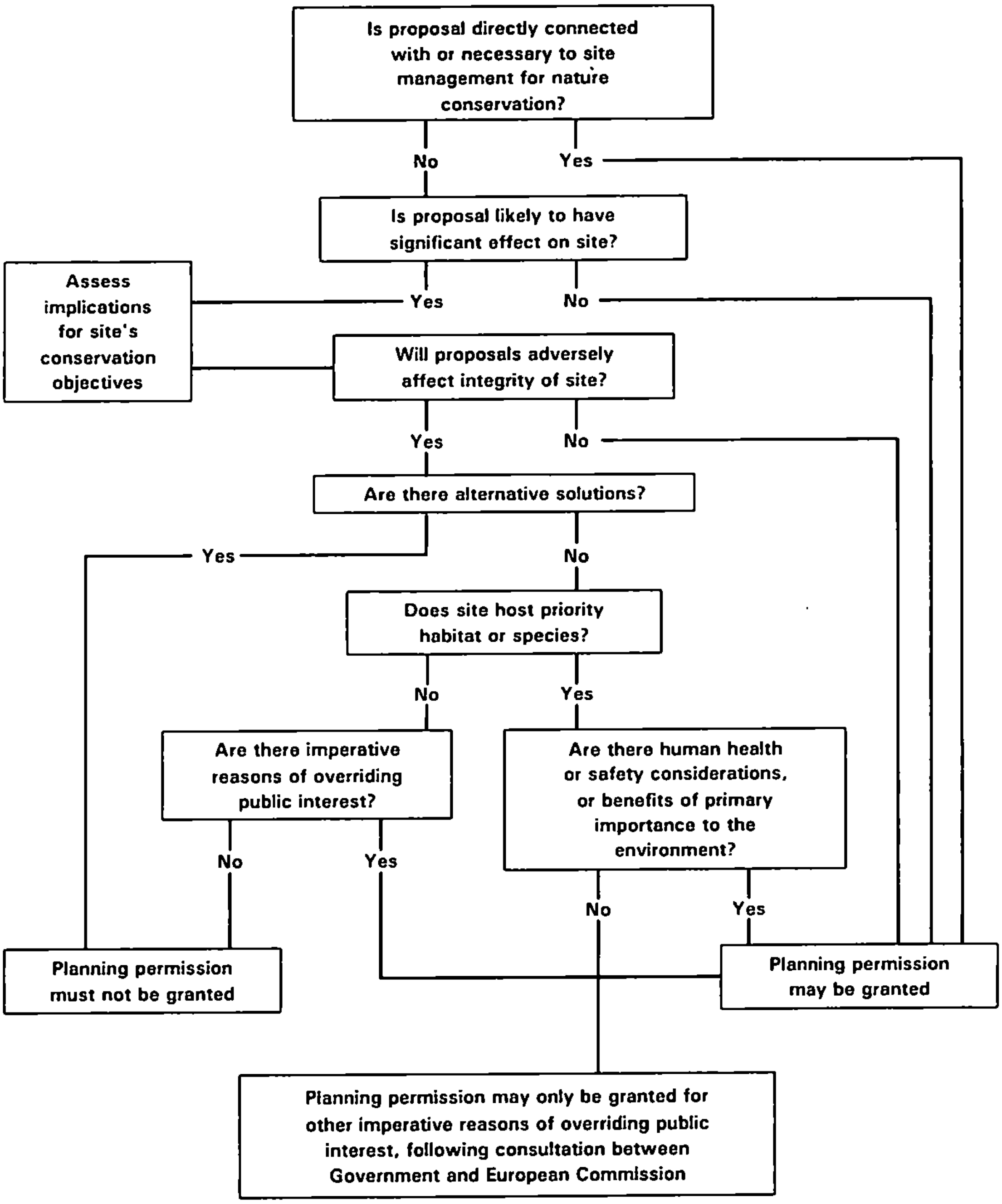

Fig.4.2: Consideration of Development Proposals Affecting SPAs and SACs

Department of the Environment [1994]

An LPA must establish whether a development proposal is likely to have a significant effect on an SPA or SAC in terms of the ecological objectives for which the site was designated [DoE, 1994]. The development may only proceed if the integrity of the site is not affected, or if there are no alternative available sites and the project has to be carried out for imperative reasons of overriding public interest, including those of a social or economic nature. However, if the site hosts a priority habitat or species, planning permission must not be granted unless the proposed development relates to human health or public safety, or is of primary importance to the environment. If such development proceeds, the Secretary of State is required to ensure that compensatory measures are taken to assure the overall coherence of the Natura 2000 network.

Further provisions are made so that developments which benefit from PDRs do not proceed in or near an SAC or SPA without consultation with English Nature [DoE, 1994]. Additionally, for Schedule 2 projects under the Town and Country Planning (Assessment of Environmental Effects) Regulations, an EA will normally be required where a potential or classified SPA, or a candidate, agreed or designated SAC could be affected. The government is also reviewing legislation such as the Water Resources Act 1991 and the Water Industry Act 1991 in order to ensure that, in line with the Directive, the activities of statutory bodies do not have an adverse effect on SPAs or SACs.

It appears, however, that few provisions have been made for the positive management of habitats, apart from providing EN with the power to re-issue notifications of PDOs, and to review management agreements in order to meet the terms of the Directive [DoE, 1994]. The problem of activities which lie outside the planning system, particularly agriculture, has not been addressed at all. The duty to "take appropriate steps to avoid... the deterioration of natural habitats" [*ibid.*, p.38] raises some interesting issues in relation to eutrophication of freshwater SSSIs. Nutrient enrichment often leads to deterioration of habitat through the loss of submerged macrophytes, which affects the entire trophic structure. The duty to manage lakes and rivers in order to conserve those species and habitats for which individual SACs are designated will fall to the National Rivers Authority, as part of their duty to promote the conservation of aquatic flora and fauna [Ball and Bell, 1994]. There is scope, however, for providing EN with greater powers to negotiate management agreements, especially in the catchments of designated sites. The costs involved can be met in part by the availability of EC funding [DoE, 1994].

The Directive states that "the Commission shall identify... those measures essential for the maintenance or re-establishment... of the priority natural habitat types" [*ibid.*, p.38]. For eutrophicated freshwater SSSIs such measures may well involve the reduction of nutrient loadings both from wastewater treatment and from agricultural runoff.

In relation to SSSIs in general, the process of strengthening certain aspects of the law, together with the publication of PPG9, may lead to an overall heightening of awareness of nature conservation interests, especially among LPAs. Apart from this indirect benefit, the only tangible additional protection afforded to SSSIs is through the definition of consultation areas around sites. Providing this measure is implemented fully, it may prove to be valuable tool in preventing development which may prove detrimental to freshwater SSSIs. The DoE has emphasised, however, that EN should give priority to defining consultation areas around sites of international importance, implying that the protection of other SSSIs is of lesser importance in this respect.

In creating a 'top tier' of designated areas, in the form of SPAs and SACs, there is the suggestion that some SSSIs are more valuable than others [Council for the Protection of Rural England, 1992]. This may give rise to the argument that lower tier sites are less strictly protected and therefore more expendable. It needs to be asserted not only that *all* SSSIs are of national importance, but also that those sites currently designated are intended to protect the "necessary minimum of British wildlife" [NCC, 1989 p.8].

4.2.3 International Conventions

Of the many international agreements to which Britain is a party, two in particular are of relevance to this study. These are the Ramsar Convention and the Bern Convention, both of which served to influence British nature conservation legislation.

4.2.3.1 The Ramsar Convention

The Convention on Wetlands of International Importance Especially as Waterfowl Habitat was adopted at Ramsar, Iran in 1971, and ratified by the UK in 1976 [Ball and Bell, 1994]. The term 'wetland' applies to areas of fen, marsh, peatland, or water. Areas of water may be naturally or artificially produced, standing or flowing, and fresh, salt or brackish. The Convention imposes quite general obligations, under which States should promote the conservation of wetlands and wildfowl, especially by establishing nature reserves. States are required to select at least one wetland for conservation. Deletion or reduction in size of a site is permissible only on the grounds of urgent national interests. Guidelines on the definition of 'international importance' have been drawn up, and there are provisions for the monitoring of wetlands, the establishment of a database, and the funding of projects.

The Ramsar Convention is implemented in Britain through the planning and SSSI systems [Ball and Bell, 1994]. Designation does not guarantee protection, but LPAs are advised of the need to promote the conservation of such sites, and to avoid the loss of wetland resources within them as far as possible. At present, there are nearly fifty Ramsar sites in England, which roughly correspond with sites designated as SPAs under the EC Birds Directive [DoE, 1994]. Ramsar sites are selected by the government in consultation with EN. The significance of this Convention is that it pioneered international efforts to protect wetlands [Reid, 1994]. It also established a List of Wetlands of International Importance, maintained by the International Union for the Conservation of Nature, which must be informed of ecological changes to the wetlands as a result of development, pollution or other human interference.

4.2.3.2 The Bern Convention

The Convention on the Conservation of European Wildlife and Natural Habitats was agreed in Bern in 1979, under the direction of the Council of Europe [Reid, 1994]. It aims to conserve wild flora and fauna and their natural habitats, to promote cooperation between countries in their conservation efforts, and to give particular emphasis to endangered and vulnerable species. Signatories are obliged to introduce legislative and administrative measures to ensure the conservation of the habitats of threatened species, and to minimise deterioration of such habitat through planning and development policies.

This Convention is given force in the UK through the Wildlife and Countryside Act 1981. Its practical impact is now somewhat diminished since the implementation of the EC Birds Directive and the EC Habitats Directive [Reid, 1994]. The Convention did, however, perform a major function in providing the framework for these Directives.

4.3 Agriculture

The contribution of agriculture to nutrient loadings on eutrophicated lakes and reservoirs is often significant (Section 2.2). Elevated levels of nutrient losses are largely the result of the use of intensive farming practices, encouraged by agricultural policy and legislation. An understanding of the development of agricultural support is therefore necessary, in order to appreciate the pressures and constraints which influenced farmers to intensify and specialise.

4.3.1 Agricultural Support in the Post War Era

Preparations for post war reconstruction in Britain presented unique opportunities for a reappraisal of government policy in relation to agriculture and land use planning [Lowe *et al.*, 1986]. Modest levels of government financial support for farming had been introduced during the 1920s and 1930s, but post war optimism led to the provision of a comprehensive legislative framework. Thus the Agriculture Act 1947 was designed to provide cheap food supplies for consumers and secure incomes for farmers. The Town and Country Planning Act of the same year assisted in the regeneration of agriculture by bestowing on farming and forestry priority rights over rural land use. The motivations for such preferential treatment was not only to restore security to agriculture, but also to protect the rural environment from urban encroachment.

The Agriculture Act provided a system of guaranteed prices for all major agricultural products [Lowe *et al.*, 1986]. The NFU was given a statutory right of consultation on the level of such prices. In return, farmers were required to maximise production through 'good husbandry', which included making full use of previously uncleared land. It was soon realised that certain sectors of production could be encouraged by capital investment grants. The Milk and Egg Marketing Boards were established in order to stimulate increased efficiency. Other developments designed to increase productivity included the introduction of fertiliser subsidies in 1952, which remained available until 1974 [Chalmers, Kershaw and Leech, 1990].

The Agriculture Act of 1957 developed even further the idea of ever-increasing efficiency and reducing unit costs of production [Lowe *et al.*, 1986]. Each year, farmers were obliged to recoup an increasing proportion of their costs through greater efficiency. This could only be achieved through accepting available grants to invest in new buildings and machinery, adopting new technologies, taking more land into production, reducing their labour force, and concentrating production in larger units. By 1963, however, world commodity surpluses had pushed prices lower and lower, escalating the cost of agricultural subsidies. The government therefore introduced minimum import prices in order to maintain British farm incomes. Such protectionism was developed further when, in 1972, Britain entered the EEC, and its agriculture became subject to the Common Agricultural Policy.

4.3.2 The Common Agricultural Policy

Britain was not, of course, the only country which attempted to create an efficient farming industry in order to supply food needs and reduce imports [Lowe *et al.*, 1986]. Widespread food shortages across Europe during the second world war meant that other countries were also keen to increase agricultural production. When the EEC was established in 1957, it provided an opportunity for cooperative efforts to strengthen European agriculture in the world market. From the beginning, agricultural support was a central tenet of the EEC.

The objectives of the Common Agricultural Policy (CAP) were established under the Treaty of Rome as being [National Audit Office (NAO), 1985]:

(a) to increase agricultural productivity
(b) to ensure a fair standard of living for the agricultural community
(c) to stabilise markets
(d) to assure the availability of supplies
(e) to ensure that the supplies reach consumers at reasonable prices

Community legislation in relation to agriculture is normally implemented through Regulations [NAO, 1985], which, as outlined in Section 4.1.2, have direct force in British law. The objectives of CAP are achieved by either guarantee or guidance measures. Guarantee measures support the prices of agricultural commodities, providing protection against imports to the EC. The main mechanisms used are intervention buying, import levies and export refunds. For some commodities, such as cereals, milk and beef, the market is fairly tightly regulated. Guidance measures are designed to give financial support for investment, to provide assistance to farming in Less Favoured Areas (LFAs), and, increasingly, to encourage farmers to pursue desirable agricultural practices.

In terms of productivity, the farming industry in Britain achieved significant increases after joining the EEC [NAO, 1985]. Between 1973 and 1983 there was a 49% increase in productivity. The Community as a whole was more than self-sufficient in most supported agricultural products, generating substantial surpluses. The costs involved in storing and disposing of agricultural surpluses imposed a considerable strain on the Community budget. This factor, coupled with assertions that the agricultural sector was stimulated at the expense of other areas of the economy, led to criticism of the CAP.

The pace of change and intensification of agriculture also caused concern over its effect on landscape and wildlife conservation, particularly in the light of the EC's Third Action Programme on the Environment, which called for:

> "the creation of an overall strategy making environmental policy a part of economic and social development. This should result in a greater awareness of the environmental dimension, notably in the fields of agriculture (including forestry and fisheries), energy, industry, transport and tourism." [House of Lords, 1984 p.viii]

In the UK, a Select Committee was appointed to investigate the effects of agriculture on the environment, and, in particular, to consider the implications of the EC's draft Regulations on improving the efficiency of agricultural structures [House of Lords, 1984]. Evidence presented to the Committee highlighted the fact that EC agricultural policy favouring high levels of production, particularly the guarantee measures, had led to the loss of wildlife habitat and landscape, through encouraging intensification of production and the reclamation of marginal land. With respect to the draft Regulations, the Committee considered that there was little real evidence of the intention to integrate environmental requirements with agricultural policy. It therefore advocated a number of amendments, including a more liberal interpretation in regard to the designation of LFAs, in order to permit financial support for the preservation of areas of particular conservation value.

A further aspect of the effect of agricultural policy on the environment was expressed during the 1985 Parliamentary review of the workings of the Wildlife and Countryside Act 1981 [Whitby and Lowe, 1994]. In relation to the practice of compensating farmers for not carrying out PDOs, it was stated:

> "The illogicality of one part of the government (MAFF) offering financial inducement to someone to do something which another part of government (DoE and related bodies) then has to pay him not to do, is clear. The primary reason for the negative character of management agreements is [that they] arise in order to control a farmer subsidised to damage the environment... [T]he underlying logic is inescapable: to prevent the conflict the farmer should not be encouraged to damage the environment in the first place." [*ibid.*, p.9]

Government Ministers became convinced of the need for reform of agricultural support, especially in respect to the environment [Whitby and Lowe, 1994]. The conclusions of the Select Committee were accepted, and led to the British government championing an important modification to the proposed EC Regulation. When Regulation 797/85 was subsequently passed by the EC in 1985, it enabled Member States to introduce national schemes in what became known as Environmentally Sensitive Areas (ESAs).

In 1987 MAFF launched a £25 million package, a central proposal of which was the expansion of ESAs [Whitby and Lowe, 1994]. Further policy initiatives by the EC and MAFF included environmental objectives to a greater or lesser degree. Introduction of the EC Set-Aside Scheme in 1988 was specifically aimed at reducing arable production and hence costs of intervention purchase, storage and disposal [Robinson, 1991]. The various options available for Set-Aside land, however, include the Countryside Premium Scheme and, more recently, the Habitat Scheme, both of which may be used to enhance nature conservation interest.

Important reforms of the CAP were agreed in 1992, under which there is now virtually no likelihood of further loss of grassland to arable production, and large operators will be under pressure to reduce beef and ewe stocking rates [Colman, 1994]. The apparent commitment of MAFF to conservation is part of a general shift in EC agricultural policy in which environmental programmes are assimilated, not necessarily through conviction, but because such schemes provide opportunities for the alleviation of surplus and budgetary problems [Whitby and Lowe, 1994]. Nevertheless, these policy changes promote what has come to be known as the 'extensification' of agriculture, which, regardless of intended objectives, provides opportunities for environmental improvement.

4.3.3 Agricultural Extensification

As described above, the combination of excess production and growing environmental awareness has engendered almost a reversal in agricultural policy in recent years, to the extent that extensification is the aim of several government and EC support schemes. The question for those concerned with nature conservation and water quality is whether such schemes, which are mainly centred on reducing agricultural production, can actually produce environmental benefits, or whether specific strategies need to be employed in order to reduce nutrient export from agricultural land. Those programmes which will be discussed here are ESAs, the various options available for Set-Aside land, and the Organic Aid Scheme.

4.3.3.1 Environmentally Sensitive Areas

Powers to designate ESAs were introduced in 1985 by Regulation 797/85 [Whitby and Lowe, 1994]. The relevant Article was implemented in the UK by the Agriculture Act 1986, which empowers Ministers to designate ESAs. It was modified in 1987 when the EC adopted Regulation 1760/87, which provided for a maximum reimbursement of 25% from the European Agricultural Guidance and Guarantee Fund.

Orders designating the First Round ESAs came into force in March 1987 [Whitby and Lowe, 1994]. Designating orders for the Second Round ESAs applied from January 1988. In November 1991, MAFF announced two new rounds of ESAs: six to become effective in 1993 and six more in 1994. The selection of ESAs by MAFF is conducted in consultation with statutory advisors such as English Nature and the Countryside Commission, and interested parties such as the NFU. Designating orders identify the boundary, set the rates of payments, and the agricultural prescriptions for each area.

Farmers in ESAs are offered a five year agreement which awards them an annual payment in return for following a prescribed set of farming practices [Whitby and Lowe, 1994]. These include restrictions on fertilizer use and grazing levels, and bans on the use of herbicides and pesticides and on the installation of new drainage. Positive prescriptions include maintenance of hedges, woods, barns and ponds. The rates of payment take account of the actual and potential profits which a farmer forgoes by following the prescribed management practices. Both practices and payments are standard within areas, but vary between one ESA and another. In some areas, however, a second, higher rate of payment is required for additional operations, such as the reinstatement of ploughed-up pastures.

According to MAFF, uptake of the ESA scheme has been appreciable, especially in England, where, by the end of the second year, about 87% of the total targeted area of land was entered [MAFF, 1989]. The attraction of the ESA scheme for farmers, compared to SSSI regulation, is the fact that it is entirely voluntary, less restrictive, simpler to enter, and dealt with by agricultural officials, rather than conservation personnel [Whitby and Lowe, 1994]. On the whole, ESAs have gained support from both the farming and the environmental lobbies. The scheme represents a significant divergence for agricultural policy, creating important precedents for the future. The selection procedure, in which initial lists of potential ESAs are submitted to MAFF by the country agencies, ensures that, to a large extent, the scheme is targeted at areas which will benefit most from designation. The acceptability of the scheme to farmers guarantees a high take-up rate. Many of the prescribed practices will have a direct impact on the reduction of diffuse pollution, including nutrient losses. Nevertheless, ESAs fall short of a fully integrated approach to environmental management and legitimate criticisms of the scheme exist.

MAFF has rejected calls both from farming and from conservation bodies for a more widespread application of the scheme, which suggests that ESAs may remain a well-publicised, but minor component of agricultural policy [Whitby and Lowe, 1994]. The scheme may even be seen as a distraction from a more general reform of agriculture. Indeed, amendment of certain aspects of MAFF and EC policy could greatly reduce the need for ESA designation in some sectors. A second criticism relates to the voluntary nature of the scheme. Those farmers who choose not to register, or who enter only part of their farm, may remain eligible for MAFF grants to undertake environmentally damaging improvements. While it is necessary to promote goodwill among farmers, some degree of cross-compliance would not necessarily detract from the appeal of the ESA scheme.

In individual ESAs, specific problems exist which require attention in revised management prescriptions. For instance, in the Suffolk River Valleys ESA targets for reductions in fertiliser rates and stocking levels were already being exceeded as part of existing practices [Russell, 1994]. A closer correlation between targets and current practices could therefore achieve greater conservation benefits. Under the scheme, only agricultural land is eligible for payments, so that in ESAs which contain important mires, fens and other wetlands, such as in Breckland, these features are not protected [O'Carroll, 1994].

For those ESAs for which monitoring and evaluation have been completed, it appears that the schemes have slowed down the rate of damage to existing landscapes, habitats and features, but that the extent to which conservation interest has been enhanced is limited to a few areas [Colman, 1994]. It seems that as EC agricultural policy generates much less pressure for agricultural intensification, and convergence with ESA policy occurs, justification for the latter may be reduced, unless the benefits of positive management become more apparent.

4.3.3.2 Options under the Set-Aside Scheme

In 1988 the EC introduced the Set-Aside Scheme for arable land under Regulation 1272/88, in an attempt to reduce production in this sector [Reid, 1994]. In Britain this has been implemented by Set-Aside Regulations 1988, along with various later amendments. The scheme provides for arable land to be taken out of production voluntarily by farmers, for a period of five years [MAFF and WOAD, 1988]. The land set aside must be no less than one hectare in area, and represent at least 20% of the arable land on the holding. In 1992, under EC Regulation 1765/92, significant reductions in price support for cereals and oilseeds are compensated by direct payments per hectare (Arable Area Payment: AAP) which, for producers with more than forty hectares of cereals, are only granted if 15% of the holding is set aside [Colman, 1994]. By 1993, about 1.5 million hectares of land was set aside, which constitutes roughly 3% of the total arable area within the Community [Ansell and Vincent, 1994].

There are a number of opportunities for farmers to take up options on Set-Aside land. Under the basic Set-Aside Scheme, land may be left fallow, permanently or in a rotation system, or used for extensive grazing, converted to woodland or used for non-agricultural purposes [Lennon, 1992]. Any agricultural production is prohibited on Set-Aside land, except for grazing fallow, to which strict rules apply. Farmers must preserve the appearance of the countryside, by maintaining existing hedges, trees, water courses, lakes, unimproved grassland, moorland, heath, and vernacular buildings. The Set-Aside with Trees option may be operated either directly under the Set-Aside regulations, or through the Farm Woodland Scheme (FWS).

The potential environmental benefits of Set-Aside were discussed in principle in Section 2.2.2. The opportunity exists to exploit Set-Aside land for nature conservation, and nutrient export reduction in particular. For instance, buffer strips along streams may be established using the fallow or woodland options. The main disadvantages of this approach, under the basic Set-Aside scheme as outlined above, are that:

(a) uptake is entirely voluntary, and therefore untargeted
(b) any additional management necessary for the benefit of wildlife or for nutrient reduction may not be eligible for compensation payments
(c) the basic Scheme applies only to arable land
(d) the minimum period for which farmers may enrol land is five years

It is clear therefore that further measures are necessary for Set-Aside land to serve environmental objectives. Two such Schemes currently exist which aim to enhance nature conservation interest on land taken out of agricultural production.

In 1989, the CC launched the Countryside Premium Scheme (CPS), following a request by the DoE to develop a programme which would act as a 'top-up' to the Set-Aside Scheme [CC, 1993]. The aim is to achieve environmental and recreational objectives through financial incentives, which are additional to payments under Set-Aside. An important difference between the CPS and Set-Aside is that the former is discretionary, with the CC retaining the right to refuse or alter inappropriate applications.

Suitable Set-Aside land, mainly permanent fallow, may be eligible for grants to establish wooded margins, meadowland, wildlife fallow, Brent Geese pasture, or for the restoration of certain wildlife habitats [CC, 1991]. The first three options may provide direct or indirect benefits in terms of reduction in nutrient losses, since no organic or inorganic fertilisers should be used in their establishment or management. Additionally, fallow and wooded margins may act as barriers to soil erosion. Habitats which may be reinstated include nutrient-poor lowland heath, chalk grassland and herb-rich meadow. Of particular interest here is the inclusion of wetland habitats. The possibility exists of applying this option to lake and stream fringes in order to enhance both the nature conservation interest and the nutrient retention capacity of wetland habitats.

A separate long-term set-aside programme, known as the Habitat Scheme (HS), is currently being applied in pilot areas. Authority for the Scheme comes from the EC 'Agri-Environment' Regulation 2078/92 which constitutes part of the CAP reforms [MAFF, 1993]. The HS aims to encourage the longer term retirement of land from agricultural production in selected areas, for the creation or improvement of a range of wildlife habitats. The Scheme offers a payment per hectare over a period of twenty, rather than five, years. Unlike basic Set-Aside, the HS applies to permanent grassland as well as arable. Land may be converted to one of two habitat types, or existing habitat created under the five year Set-Aside Scheme may be transferred to the HS.

Of particular interest here is the option to recreate, restore or extend Water Fringe habitats, regulations for which came into force on 6 June 1994 [*Environmental Law Bulletin*, 1994a]. Selection criteria for pilot areas include rivers and lakes designated as SSSIs, or those suffering from eutrophication [MAFF, 1993]. MAFF has so far designated six pilot Water Fringe Areas, including Slapton Ley and its catchment [MAFF, 1994a]. The scheme is structured around five management options, which may be applied to different areas of a farm:

1A: withdrawal of permanent grassland from production

1B: withdrawal of arable land from production

2A: extensive grassland management on existing permanent grass

2B: extensive grassland management on currently arable land

3: raised water level supplement

Under options 1A and 1B, strips of land with an average width of between 10m and 30m wide adjacent to water courses and lakes may be entered into the scheme [MAFF, 1994b]. After the establishment of a permanent grass sward on previously arable land, subsequent management under both options includes removal of livestock and cessation of fertilisation. A management plan must be agreed with MAFF, under which the land may be allowed to develop into scrub, or mown every year, or a combination of these. In cases where nutrient levels are high, vegetation may be cut more than once a year.

Under options 2A and 2B, whole fields may be taken out of production and used for grazing at a stocking density of less than 0.75 livestock units per hectare [MAFF, 1994b]. Stockproof fencing may be necessary to protect the water's edge from erosion. Cutting or topping grass is only allowed after the beginning of July and before the end of August, in order to encourage flora and fauna. Any land which is accepted under these four options may also be entered under option 3, for which a supplement is payable. The purpose of this option is to intercept field drainage flows in order to reduce nutrient and sediment export, or to maintain water levels in ditches during winter for the benefit of wildlife. The supplement lasts for the full twenty years on land entered into options 1A and 1B, but for only ten years under the extensive grazing options.

Since the Water Fringe option of the HS is currently at the pilot stage, modifications may yet be applied. Nevertheless, the inclusion of interception of field drainage only as an additional option diminishes the effectiveness of the scheme to reduce diffuse nutrient loads on eutrophicated rivers and lakes. Some benefit to water quality may accrue from the cessation of fertilisation and from lower stocking densities, but without the removal of underdrainage it is unlikely that improvements would be detectable. In addition, although the scheme is targeted to the extent that sites are carefully chosen by MAFF, in consultation with English Nature and other appropriate agencies, there remains the usual problem associated with a voluntary programme. In the catchment of Slapton Ley, only 5% of eligible bankside was under the scheme up to January 1995, and none was entered into the raised water levels option [Hooper, 1995].

This low level of uptake may be related to concerns that the payments may not be adequate to cover the costs of appropriate management. A further drawback from the farmer's perspective is that, initially, arable land entered into the scheme did not count towards their Set-Aside commitment under the AAP Scheme [MAFF, 1994a]. MAFF successfully lobbied the EC, however, and arable land under the HS now counts towards AAP [M. Gibson, EN, pers. comm.], which may improve uptake. Despite the amount of land entered into the scheme at this stage, it represents an important progression towards the integration of environmental objectives into agricultural policy, and, in particular, an acknowledgement of the effects of intensive farming on water quality.

4.3.3.3 Organic Aid Scheme

The Organic Aid Scheme (OAS) was introduced by MAFF in 1994 in order to assist farmers in the conversion of land to organic production [MAFF, 1994c]. Authority for the OAS comes from the EC 'Agri-Environment' Regulation 2078/92, implemented through the Organic Aid Scheme Regulations 1994. Any agricultural land not already in organic production may be entered into the OAS. A minimum of 1 hectare and a maximum of 300 hectares will be eligible for aid. Farmers must apply to register the land as 'in conversion' with an approved body, such as the Soil Association (SA). Parcels of land may be entered into the Scheme at different times over a four year period, and each will receive aid for five years. Payments decrease following the first two years, since this is the period after which produce from non-perennials may be sold as organic and attract a premium. The OAS is intended to increase the area of land under organic management to 150,000 hectares [SA, 1994].

According to the SA, schemes introduced under the EC 'Agri-Environment' Regulations 2078/92 are "grossly underfunded" [SA, 1994 p.4], attracting only 4% of the total amount paid to farmers under the CAP, compared to 43% spent on Set-Aside and AAP. It considers that the OAS has the potential for widespread application as the basis for sustainable CAP reform. In terms of habitat and wildlife conservation, output reduction, public support and suitability for whole farm entry, the SA regards the OAS as the most applicable scheme for the full integration of environmental and agricultural objectives.

It has been argued that the costs to the CAP of conventional Set-Aside are considerably higher than those which would be incurred by conversion of the same land to organic production [Midmore and Lampkin, 1988]. However, a model developed by Lampkin [1993], predicts that 30% of UK agriculture would need to be converted to organic production in order to achieve the 15% reduction in arable output currently being sought through the Set-Aside programme. It has been demonstrated that the market for organic produce is strong enough to cope with such a significant increase in supply, without a major effect on the premiums currently obtained [Lampkin, 1989]. Considering, however, that, at present, less than 1% of agricultural land is under organic production [SA, 1994], an increase to 30% seems, at best, hypothetical, even with further financial support for conversion. The contribution of the OAS to the prime objective of extensification, output reduction, will probably remain marginal. The benefits of organic farming to nature conservation, and, to a certain extent, water quality, are, however, well established (Section 2.2.3). The most effective use of OAS may therefore entail targeting the scheme at those catchments which may benefit from the less intensive agricultural practices which are the focus of organic farming.

4.4 Water Quality

Water quality legislation performs an increasingly important role in controlling nutrient concentrations in rivers and lakes. It is within this sphere that future policies in relation to eutrophication are most likely to be enacted. Although water quality standards in this country are highly influenced by EC legislation, Britain (effectively England and Wales in this context) possesses its own history of regulation, which will be briefly described. The most noteworthy event in the development of water quality legislation in Britain was the implementation of the Water Act 1989. Despite fierce arguments against those Sections of the Act dealing with privatisation of the water industry, there is little doubt that the creation of the NRA was an unprecedented step in terms of regulation and enforcement of water quality standards. In addition, provisions in the Act allowed for the designation of catchments in order to protect water quality, and the applicability of these to the problem of eutrophication will be discussed. Those EC laws of particular relevance, the Directives on Nitrates, and Urban Waste Water Treatment, and the draft Directive on Ecological Quality of Surface Waters, will be addressed in some detail.

4.4.1 The Development of Water Quality Legislation

The development of institutional arrangements for water management is complex [Pitkethly, 1990]. The main pieces of legislation which determined this development are shown in Table 4.2. Up to the second world war, the responsibility for water supply and sewage disposal lay largely with local authorities. The Water Act 1945 enabled the creation of water boards, but these did not become widespread until the 1950s and 1960s. Pollution control remained within the remit of the local authorities through their public health responsibilities [Ball and Bell, 1994].

Table 4.2: Post War Legislation Regulating Water Management

Pitkethly [1990] and Ball and Bell [1994]

Year	Legislation	Provisions
1945	Water Act	Created Water Boards
1948	River Boards Act	Established River Boards
1951	Rivers (Prevention of Pollution) Act	Consents required for new industrial and sewage discharges
1961	Rivers (Prevention of Pollution) Act	Consents required for discharges commenced before 1951
1963	Water Resources Act	River Boards changed to River Authorities. Functions extended to abstraction licensing and control of certain discharges to groundwater
1973	Water Act	Created RWAs responsible for full range of water management functions
1974	Control of Pollution Act	Consent system extended and public registers set up
1983	Water Act	Changed management structure of RWAs, facilitating moves towards privatisation
1989	Water Act	Introduced privatisation of water industry and created NRA
1991	Water Resources Act Water Industry Act	Consolidated Water Act 1989

An attempt to create a more logical system led to the River Boards Act 1948, which created 32 river boards, organised on a catchment basis. In 1951, under the Rivers (Prevention of Pollution) Act, new industrial and sewage discharges required consents from the River Boards. This system was extended in 1961, by another Act of the same name, to include those discharges commenced before 1951. The River Boards were converted into 27 River Authorities by the Water Resources Act 1963, and were given a number of regulatory powers, including those over certain discharges to groundwaters, and water abstraction licensing.

At this time, as well as the River Authorities, there were also a total of 157 water supply and 1,393 sewage authorities [Ball and Bell, 1994]. This rather fragmented system was transformed by the Water Act 1973, which established a fully integrated scheme of river and water management. The Act created ten Regional Water Authorities (RWAs), based on river basin areas. These became responsible for the full range of water management functions, including treatment, supply, sewerage and sewage, land drainage, flood defence and pollution control. Some 27 private water companies were not involved in the reorganisation and continued to supply water in defined areas.

A major progression in terms of water quality regulation came in the form of the Control of Pollution Act (COPA) 1974, which further extended the coverage of discharge consents, creating the basis for the current system. It also established a precedent in public accountability, by establishing registers of information.

The advent of privatisation of the water industry was promoted through the Water Act 1983 [Pitkethly, 1990]. This replaced the rather unwieldy, but representative, boards of the RWAs with much smaller groups of appointees, overseen by executive chairpersons, many recruited from industry, who soon called for privatisation. There were undoubtedly major problems surrounding the operation of the RWAs, most notably substantial underfunding by government of capital projects, and the conflict of interest between the functions of sewage treatment and pollution control. Nevertheless, the government's argument for privatisation was largely based on the idea that the industry should be accountable, although to whom was unclear at the time [Ball and Bell, 1994]. Supporters of the old system were outspoken in their condemnation of privatisation:

> "The Thatcher government... has changed the overriding goals towards which water management works to include greater regard for economic and market principles... [I]n the process the institutional structure for water management in England has changed from a model for reorganization elsewhere to a model of how a hostile government can undermine concepts of holistic resource management." [Pitkethly, 1990 p.119]

Whatever the political ramifications of privatisation, the Water Act 1989 nevertheless created a wide-ranging and independent regulatory body in the form of the NRA, the role of which will be discussed in Section 4.4.2.1. Arrangements set out under the Act were subsequently consolidated in the Water Resources Act 1991, which deals with regulatory aspects, and the Water Industry Act 1991, covering the privatised water companies. The post of Director General of Water Services, created under the Act, bears responsibility for regulation of the water industry [Ball and Bell, 1994]. Those features of the Water Industry Act which are relevant to this study, that is duties imposed on the water industry with respect to conservation, are outlined in Section 4.2.1.4, and will not be addressed further.

4.4.2 Water Resources Act 1991

The Water Resources Act (WRA) 1991, which applies to England and Wales, superseded Part II of COPA which dealt with most of the legislation covering water pollution [Ball and Bell, 1994]. The system of consents for discharges remained much the same, but some important changes occurred in other areas. These include the introduction of statutory water quality standards, and of a system of charges for trade and sewage discharges, and improvements in available powers for the prevention and remediation of pollution.

Under the WRA, it is an offence to cause or knowingly permit any poisonous, noxious or polluting matter, or solid waste, to enter controlled waters, which include virtually all inland and coastal waters [Ball and Bell, 1994]. Many of the words used in the Act carry no statutory definition, however, and must be obtained through practice or from litigation. The term 'polluting matter' appears to refer to substances with the potential for harm to organisms. This general offence is related to a more specific one of discharging trade or sewage effluent without a consent. Such an illegal discharge also constitutes a general offence if it causes pollution. Non-point discharges, such as agricultural runoff, are potentially covered by the general offence.

The main instrument for precautions against pollution lies in the power of the Secretary of State to make regulations in relation to any polluting substances in order to prevent it from entering controlled waters [Ball and Bell, 1994]. These may apply, for instance, to the storage of industrial chemicals and of farm wastes. The only regulations made so far are the Control of Pollution (Silage, Slurry and Agricultural Fuel Oil) Regulations 1991. These introduce specific controls over the storage of these substances, and apply to new or substantially altered facilities. Existing stores may be brought under control by the NRA if it believes there is a significant risk of pollution. The previous defence that a pollution incident was attributable to "good agricultural practice" is removed.

The Secretary of State and MAFF may approve new codes of good agricultural practice in order to promote practices for avoiding or minimising water pollution [Ball and Bell, 1994]. This provision resulted in the *Code of Good Agricultural Practice for the Protection of Water* [MAFF and WOAD, 1991]. The Secretary of State may also approve codes of practice in order to give guidance to the NRA and the water and sewage undertakers, with respect to general environmental duties.

4.4.2.1 The Role of the National Rivers Authority

The administrative framework under which water management in England and Wales is conducted is illustrated in Fig.4.3. The WRA requires the NRA to undertake a wide range of duties in relation to pollution control, flood defence, water resource planning and management, including abstraction licensing, navigation, freshwater fisheries, and water-related conservation and recreation [Ball and Bell, 1994]. The NRA is required to take specific steps in order to attain objectives set by the Secretary of State for maintaining and improving river quality. It is also required to monitor the extent of pollution in all controlled waters, including streams, lakes and coastal waters.

The primary means by which control of polluting discharges is exercised is through the system of consents. A consent to discharge is required from the NRA for any discharge of trade or sewage effluent into controlled waters. The system for acquiring a consent is that the applicant applies to the NRA, which will require details such as the place, quantity, rate of flow, composition and temperature of the proposed discharge [Ball and Bell, 1994]. The NRA must publicise the application in a local newspaper and in the *London Gazette*, and notify any relevant local authorities and water undertakers at the applicant's expense. However, if the NRA considers that the discharge will have 'no appreciable effect' on the receiving waters, this publicity may be dispensed with, as is the case for about 90% of applications. The NRA has the power to grant or refuse a consent.

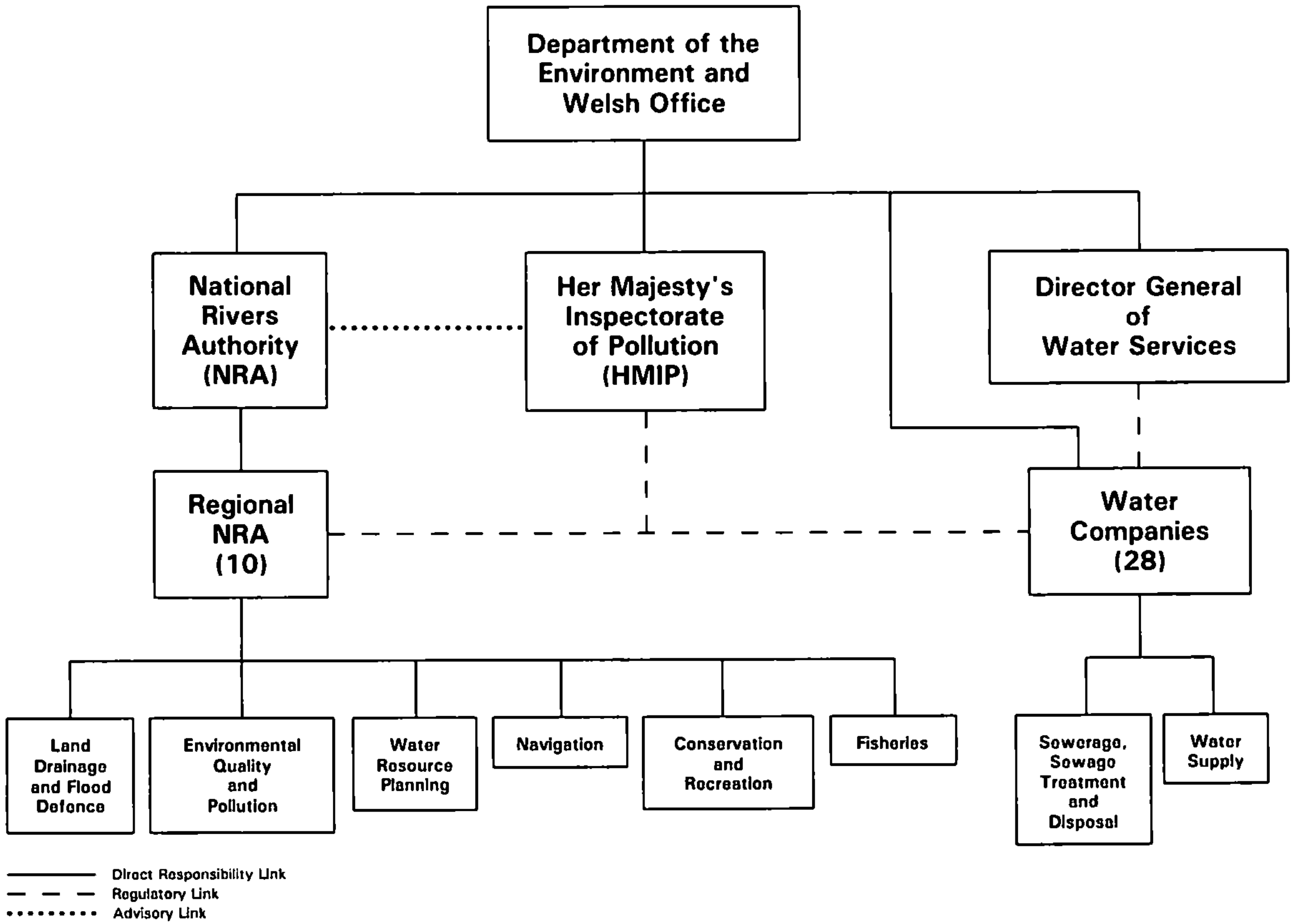

Fig.4.3: Water Management in England and Wales

Based on House [1991]

Conditions are usually attached to a consent, the most significant of which relate to biochemical oxygen demand, levels of toxic or dangerous materials, and suspended solids [Ball and Bell, 1994]. As a general rule, nutrient levels in discharges are not subject to consent limits.

For industrial discharges, absolute numerical limits are normally attached, but for sewage discharges a 95% compliance rate over twelve months is more usual [Ball and Bell, 1994]. This degree of leniency is partly a legacy of past practices, but it also allows for the variable quality of sewage effluent, which is sometimes outside the control of the undertakers. In 1991, the NRA conducted a comprehensive overhaul of the discharge consent system, one of the main objectives of which was to eliminate this inconsistency [*ENDS Report*, 1991b]. It is therefore in the process of phasing in new upper tier limits for sewage discharges. The only conditions under which consents will allow a breach will be during exceptional weather.

More recently, however, the DoE put forward proposals to amend the consent system as part of the government's deregulation initiative [*ENDS Report*, 1993d]. At present, consent conditions may not be altered within two years of issue. This period is to be extended to four years, which, in theory, will require greater foresight on the part of the NRA. In practice, consents are very rarely amended after two years. A more significant proposal concerns the consent appeal system. Dischargers are allowed the right of appeal against an NRA decision on a consent, but, at present, they must comply with the conditions until the appeal is determined. The DoE proposes that, where an appeal concerns the revocation or modification of an existing consent, it will not have effect until a decision is reached on the appeal. This revision would have little effect if the DoE's appeal system were efficient and rapid. As it is, a number of appeals dating from the period immediately after privatisation of the water industry were still outstanding at the end of 1993. In the interim period, the present appeal provisions have provided safeguards against peak discharges from sewage treatment.

Under the EPA 1990, discharges from processes subject to integrated pollution control, such as those involved in the petrochemical industry, became the responsibility of Her Majesty's Inspectorate of Pollution (HMIP), rather than the NRA [Ball and Bell, 1994]. HMIP operates and enforces a system of authorizations for such discharges, but it must consult the NRA beforehand. Any recommendations made by the NRA must be adhered to in making such authorizations.

The NRA is granted power of entry to property if it has grounds to suspect that breaches of control have occurred [Ball and Bell, 1994]. It also possesses the right to take samples of water or effluent, and to install monitoring equipment. When a pollution incident occurs, the NRA is empowered to carry out remedial or restorative work, the costs of which may be recovered from the person or company responsible. Prosecutions for water pollution have increased significantly since the establishment of the NRA, and, owing to the 'strict liability' nature of the offences, most are successful.

Under the WRA, a public register must be kept by the NRA of all applications for consent, together with details of conditions attached, as well as information on authorizations granted for processes subject to integrated pollution control [Ball and Bell, 1994]. The register must also include results of analyses of water and effluent samples. Any member of the public may inspect the register free of charge. The inclusion of a public right of access to monitoring data in the 1989 Act was influenced by the prospect of the EC Directive 90/313 on Freedom of Access to Information on the Environment.

The water quality implications of a proposed development acts as a 'material consideration' in planning decisions, and as such, the NRA is a statutory consultee for all applications under the General Development Order, although its advice may be disregarded [Ball and Bell, 1994]. The NRA also exercises a more general power to influence development plans, either directly or through the publication of policy statements. The NRA's *Guidance Notes for Local Planning Authorities on the Methods of Protecting the Water Environment Through Development Plans* [NRA, 1994] contains strong advice to resist development which could adversely affect water quality. LPAs are also recommended to avoid allowing development on recognised floodplains, and to support initiatives to conserve and enhance river corridors, in recognition of their importance for water quality and nature conservation, of their contribution to the character of the landscape, and as amenity areas.

Further duties placed on the NRA relate to nature conservation issues. As in the case of the water companies, it is required to further the conservation of flora and fauna in the course of carrying out its other duties [Reid, 1994]. It is also under a more specific obligation to promote the conservation and enhancement of the natural beauty of inland waters, and the conservation of aquatic species. These provisions are much stronger than in previous legislation, and have given rise to a range of projects undertaken by the NRA, often in conjunction with local and national organisations, specifically to improve the nature conservation interest of aquatic habitats. Examples include the development of water management plans for the Pevensey Levels [*Water Guardian*, 1995a], and reinstatement of the original meandering course of a section of the River Little Ouse in Norfolk [*Water Guardian*, 1994]. The NRA has also been highly involved in nutrient reduction programmes and restoration work in the Norfolk Broads [Phillips and Jackson, undated]. Of the total R & D budget for 1993/94, 7% was allocated for conservation related research [NRA, 1993a]. Projects undertaken include work on buffer zones for the conservation of rivers and bankside habitats, and the development of management strategies for the protection, rehabilitation and recreation of wetlands.

All the NRA functions, with the exception of coastal responsibilities, are managed within a catchment framework [Chandler, 1994]. In order to integrate these functions for planning purposes, the same approach is necessary. Catchment management plans provide the framework for the NRA to resolve potential conflicts between its own functions, as well as the activities of others. Plans are devised through an assessment of current and potential uses of water resources within particular catchments, and of their environmental implications, in order to produce prescriptions of priority issues and necessary remedial action. After a period of consultation, the final plans contain a phased programme of specific management actions and objectives. The use-related objectives incorporated into catchment management plans mean that they are the ideal method by which to implement forthcoming Statutory Water Quality Objectives (Section 4.4.2.4).

The NRA is soon to be subsumed into the government's new Environment Agency, which will also replace HMIP and the Waste Regulation Authorities, under the forthcoming Environment Act 1995 [*Environmental Law Bulletin*, 1994b]. The reasons for creating a new Agency are to remove the current areas of overlap and potential conflict between the present agencies, to ensure that decisions about pollution control select the best practicable environmental option, and to provide a comprehensive and coherent regulatory focus [DoE, 1991c].

However, an early draft of the Bill to introduce the Agency aroused much controversy. There were questions about whether particular clauses would weaken existing environmental laws [*ENDS Report*, 1994f]. Two aspects caused some concern and are also of relevance here. First, a clause was included which places a duty on the Agency to "take into account the costs which are likely to be incurred, and the benefits which are likely to accrue," [*ibid.*, p.23] in deciding on a course of action. There were fears that this clause would leave the Agency open to judicial review, whereby dischargers might contest consent conditions in the courts on the grounds that the Agency had not fulfilled its cost-benefit duty. The government's response to this criticism was to change the wording of the clause so that, in the opinion of legal advisors, such a challenge would not be allowed [Brown, 1994a]. The problem still remains, however, that the Agency must demonstrate that the benefits of a particular action outweighs the costs.

The second aspect of concern related to the limited conservation duty placed on the Agency [*ENDS Report*, 1994f]. In the early draft, it was obliged to "have regard to the desirability" [*ibid.*, p.23] of conserving and enhancing natural beauty and wildlife, rather than the present duty on the NRA to *further* nature conservation interests. After intense criticism of this aspect of the draft the government inserted a new clause in the Bill which imposes a duty on the Agency to further nature conservation [Brown, 1994b]. Nevertheless, this will not apply to the Agency's pollution control functions, leaving it in the position of having a weaker conservation duty than the water companies and internal drainage boards which it will regulate [*ENDS Report*, 1994g]. More positive aspects of the Environment Bill include the introduction of the power to serve enforcement notices for breach of discharge consent conditions, and the abolition of the tripartite sampling requirement for water pollution prosecutions. The Agency is expected to become operational in April 1996.

4.4.2.2 Water Protection Zones

Under the WRA, an area may be designated as a Water Protection Zone (WPZs) by order of the Secretary of State, in response to an application by the NRA [Ball and Bell, 1994]. The order confers on the NRA the power to prohibit or restrict activities which are likely to result in the pollution of controlled waters. Precise publicity arrangements are defined, which require the NRA to inform all local authorities and water undertakers in the area concerned, so that they may comment. The Secretary of State has the power to hold a public inquiry if necessary. If the proposed WPZ is in England, MAFF must be consulted. The provisions are essentially those contained in COPA, but never brought into force. The main difference is contained in a new provision which states that WPZs may not be used to control nitrate losses from agricultural land, since this function is deemed to be provided by the designation of Nitrate Sensitive Areas (Section 4.4.2.3).

No WPZs have so far been designated, yet they provide an ideal legal instrument for reducing diffuse pollution of both surface water and groundwater [Ball and Bell, 1994]. In particular, they could be used to address eutrophication at the catchment level [NCC, 1991a]. There is, however, no provision in the Act for compensation payments, although an order may include such provisions as the Secretary of State deems appropriate [Ball and Bell, 1994]. The NRA, which must initiate the process of designation, appears to possess an ambivalent attitude to WPZs. Towards the end of 1992, it launched its groundwater protection policy, which carries no statutory weight, rather than pressure the government to introduce WPZs for this purpose [*ENDS Report*, 1992c]. Yet in the *NRA Water Quality Strategy* [NRA, 1993b], in relation to the prevention of diffuse sources of pollution, it is stated:

> "The NRA will continue to... develop a policy for the identification of Water Protection Zones to be designated under the Water Resources Act, including Groundwater Protection Zones, and take account of these in catchment planning." [*ibid.*, p.11]

The only application by the NRA for WPZ designation is not, however, directed at diffuse, but industrial, sources of pollution [*ENDS Report*, 1994b]. The proposal is for the River Dee catchment, where a release of phenol in 1984 tainted the drinking water supply to two million people, costing £20 million in remedial measures and £400,000 in compensation. The NRA believes that WPZ designation, which would require companies to carry out risk assessments and take precautionary measures to prevent accidental spillages, would reduce the risk of future pollution and protect water supplies.

The reluctance of the NRA to utilise WPZs for the prevention of diffuse pollution may be a reflection of its unwillingness to press government for implementation of mandatory schemes for which there is no compensation payable, and which focus on sources that are "less easily controlled... [and] cannot be reliably sampled by conventional means" [NRA, 1993b p.11]. With the prospect of widespread compulsory regulation of nitrate losses from farmland under the EC Directive on Nitrates (Section 4.4.3.1), it is not surprising that the implementation of WPZs to control other diffuse losses has always been strongly opposed by agricultural interests [Ball and Bell, 1994]. The case for mandatory controls on diffuse sources of phosphorus is not easily made under these circumstances. It is unlikely therefore that WPZs, in their current form, will be used to address eutrophication, despite their evident applicability.

4.4.2.3 Nitrate Sensitive Areas

Provisions for the designation of Nitrate Sensitive Areas (NSAs) were belatedly included in the 1989 Act, as a result of action against the British government for non-compliance with the EC Directive on Drinking Water [Ball and Bell, 1994]. The EC Directive on Nitrates (Section 4.4.3.1) was in draft form at that time, and anticipation of its requirements may also have had some bearing on the government's decision.

Upon application by the NRA, MAFF may make an order designating land as an NSA, in order to prevent or control entry of nitrate into controlled waters as a result of agricultural practices [Ball and Bell, 1994]. The NRA must identify both the waters likely to be affected and the agricultural land to which nitrate losses may be attributed. The WRA provides for two types of order, imposing either voluntary or mandatory controls. Under voluntary orders, a management agreement is drawn up with MAFF, and compensation is payable for income foregone as a result of prohibitions and restrictions on activities. A mandatory order may require positive works to be carried out, but compensation may be payable to anyone affected by the obligations. NSAs are designed primarily for the protection of groundwaters, but they may also be used for surface waters, providing only agricultural activities are involved. In keeping with stated policy, the government has so far only used the voluntary approach, but it has conceded that compulsory measures may be necessary if response from farmers proves to be ineffective [Howarth and Somsen, 1991].

A pilot scheme was introduced by the Nitrate Sensitive Areas (Designation) Order 1990 [Ball and Bell, 1994]. Its objective is to select specific areas where nitrate concentrations in water sources exceed, or are at risk of exceeding, the limit of 50mg/l specified in the Directive on Drinking Water. Ten NSAs were initially selected, within which farmers may enter either a 'basic' scheme aimed at limiting fertiliser usage, or a 'premium' scheme which involves the conversion of arable land to unfertilised or lightly fertilised grassland [*ENDS Report*, 1993a]. Nine Nitrate Advisory Areas were also selected which are subject to intensive campaigns providing farmers with free advice on methods of reducing nitrate leaching. No compensation is paid to such farmers.

Proposals to establish up to thirty new NSAs, issued by MAFF in 1993, introduced two major changes to the scheme [*ENDS Report*, 1993a]. First, the programme is focused more on land use changes similar to measures in the original premium scheme. Second, the designated areas will be confined to small zones around boreholes, rather than entire catchments. Details of the finalised scheme revealed that major concessions were included in order to make the scheme more attractive to farmers [*ENDS Report*, 1993c]. It appears that changes have been made in order both to counteract the poor response to the premium scheme in the pilot NSAs, and to encourage the higher take-up rate necessary in smaller NSAs. The new scheme came into effect in autumn 1994.

The relevance of the NSA scheme to lake eutrophication is limited by several factors. First, it relates entirely to nitrate losses. Although several standing waters in England, such as some of the Shropshire-Cheshire meres, are thought to be nitrogen-limited, this is due to the presence of geological sources of phosphorus [Reynolds, 1979], rather than excessive diffuse agricultural loadings. Second, at present, NSAs are targeted at losses to groundwater, rather than surface waters. Third, nitrate fertiliser application rates may be adjusted without necessarily changing the corresponding amount of phosphorus [Chalmers, Kershaw and Leech, 1990].

Nevertheless, the various land use options available under the scheme, such as converting arable or intensive grassland to extensive grassland, and the use of autumn, rather than spring-sown crops, would be beneficial in reducing erosion, and therefore transport of particulate phosphorus. The wider requirements of the Directive on Nitrates (Section 4.4.3.1) may well improve the impact of controls on agricultural losses of both nitrogen and phosphorus.

4.4.2.4 Statutory Water Quality Objectives

The power to introduce Statutory Water Quality Objectives (SWQOs) under the WRA represents an important innovation, in that it provides the potential for establishing a coherent national policy on water quality. SWQOs will be legally binding targets for water quality to be achieved within certain timespans, and may form the basis for setting discharge consent conditions. An additional General Quality Assessment Scheme will facilitate a periodic statement to be made on the quality of controlled waters.

Various processes are involved in the determination of SWQOs [Ball and Bell, 1994]. The initial stage consists of the definition of Use Classifications (UCs). Six have been agreed, and are illustrated in Table 4.3. The 'Special Ecosystem' UC applies to sites requiring protection for nature conservation reasons. Classification regulations then need to be introduced in order to set the standards which waters must reach in order to fall within certain categories.

Table 4.3: Use Classifications for the Setting of SWQOs

Based on NRA [1992b]

Use Classes	Rivers and Canals	Lakes	Estuaries	Coastal Waters	Ground waters
Fisheries Ecosystem Classes 1 - 6	*	*	*	*	*
Abstraction for Potable Supply	*	*			*
Industrial and Agricultural Abstraction	*	*	*	*	*
Water Sport Activity	*	*	*	*	
Special Ecosystem	*	*	*	*	
Commercial Harvesting of Fish and Shellfish for Human Consumption			*	*	

Once UCs are established by appropriate Regulations, the Secretary of State will set SWQOs for a number of shortlisted pilot catchments, incorporating EC standards where relevant [Ball and Bell, 1994]. This process includes three months' publicity of the proposed SWQOs, allowing representations and objections by persons affected by them, including dischargers, to be considered, and any modifications made. A public local inquiry may be held. After SWQOs are formally confirmed for a particular catchment, the NRA and the Secretary of State are under a duty to use their powers to achieve the required standards, if practicable. For the NRA, such powers include those related to setting consents, as well as those concerned with enforcement, preventative and remedial actions. The NRA has intended to implement SWQOs through catchment management plans, since both are based on the concept of use-related objectives [Ayton, 1994].

Classification Regulations are currently in operation in relation to EC Directives on drinking water, bathing waters and dangerous substances [Ball and Bell, 1994]. A consultation paper on draft regulations for the 'Fisheries Ecosystem' classification was issued by the DoE during late 1993 [*ENDS Report*, 1993e]. When the Regulations came into force in May 1994, the title had been altered to Surface Waters (River Ecosystem) (Classification) Regulations [Statutory Instrument 1994/1057]. They contain a Schedule referring to five, rather than six, Classes, and compliance values for only eight parameters, namely, dissolved oxygen, biochemical oxygen demand (BOD), total ammonia, pH, un-ionised ammonia, dissolved copper and total zinc. The catchments selected by the DoE as pilot areas were recently announced, but the SWQOs set for them will not apply until after the three month consultation period, or, possibly, a public inquiry.

In some circumstances, the Regulations grant considerable leeway to the NRA in assessing whether the results obtained from samples indicate compliance with SWQOs [*ENDS Report*, 1993e]. For instance, where the NRA believes that high algal levels are responsible for elevated BOD results, it may ignore the readings. In other words, non-compliant waters will appear to meet the standards where failures are the result of eutrophication. Similarly, the Regulations allow non-complying values for pH, zinc and copper to be disregarded in cases where acid rain or old mine workings are thought to be contributory factors. The NRA claims, however, that the system will protect such exempted rivers from further deterioration. The DoE state that the first tranche of SWQOs would be set only "to consolidate existing objectives and not to take big steps forward to higher environmental standards" [*ENDS Report*, 1993b p.35].

The sections of the WRA dealing with SWQOs did not include a timetable for their implementation. The process appears to be considerably delayed, however, not least by concessions to the water industry because of the likely cost implications [*ENDS Report*, 1992b]. The timescale for achieving SWQOs once they are applied has also lengthened from the three to five years originally envisaged by the NRA in 1993, to a more long-term five to fifteen years, according to the DoE. The extent to which the implementation of SWQOs will achieve real environmental improvement is also marred by the adoption of 1990 river data as the baseline for maintaining water quality [*ENDS Report*, 1993b]. In that year, water quality was the worst on record up to that time, partly as a result of drought. The acceptance of 1990 as a baseline therefore ignores the deterioration which occurred during the 1980s.

The initial round of SWQOs applies only to rivers, and not to lakes, estuaries or coastal waters. The development of a classification system for lakes is being addressed through the NRA's national R & D programme, and is discussed in Section 3.2. Criteria for other UCs are still under development, and it may be some time before these are addressed [*ENDS Report*, 1993b]. Those for 'Special Ecosystem' UC will include nutrients, but EN believes that the deferral in setting limits will result in continued deterioration of eutrophicated sites, and delayed recovery. The accelerated introduction of SWQOs has been suggested not only by the Chief Scientist of the NRA, Dr Jan Pentreath, who believes it is the only way to fill the large credibility gap in government policy on this issue [*ENDS Report*, 1994a], but also by the Chairperson of the NRA, Lord Crickhowell, who fears that otherwise nature conservation interests may not be addressed under the new Environment Agency [M. Gibson, EN, pers. comm.].

4.4.3 EC Directives

The impact of the EC on water quality legislation has been highly significant over the last few years [Ball and Bell, 1994]. Water protection was singled out as a priority issue in the First Action Programme on the Environment in 1973, and a large number of Directives, covering a wide range of subjects, have ensued. Objectives are attained either through the adoption of general emission standards, or by the imposition of specific quality criteria according to water use.

Whether general or specific in their nature, Directives usually share certain features [Hughes, 1992]:

(a) use of imperative (I) values, which must be observed, and guide (G) values, which Member States should aim to achieve
(b) discretion is normally vested in Member States to select waters for designation
(c) compliance with a Directive should not lead to deterioration of water quality
(d) a competent national authority is required to sample, analyse and inspect waters
(e) provisions are established for updating I and G values

Several Directives contain objectives for nutrients, but very few are aimed specifically at the issue of freshwater eutrophication. Directive 75/440 on the Quality of Surface Water Intended for the Abstraction of Drinking Water specifies 50mg/l (I) and 25mg/l (G) for NO_3, and 0.4 to 0.7mg/l (G) for P_20_5 [National Society for Clean Air and Environmental Protection (NSCA), 1991]. Directive 76/160 Concerning the Quality of Bathing Water, which relates to fresh as well as coastal waters, states that concentrations of nitrates and phosphates are to be checked when an inspection shows that the quality of the water has deteriorated. Ammonia and Kjeldahl nitrogen must be determined where there is a tendency towards the eutrophication of the water, but no I or G values are given for these parameters.

Directive 78/659 on the Quality of Fresh Waters Needing Protection or Improvement in Order to Support Fish Life lays down quality requirements for salmonid and cyprinid waters, but since there is no *duty* to designate the main effect has been to increase sampling and monitoring of relevant waters [Ball and Bell, 1994]. For lakes of less than 18m mean depth, total phosphorus limits of 0.2mg/l for salmonid, and 0.4mg/l for cyprinid waters, "may be regarded as indicative in order to reduce eutrophication" [Council of the European Communities, 1978 p.7]. In the case of deeper lakes, an areal loading formula is recommended. Directive 80/778 Relating to the Quality of Water Intended for Human Consumption sets quality standards for some sixty substances, including 50mg/l (I) and 25mg/l (G) for NO_3, and 5,000μg/l (I) and 400μg/l (G) for P_2O_5 [NSCA, 1991].

Three further Directives deal directly with nutrient levels in freshwaters, and specifically address the problem of eutrophication as part of a wider concern for water quality. These are the Directives on Nitrates, and Urban Waste Water and the proposed Directive on the Ecological Quality of Surface Waters.

4.4.3.1 Directive on Nitrates

The Directive Concerning the Protection of Waters Against Pollution Caused by Nitrate From Agricultural Sources 91/676 requires Member States to identify waters affected by nitrate pollution [*ENDS Report*, 1992a]. These are defined as being surface freshwaters or groundwaters intended for drinking which are likely to contain more than 50mg/l nitrate, or any freshwater lakes, estuaries or coastal waters which are liable to suffer from eutrophication if protective action is not taken. For the purpose of the Directive, pollution is defined as losses of nitrogen compounds which result in "hazards to human health, harm to living resources and to aquatic ecosystems, damage to amenities or interference with other legitimate uses of water" [Council of the European Communities, 1991b p.2], and eutrophication as the enrichment of water by nitrogen compounds.

Member States must designate as vulnerable zones those areas of land which drain into such waters and which contribute to pollution [*ENDS Report*, 1992a]. An action programme must be drawn up for vulnerable zones to include limitations on application of nitrogen fertiliser, restrictions on disposal of manure and sewage sludge to land, and adequate storage of animal wastes. Monitoring plans are to be implemented in order to assess the effectiveness of the action programme. If it becomes apparent that the programme is not meeting objectives, additional measures must be taken.

This Directive adopts a far more interventionist system than that which is operated through the provisions on NSAs [Ball and Bell, 1994]. It carries immense implications for the agricultural sector, since it will mean an extension of designated areas, and will probably necessitate the use of mandatory orders. The government has estimated that nearly two million hectares of land may need to be designated [*ENDS Report*, 1992a].

Proposals by DoE and MAFF to designate 72 Nitrate Vulnerable Zones (NVZs) were produced in May 1994, but they appear to fall short of the requirements of the Directive in several respects [*ENDS Report*, 1994c]. The area covered by the proposed NVZs is a mere 650,000 hectares, and extend over the catchment areas of groundwaters and surface waters used for public supply. The discrepancy between the area of land proposed for designation and the government's own estimate of 1992 may be linked to a narrow interpretation of the Directive, which requires *all* areas which exceed the limit to be designated, not just those used for drinking water supply. Nitrogen-limited lakes, estuaries and coastal waters subject to eutrophication will remain unprotected through the government's interpretation. Additionally, the NRA has reservations about whether the proposed measures will ensure compliance in those areas which will be designated. Action programmes for each NVZ need to be drawn up by December 1995, and designations reviewed every four years. Meanwhile, the NRA will be monitoring nitrate levels, and may request tighter controls if the measures prove ineffective.

Although this Directive is specific to nitrate, and is therefore of limited relevance to this study for the reasons outlined in Section 4.4.2.3, the strength of the measures imposed are likely to elicit an indirect effect on phosphorus losses. Additionally, provided the Directive is implemented in full in this country, the identification of nitrogen-limited lakes which are, or may become, eutrophicated, and designation of the areas of land draining into them, may well address this aspect of the problem. But, with respect to the majority of eutrophicated lakes and reservoirs which are phosphorus-limited, more specific measures are necessary.

4.4.3.2 Directive on Urban Waste Water Treatment

The Directive Concerning Urban Waste Water Treatment 91/271 was adopted by the Council of the EC in May 1991 [*ENDS Report*, 1994d]. It sets new criteria for sewage effluents discharged to surface waters, which depend on the size of the STW and the sensitivity of the receiving water. The Directive represents the first attempt to deal directly with nutrient enrichment of freshwaters by sewage effluent.

There are basically three main measures contained in the Directive [*ENDS Report*, 1994d]. First, sewage discharges from STWs serving over 2,000 population equivalent (p.e.) must be treated to a minimum standard, except for those to 'less sensitive areas'.

Second, disposal of sewage sludge to sea must be phased out. Third, sewage discharges from STWs serving over 10,000 p.e. affecting designated 'sensitive areas' must be treated to remove phosphorus and/or nitrogen.

The third requirement is of direct relevance to this study and will be discussed in further detail. 'Sensitive areas' are defined in an annex as natural freshwater lakes, estuaries, or coastal waters which are, or may become, eutrophicated, or surface waters used for the abstraction of drinking water which could contain more than 50mg/l nitrate [Council of the European Communities, 1991a]. For STWs discharging into such waters, and serving over 10,000 p.e., effluent must comply with values for concentration, or percentage reduction, for total phosphorus and/or total nitrogen, depending on the local situation. Table 4.4 details the requirements for discharge to sensitive areas.

Table 4.4: Values for Concentrations and Percentage Reductions of Phosphorus and Nitrogen in Effluent Discharged to Sensitive Areas

[Council of the European Communities, 1991a]

Parameter	Population Equivalent*	Concentration	Minimum Percentage Reduction
Total Phosphorus	10,000 to 100,000	2 mg/l	80
	more than 100,000	1 mg/l	
Total Nitrogen	10,000 to 100,000	15 mg/l	70 to 80
	more than 100,000	10 mg/l	

* 1 p.e. equals the organic biodegradable load having a five-day biochemical oxygen demand of 60 g of oxygen per day

The selection of sites for designation as sensitive areas was left to Member States, as is usually the case. In the UK, this led to considerable controversy, and served to highlight the disparity between the government's assessment of the extent of eutrophication, and those of EN and the NRA [*ENDS Report*, 1991a]. In evidence to the House of Lords Select Committee on the European Communities, which considered the implications of the Urban Waste Water Treatment (UWWT) Directive, the DoE maintained that freshwater eutrophication was restricted to a few localised sites, whereas EN identified over one hundred SSSIs which may be affected, and the NRA stated that 169 sites in England and Wales experienced problems with blue-green algae [House of Lords, 1991].

In May 1994 the government announced that a total of 33 sites in England and Wales would be designated under the Directive as sensitive areas [*ENDS Report*, 1994d]. At least forty STWs will require phosphorus and nitrogen removal by the end of 1998.

All Member States were required to introduce regulations implementing the Directive by 30 June 1993, but it was not until 30 November 1994 that the Urban Waste Water Treatment (England and Wales) Regulations 1994 came into force [*Environmental Law Bulletin*, 1994c]. Effluent standards will be regulated largely through the NRA discharge consent system. In a bid to reduce the costs of tertiary treatment to the water industry, the DoE has decided to adopt the percentage reduction regime, rather than the alternative uniform discharge limits [*ENDS Report*, 1994a]. The requirement to provide tertiary treatment will not apply, however, where the NRA is satisfied, as a result of monitoring, that the minimum percentage reduction of the *overall* load entering the catchment of a sensitive area is at least 75% for total phosphorus and at least 75% for total nitrogen. In other words, where a number of STWs are involved the average percentage reduction will suffice, and at a level lower than that stipulated by the Directive.

In response to a consultation paper on the Regulations, the Institution of Water and Environmental Management (IWEM) considered that an essential element, missing from the UWWT Directive, was reference to water quality objectives on a site-specific basis [IWEM, 1991a]. The Institution supported nutrient reduction where there might be a risk of eutrophication, but it felt that concentrations in receiving water were more important. The NRA is concerned about the adoption of the percentage reduction approach for similar reasons [*ENDS Report*, 1994a]. Its Chief Scientist, Dr Jan Pentreath, told the DoE that its strategy for implementation may bring about some environmental improvements, but that it could not act as a substitute for uniform discharge limits for STWs, or for the introduction of SWQOs.

Nine other sites recommended by the NRA for designation as sensitive areas were rejected by the DoE following representation from the water companies [*ENDS Report*, 1994d]. These and a further twenty sites will be monitored by the NRA and re-evaluated for designation by the end of 1997. EN is concerned that the designations will not protect a large number of SSSIs suffering from eutrophication. In a briefing for the House of Lords debate on the Directive it was stated:

> "The potential benefit of the Directive to nature conservation has been removed by setting a size limit of sewage works where phosphate stripping should be installed... Small sewage discharges or a series of works along a river cause most of the problem. Indeed, the Directive would not require any of the phosphate stripping presently operated by Anglian Water (8 works) or by North West Water at Esthwaite or Windermere in the Lake District." [EN, 1991 p.2]

None of the natural lakes included in this study receives effluent from individual STWs serving over 10,000 p.e. In a small number of cases the combined p.e. of STWs discharging into rivers feeding these lakes approaches or exceeds 10,000, as in the case of Loe Pool (Section 6.1.1). This site will be one of those monitored by the NRA and reassessed for designation. The majority of the remaining sites are influenced by discharges from STWs which fall far short of the required minimum p.e., and will remain unaffected by the Directive's implementation.

4.4.3.3 Proposed Directive on the Ecological Quality of Surface Waters

A draft Directive on the Ecological Quality of Surface Waters was published by the EC in August 1994 [*ENDS Report*, 1994e]. The proposal is intended to supplement the Nitrate and UWWT Directives, and the draft Directive on integrated pollution control, which deal with inputs from agriculture, sewage treatment and industry, respectively. The new proposals aim to provide a framework for the control of pollution from point and diffuse sources, so as to maintain and improve the ecological quality of all surface waters. The Commission hopes that the Directive will be enforced by the end of 1997, preparing the ground for the repeal of several existing Directives on water quality.

A monitoring and classification system of surface waters is required in the first stage, followed by a qualitative and quantitative assessment of point and diffuse sources of pollution which may affect ecological quality [*ENDS Report*, 1994e]. Operational targets must then be defined, and an integrated programme designed to achieve these targets should be implemented. The ultimate aim of the Directive is for all Community waters to achieve good ecological quality, which is broadly defined as that which allows aquatic ecosystems to be self-sustaining. A list of parameters are provided for the determination of good ecological quality, which includes the provision that the diversity of populations of aquatic flora and fauna should resemble or reflect that of similar water bodies with insignificant anthropogenic disturbance. On this basis alone, eutrophicated inland waters will require significant remedial action to achieve good ecological quality. Waters may also be classified as being of high ecological quality, that is, those which are not significantly affected by human activities.

Member States are granted a great deal of discretion in setting operational targets for good ecological quality, and there is no deadline for the achievement of the ultimate aim [*ENDS Report*, 1994e]. The UK's system of SWQOs, when it is introduced fully, is likely to provide the framework for the integrated programme required by the Directive, but several difficulties are posed which will probably necessitate considerable negotiation time before the final version comes into force. First, the definition of good ecological quality is provided in an annex containing a large number of rather vague terms such as 'resemble' and 'normal', which may cause some difficulty in legal and scientific interpretation. Second, in the UK, a system of biological assessment is not sufficiently refined to be applied to such a wide-ranging programme of monitoring. Third, the Directive tends to assume that the biological effects of all pollutants is comprehensively understood, which is not the case.

Despite the areas of contention in this Directive, and the degree of discretion given to Member States in its application, it may provide a more comprehensive framework for water quality objectives than any other piece of EC legislation. In terms of eutrophication, point *and* diffuse sources of nutrients will need to be identified, monitored and addressed. Rather than adopting a sectorial approach, the Directive attempts to deal with all aspects of pollution of surface waters, and, as such, may provide the closest approximation to eutrophication legislation.

4.5 Synthesis

As this Chapter demonstrates, a wide range of laws and policies may be applied to the reduction of nutrient inputs to eutrophicated freshwater SSSIs. Nature conservation laws are applicable in the sense that such sites are theoretically protected through the SSSI system. Intensive agricultural practices are responsible for significant losses of nutrients, so that legislation aimed at extensification is also relevant. Nutrient enrichment is, essentially, a water quality issue, but laws in this area are not yet sufficiently developed to address the problem in a comprehensive manner. The lack of specific laws to deal with the problem of nutrient enrichment means that an integration of aspects of legislation relating to nature conservation, to agriculture, and to water quality (Fig.4.4) is necessary in order to assess the extent to which the problem is addressed.

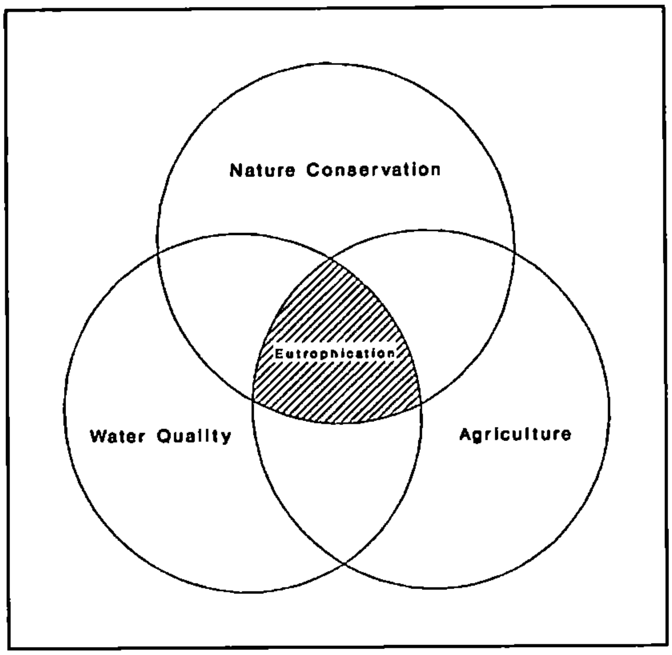

Fig.4.4: The Spheres of Legislation Applicable to Eutrophication of Freshwater SSSIs

Among the criticisms which may be levelled at the SSSI system of nature conservation, the most compelling is that it has failed to protect the integrity of a substantial number of sites, not only from immediate damage in the form of PDOs and development, but also from long-term external influences. It is not surprising, therefore, that nature conservation legislation has proved inadequate to prevent eutrophication of freshwater SSSIs. Until very recently, agricultural laws and policies have tended to operate almost in direct opposition to nature conservation interests, in the face of which the SSSI system has proved vulnerable. The integration of environmental objectives into agricultural policy, whether as an indirect benefit of commodity surplus reduction, or as an end in itself, is gradually changing the relationship between nature conservation and agriculture. Increasingly, legislation related to extensification of agriculture includes an environmental imperative, a trend which will inevitably extend to nutrient conservation.

Although it has long been recognised that catchment land use changes may exert considerable influence on nutrient loadings to freshwater sites, the SSSI system is not designed to protect water quality at the catchment level. The responsibility for management in relation to nutrient losses to freshwater SSSIs therefore lies with the NRA, rather than with EN. The present regulatory system operated by the NRA contains the necessary mechanisms to address point sources of nutrients, but lacks the legislative power to reduce inputs to the majority of eutrophicated sites. The system is not constructed to deal effectively with diffuse sources of nutrients, since these tend to arise from the cumulative effect of many activities.

Finally, the common perception of eutrophication as purely an issue affecting amenity value and water supply tends to limit the extent to which nature conservation interests may be served by current legislation. The insidious nature of eutrophication in its early stages, and the prevalence of diffuse sources of nutrients, combine to obscure the problem until the appearance of conspicuous algal blooms. By that time, aquatic community structure is often severely disrupted, and restoration may pose serious difficulties. Eutrophication therefore needs to be addressed at an early stage, in order both to protect nature conservation interests, and to avoid the necessity for restoration measures. Current legislation is not, however, sufficiently focused on the principle of precautionary action to achieve this aim.

Chapter 5

Methods

A critique of the methodology adopted in this study, that is, the use of export coefficients to predict phosphorus loadings on shallow eutrophicated lakes and reservoirs, and of the Vollenweider-OECD model to evaluate the effects of reductions on trophic status, was carried out in Chapter 3. Conclusions extracted from the review will be used in applying the models to the sites under assessment.

5.1 Selection of Sites for Evaluation

As part of a monitoring programme of freshwater SSSIs, the Nature Conservancy Council for England (English Nature: EN) maintains a database on those sites thought to be affected by eutrophication. This information was initially collated in 1991 through a questionnaire sent to EN Conservation Officers (COs). In particular, COs were asked whether the special interest of the site had been affected, and to identify nutrient sources where known. This database, covering rivers and canals, as well as lakes, eventually included over one hundred sites. It was used to compile a draft list of 25 lakes for inclusion in this project. The sites are listed in Table 5.1, and full site descriptions of those selected for assessment are produced in Chapter 6.

It was apparent that some sites were, however, unsuitable for assessment. After site visits and meetings with the relevant COs, the following lakes were fully or partially excluded from evaluation:

Upton Warren Pools. Two of these lakes were created by subsidence associated with salt extraction, while a third is a worked out gravel pit [NCC, 1984d]. Although a small stream runs through the site, this is not connected *via* surface flow to any of the Pools [Ordnance Survey, 1991b]. It has to be assumed, therefore, that their water budget is almost entirely contributed by groundwater. As explained in Section 3.1, this factor renders the site unsuitable for the application of export coefficients.

Table 5.1: Draft List of SSSI Lakes and Reservoirs for Assessment

Site	County	SSSI Name (if different)
Aqualate Mere	Staffordshire	
Barton Broad	Norfolk	Ant Broads and Marshes
Bay Pond and Leigh Place Pond	Surrey	Godstone Ponds
Chapel Mere	Cheshire	
Chasewater	West Midlands	Chasewater Heath
Comber Mere	Cheshire	
Elter Water	Cumbria	
Esthwaite Water	Cumbria	
Grafham Water	Cambridgeshire	
Hammer Pond	Surrey	Thursley, Hankley and Frensham Common
Hedgecourt Lake	Surrey	Hedgecourt
Loe Pool	Cornwall	
Malham Tarn	North Yorkshire	Malham-Arncliffe
Norbury Meres	Cheshire	
Pettypool	Cheshire	Pettypool Brook Valley
Pitsford Reservoir	Northamptonshire	
Quoisley Meres	Cheshire	
Rutland Water	Leicestershire	
Semerwater	North Yorkshire	
Slapton Ley	Devon	
Sutton Park	West Midlands	
Tabley Mere	Cheshire	
Tatton Mere	Cheshire	
Upton Warren Pools	Hereford and Worcester	
Windermere	Cumbria	

Bay Pond and Leigh Place Pond. Bay Pond is upstream of Leigh Place Pond and may, therefore, as explained in Section 3.1, exert an influence on its nutrient loading. Added to this, opinion has been expressed to the effect that the status of Leigh Place Pond as an SSSI is debatable [pers. comm., Peter Tinning, EN, Lewes]. It was therefore decided that only Bay Pond would be assessed.

Hammer Pond. Upstream of this site lie a series of four small lakes [Ordnance Survey, 1988a], which may influence the phosphorus budget of Hammer Pond. For this reason the site was excluded from assessment.

Sutton Park. The park incorporates the headwaters of two streams which have been dammed at intervals to form six main pools [NCC, 1987f]. Their hydrology is further complicated by the fact that the catchment outside the park is entirely urban, so that the natural drainage has been severely disrupted. For these reasons, a full assessment of the lakes of Sutton Park will not be undertaken. However, it was felt that sufficient indications of phosphorus sources was available to produce recommendations.

Chasewater. The Chasewater Heaths SSSI lies adjacent to Chasewater reservoir and is of special interest for the wet and dry heathland communities and associated valley mires [NCC, 1987c]. Although the SSSI boundary encompasses an area along the shore of the reservoir, this site is essentially terrestrial. For this reason it was felt that it was outside the scope of this project.

Elterwater, Esthwaite Water and Windermere. These sites were completely excluded from the study. The Institute of Freshwater Ecology (IFE) at Windermere was "not in a position to release unpublished data" on these lakes [letter from Professor A.D. Pickering, IFE, Ambleside, 20th July 1992]. Additionally, the IFE's own work in modelling the water quality of these lakes has produced strategies for the reduction of nutrient loading, some of which are being implemented [e.g. Stocks *et al.*, 1994].

5.2 Data Collection

The use of the export coefficient approach to predicting total phosphorus (TP) loadings necessitated the collation of particular data for each lake. The extent of the catchment, its land use and livestock levels, together with details of consented discharges, provided information on the sources of phosphorus. For diffuse sources, each type of land use and livestock was assigned an appropriate phosphorus export coefficient, based on literature values. Point sources, identified through information on consented discharges, were treated similarly, in that the level of phosphorus normally associated with such effluent was used to calculate the amount from each discharge. The sum of all known sources within the catchment was taken to be the total potential TP loading on a particular lake. In order to predict the impact of such loadings on trophic status, according to the Vollenweider-OECD model, data on mean depth and hydrological residence time were obtained or calculated for each lake.

In order to assess the nutrient enrichment history of individual lakes, a visit was made to each site and to the relevant EN County offices. The COs concerned were interviewed and the scientific files of the sites searched for data and general information. Data on pumped inputs to the Ruthamford reservoirs was provided through an interview with the Quality Standards Manager and Senior Scientist (Limnology) of Anglian Water Services Ltd. An extensive literature review was also undertaken.

5.2.1 Catchment Definition

Using Ordnance Survey 1:50000 Landranger maps, the extent of the catchment of each lake was determined by inspection of contour lines and spot heights. The area was then calculated by the cut and weigh method [Lind, 1979].

5.2.2 Land Use

The extent of each type of land use within each catchment was obtained through the Countryside Survey Information System, developed by the Institute of Terrestrial Ecology (ITE) at Merlewood Research Station, Cumbria. The facility utilises satellite imagery coordinated with ground truth information [Bunce *et al.*, 1992], and data are available on the basis of the Ordnance Survey kilometre square grid. This method of obtaining land use data was chosen since it was likely to generate more accurate results than the use of Parish Returns. A comparison of Fig.5.1 and Fig.5.2 illustrates the fact that groups of kilometre squares can be made to coincide roughly with catchment boundaries, whereas most parishes bear little relation. A recent comparison between the remotely sensed land use data and those obtained through ground survey concluded that, for many applications, the former is perfectly adequate [Cherrill *et al.*, 1995]. One limitation of the method, which is relevant to this study, is that linear and small areal features, such as hedges and small woodlands, are not consistently recognised.

Blocks of kilometre squares were selected to coincide as closely as possible with catchment boundaries (Fig.5.1). Coordinates for each block were entered into the database and land use information produced. These data were summed by land use and the totals adjusted, where necessary, to the percentage area represented by the actual catchment, compared to that of the total kilometre squares. Land use categories thus produced by the ITE System were: urban land, roads and railways, rivers, open water, woodland, arable, leys (temporary pasture), permanent pasture, and rough grazing.

5.2.3 Livestock Levels

Data on livestock levels within each catchment were obtained through the MAFF County and Parish Summaries of Agricultural Returns, held at the Public Record Office in Kew, London. The accuracy of these data, and their applicability to catchments, is discussed in Section 3.2. Using Ordnance Survey 1:100000 Local Government and Parliamentary Constituency Boundaries maps, parishes which lay within or impinged upon each catchment were delineated and listed by county. In instances where less than 10% of a parish lay within a catchment, that parish was disregarded. For all other parishes, the total area of agricultural land, and the number of cattle, sheep, pigs, chickens and other poultry were noted. The number and types of holdings in each parish were also recorded, so that the presence of any large, intensive units could be detected. In order to establish whether such units were situated within particular catchments, a search was carried out using Electronic Yellow Pages, and, where possible, their position located.

The figures for each parish were summed by livestock type and the totals reduced (or in a few cases increased) by the percentage of agricultural land in the catchment, compared to that of the combined parishes (Fig.5.2). In the case of parishes which lay completely within the boundary of a catchment, the data were not adjusted.

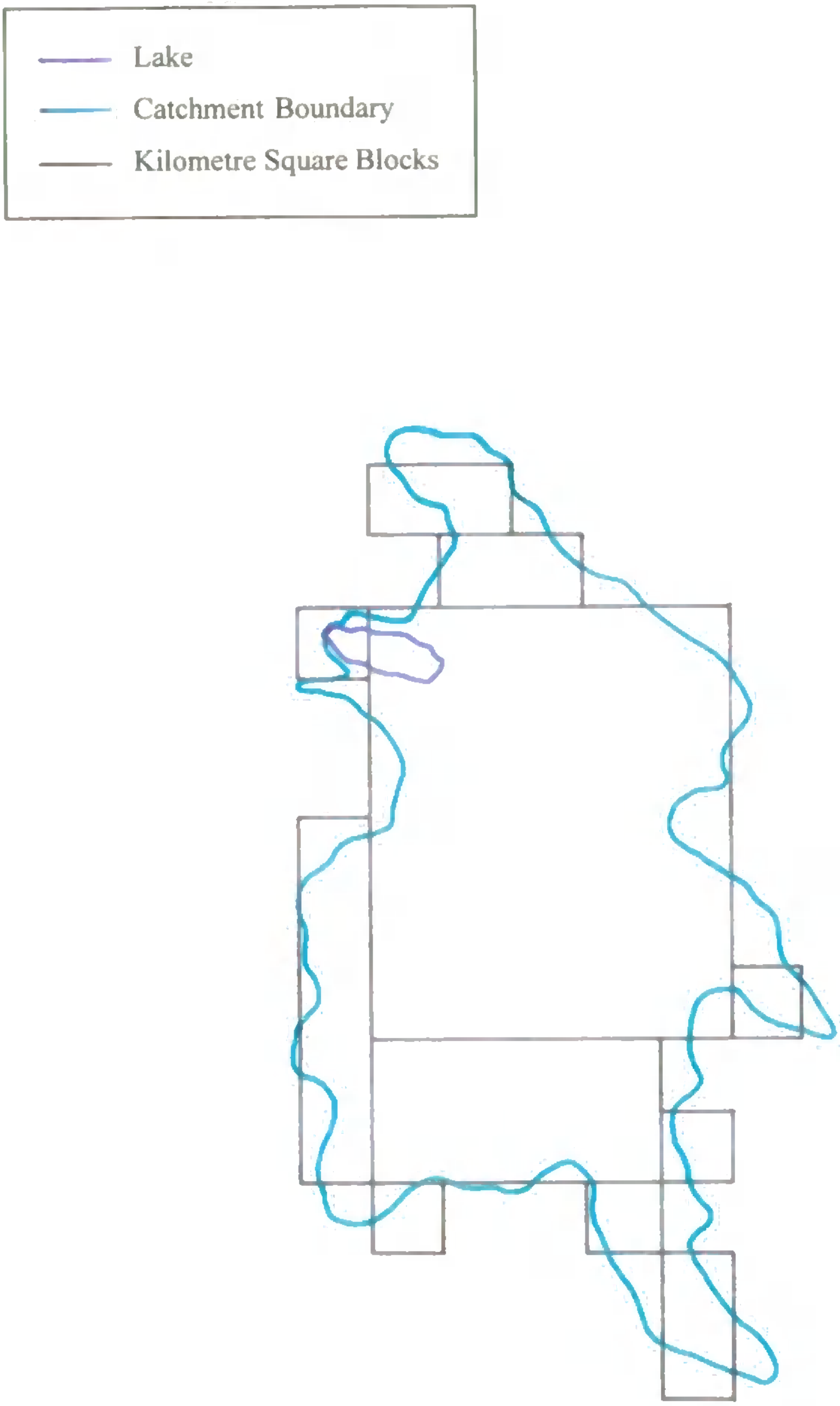

Fig.5.1: Example of the Use of Kilometre Squares for Obtaining Catchment Land Use Data

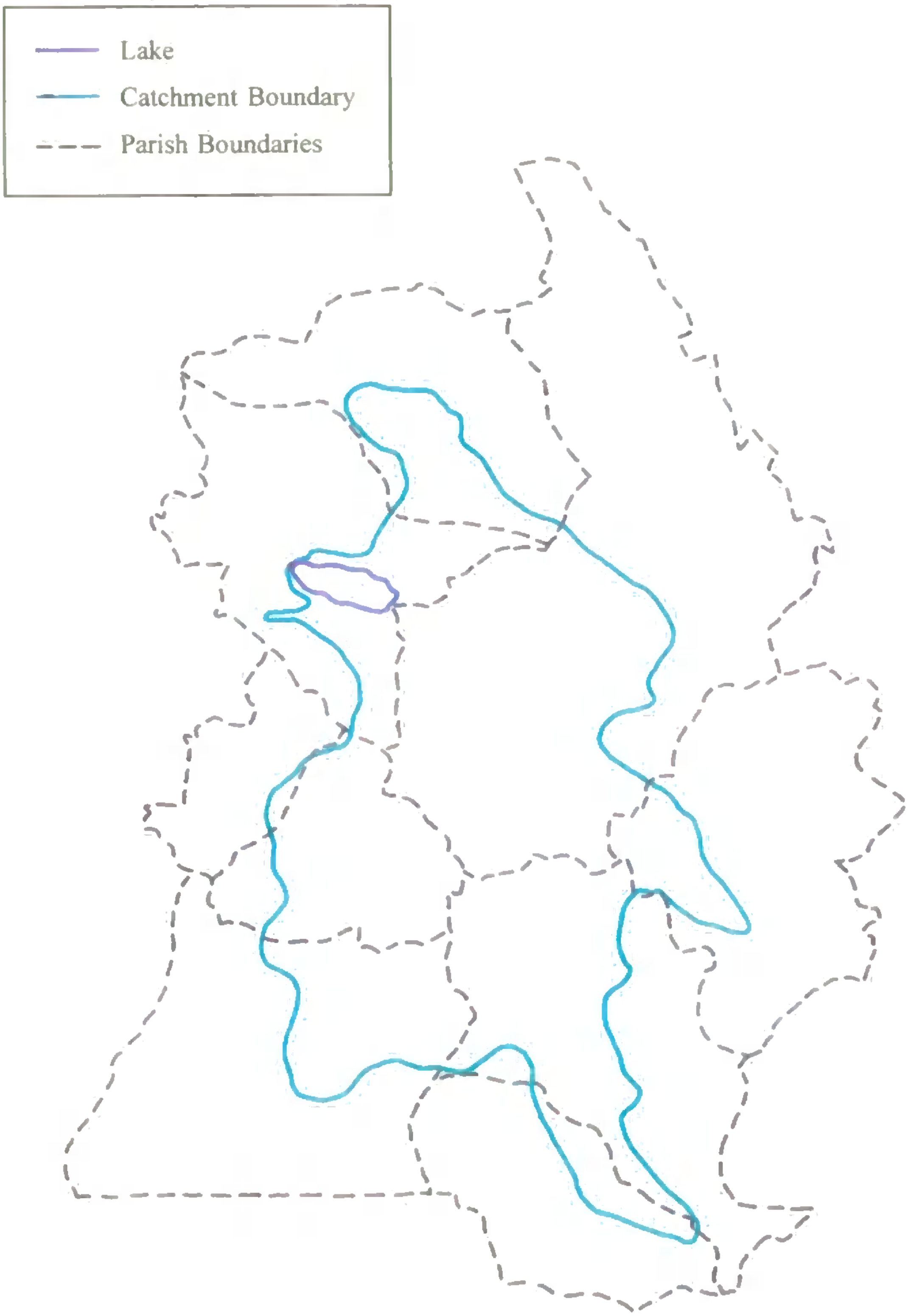

Fig.5.2: Example of the Use of Parishes for Obtaining Catchment Livestock Data

5.2.4 Consented Discharges

In order to obtain details on consented discharges, letters were sent to the relevant National Rivers Authority (NRA) office for each catchment, requesting data on the nature and location of such discharges. In the case of sewage treatment works, details of the population equivalent served were also requested.

5.2.5 Morphometric Data

Data for surface area, volume, mean depth and water residence time were required for each lake. For many sites, values were available from published literature and from EN files. For those lakes where the area was unknown, the cut and weigh method [Lind, 1979] was utilised. Since volume equals mean depth multiplied by surface area, where two of these values are known, the third could be calculated. However, there still remained several lakes for which values were unknown. For Bay Pond and Hedgecourt Lake in South East England, together with Semerwater in North Yorkshire, the data were obtained through studies conducted by undergraduate students [Gilbert, 1994; Denley, 1994; Barlow, 1994]. Their work was well supervised and the results thoroughly examined. The mean depths of four other sites, namely, Comber, Tabley and Tatton Meres, and Petty Pool were also unknown. Advice was sought from limnologists familiar with the Shropshire-Cheshire Meres, who advised assuming mean depth to be half maximum depth [letters from B. Moss, University of Liverpool, 22nd August 1994, and C. Reynolds, Freshwater Biological Association, 19th September 1994]. However, some data, namely, water residence times for the Norbury Meres and Sutton Park Pools, remain as unknowns.

5.2.6 Export Coefficients

After an extensive survey of the literature, a number of problems were encountered in selecting export coefficients suitable for use in this study:

1. A wide range of export coefficients are available for similar types of land use and other sources, as illustrated in Fig.5.3.

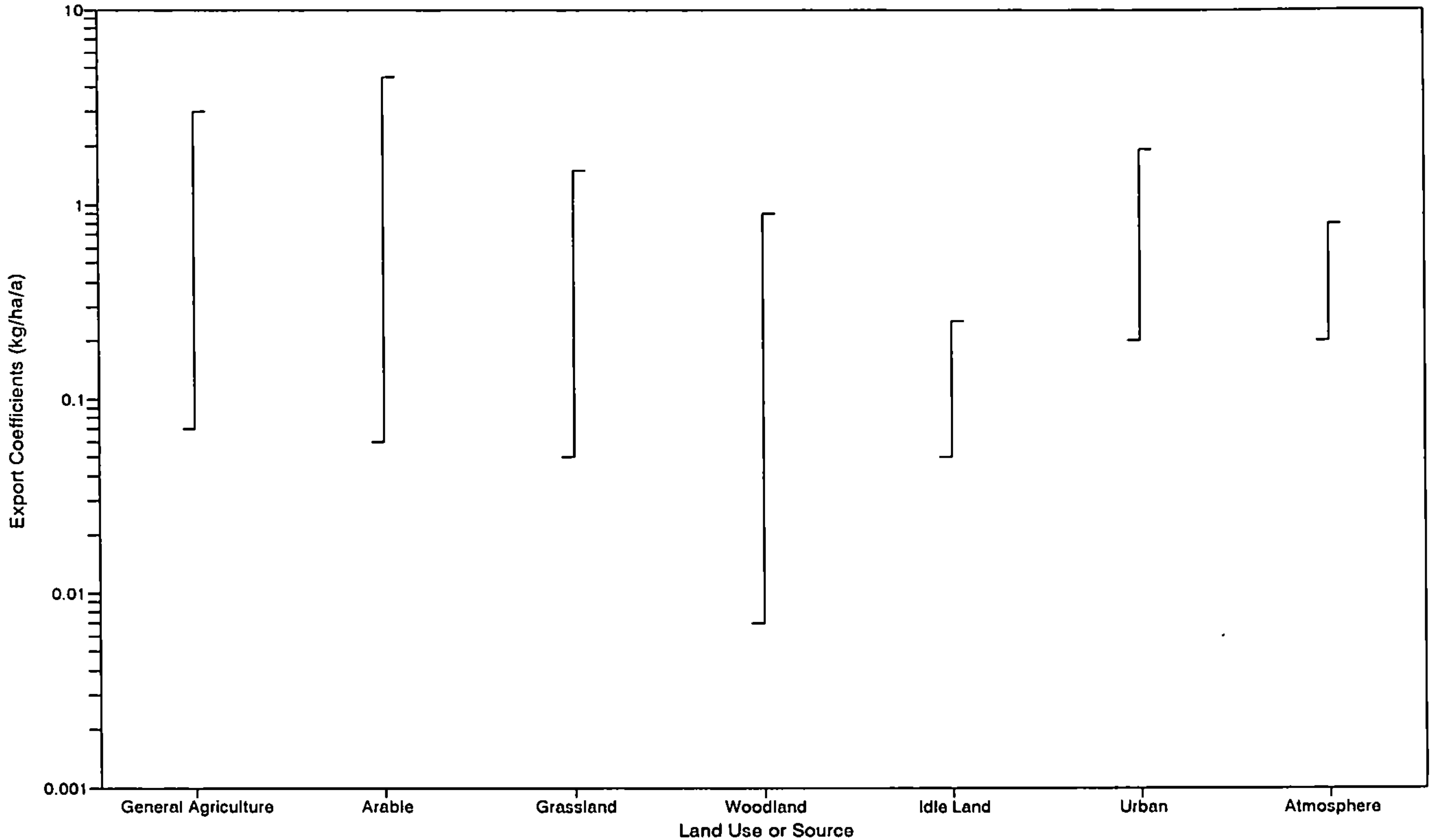

Fig.5.3: The Range of Export Coefficients Available for Phosphorus

2. The types of land uses to which coefficients apply are not necessarily those required for application in this study. For instance, coefficients for grassland are not related specifically to either temporary or permanent pasture.

3. Although several authors state that phosphorus export is of organic, i.e. animal, origin (Table 3.3), no papers refer specifically to inorganic sources.

As stated by Reckhow, Beaulac and Simpson [1980], the selection of export coefficients is necessarily a subjective exercise, aided by detailed knowledge of the catchment in question. Their advice applies to studies in which specific coefficients are to be selected for a particular catchment. In order to produce generalised coefficients for a number of differing catchments, as is necessary here, it seemed more realistic to select a range of values for each type of export, rather than a single figure.

For organic sources this process proved relatively straightforward, since values were available which referred specifically to losses from animal waste. Those shown in Table 3.3 were used to produce a loss rate range of 0.5% to 1.5%. Some variation was encountered in data on the amount of TP produced by different types of animals, the mean of which was calculated to produce the figures in Table 5.2. No allowances were made for losses of phosphorus during storage of animal waste.

Table 5.2: Phosphorus Produced in Animal Waste

Type of Animal	**Phosphorus Produced** kg/ca/a
Cattle[1]	11.40
Pigs[2]	3.78
Sheep[3]	1.59
Poultry[4]	0.18
Other Fowl[5]	0.32

Based on values from [1] (a) (b) (c) (d) (e) (f) (g) [2] (a) (c) (e) (f) (g) [3] (a) (b) (c) (f) [4] (a) (b) (c) (e) (f) (g) [5] (a) (b) (c)

(a) Vollenweider [1968]
(b) Uttormark, Chapin and Green [1974]
(c) Cooke [1976]
(d) Owens [1976]
(e) Richardson [1976]
(f) Gillespie [1989]
(g) Nix [1992]

For inorganic sources a more elaborate procedure was necessary, in which the data for TP losses from unspecified sources were exposed to subjective evaluation. Fig.5.3 includes all values found in the literature, but the application of such a wide range would produce rather meaningless results. Outliers therefore needed to be identified. The number of values available for each land use type were not large enough to carry out a statistical analysis in order to exclude outliers. An alternative process was therefore employed in which the circumstances under which particularly low or high values were generated were examined, and, if felt to be exceptional, those values were excluded. A bias towards the lower end of the range was introduced, in order to allow for the possibility that organic sources of phosphorus may be included in higher figures. However, it was still necessary to devise a separate range of values for land uses not specified in the literature, which included temporary and permanent pasture, and rough grazing. For temporary pasture those values at the higher end of the range for grassland were selected, whereas for permanent pasture lower values were deemed appropriate. In the case of rough grazing, no export coefficient was applied, since this type of land normally receives no *inorganic* fertiliser. This process produced the ranges of values shown in Table 5.3.

Table 5.3: Export Coefficient Ranges Selected for Inorganic Sources

Land Use or Source	**Export Coefficient Range** kg/ha/a
Temporary Grassland[1]	0.15 - 0.45
Permanent Grassland[2]	0.05 - 0.15
Arable[3]	0.40 - 0.80
Woodland[4]	0.01 - 0.05
Settlements and Roads[5]	0.50 - 0.85
Precipitation[6]	0.20 - 0.50

Based on values from [1] (c) (e) (l) [2] (c) (e) (l) [3] (e) (g) (i) [4] (b) (c) (e) (f) (i) (k) [5] (d) (f) (h) [6] (a) (d) (f) (j) (l)

(a) Allen *et al.* [1968]
(b) Uttormark, Chapin and Green [1974]
(c) Cooke [1976]
(d) Owens [1976]
(e) Jorgensen [1980]
(f) Reckhow and Simpson [1980]
(g) Reckhow, Beaulac and Simpson [1980]
(h) Sheaffer and Wright [1982]
(i) Delwiche and Haith [1983]
(j) Rast and Lee [1983]
(k) Clesceri, Curran and Sedlak [1986]
(l) Loehr, Ryding and Sonzogni [1989]

Values were also required for the amount of TP present in domestic wastewater, and again a range of values were available in the literature. Several factors were taken into consideration in deciding upon a value, especially in relation to the contribution of detergents to the TP load. As outlined in Section 2.1.3, the UK has adopted the use of non-phosphate detergents at a different rate than the USA and continental Europe, so that data from abroad on TP levels in wastewater is of limited relevance. Surveys conducted in order to establish per capita TP loadings to STWs in the UK date from the mid 1970s, but two significant changes have occurred since then. First, the market share of phosphate detergents in the UK at that time was about 85% [Devey and Harkness, 1973], but it now represents about 54% [Wilson and Jones, 1994]. Second, in the mid 1970s the proportion of phosphate in detergents was about 37% [Devey and Harkness, 1973], compared to the present level of 15% [pers. obs.]. Separate data on the contribution of phosphorus from non-detergent sources date again from the mid 1970s, but for this component it may be assumed that there is much less variation. Finally, increases in per capita consumption of synthetic detergent during the late 1970s and early 1980s, have been compensated for by subsequent decreases in dosage per wash, as a result of increased efficiency [Smulders and Krings, 1990].

Taking all these factors into consideration, the following result was achieved:

Table 5.4: Total Phosphorus Loading from Domestic Sewage

(kg/ca/a)

Adjustment Factors	Phosphorus from Detergent Sources	Phosphorus from Non-Detergent Sources	Total
Mid 1970s Level[1]	0.70	0.48	1.18
Reduction in Market Share of Phosphate Detergents (31%)[2]	0.48	0.48	0.96
Reduction in Amount of Phosphate in Detergents (22%)[3]	0.38	0.48	**0.86**

[1] Devey and Harkness [1973]; Alexander and Stevens [1976]
[2] Devey and Harkness [1973]; Wilson and Jones [1994]
[3] Devey and Harkness [1973]; pers. obs.

It was assumed that all STWs operating within the catchments of sites under assessment carry out secondary treatment, for which a phosphorus removal rate of 30% was applied [Wilson and Jones, 1994]. An exception was made for STWs where it was known that effluent was subjected to tertiary treatment, that is, those discharging into the River Ant above Barton Broad. Data on phosphorus removal rates for these particular STWs were unavailable [letter from A.A. Chilvers, NRA Anglian Region, Haddiscoe, 26th April 1994], but information on percentage reduction for similar STWs discharging into the River Bure were provided, and a mean of 82% calculated. For catchments where the number of persons connected to septic tank facilities was known, a phosphorus removal rate of 70% was assumed [Loehr, Ryding and Sonzogni, 1989].

5.3 Prediction and Evaluation

In order to predict the present trophic status of the sites under assessment, the export coefficients selected were applied to the data on land use and livestock levels to produce TP loadings. The results were plotted on a graph which synthesises the standard OECD equations to provide approximative indications of the expected trophic category, given the mean inflow TP concentration and mean water residence time. A range of phosphorus reduction strategies were applied on a site specific basis, and the effects of these on trophic status evaluated using the OECD graph.

5.3.1 Computing the Data

The export coefficients selected, together with the data on land use, livestock levels and lake morphology, were entered into a Quattro Pro 4.0 programme in order to calculate predicted phosphorus loadings. Separate spreadsheets were used for each site, but links were established between them so that common values, such as export coefficients, could be altered simultaneously. Results from this part of the exercise were produced in terms of total load (t/a) and areal load ($g/m^2/a$).

Predicted in-lake TP concentrations ($g/m^3/a$) were calculated from the areal loading using the following equation from Vollenweider [1976]:

$$[\bar{P}]_\lambda = \left(\frac{L_p}{q_s \, (1 + \sqrt{\tau_w}\,)} \right)$$

where L_p is areal loading (g/m^2/a), q_s is hydraulic load (mean depth divided by water residence time; m/a), and τ_w is water residence time (years). For consistency of units, the results of this calculation were converted to μg/l/a by multiplying by 1000. In order to evaluate the effects of loading reduction strategies on lake trophic status, using the graph provided for this purpose in OECD [1982 p.95], the results required conversion to mean inflow TP concentration. This was achieved by applying the OECD equation which represents the generalised relationship between in-lake TP concentration and inflow TP concentration [*ibid.*]:

$$[\bar{P}]_\lambda = 1.55 \left(\frac{[\bar{P}]_i}{1 + \sqrt{\tau_w}} \right)^{0.82}$$

which may also be expressed as:

$$[\bar{P}]_i = \sqrt[0.82]{\frac{[\bar{P}]_\lambda}{1.55}} \; (1 + \sqrt{\tau_w})$$

In order to calibrate the model, data on mean inflow TP (as opposed to orthophosphate alone) concentrations were required for as many sites as possible. Despite exhaustive searches insufficient information of this nature was obtainable. Where inflow data were available, they did not cover a sufficient number of years for the figures to be representative. The exception was Barton Broad, but special circumstances are thought to govern phosphorus transport in the region [Johnes, Moss and Phillips, 1994], rendering the site unsuitable for calibration purposes. However, for those sites for which data were available, comparisons were made between predicted and measured values, in order to achieve a qualitative assessment of the validity of the results.

5.3.2 Phosphorus Loading Reduction Strategies

Reduction strategies were selected on a site-specific basis, according to the relative contribution of sources of phosphorus, and in the light of information gained from site visits, EN files and NRA data. However, a degree of standardisation was introduced, according to criteria derived from information included in Chapters 2 and 4.

Reductions in inputs from sewage sources by tertiary treatment were calculated at a standard rate of 80%, since this figure is the minimum stipulated by the Urban Waste Water Treatment Directive for discharges into 'sensitive areas' (Section 4.4.3.2), and is achievable with current technology (Section 2.1.1). Diversion of sewage effluent to sea (Section 2.1.4) necessarily involved reduction by 100%.

For diffuse agricultural sources, two approaches were adopted. First, a general lessening in the intensity of agriculture in a catchment may be achieved through a reduction both in stocking levels and in inorganic fertiliser application rates (Section 2.2.3). A figure of 20% was selected as a reasonable rate of reduction, since a lower figure may not be sufficient to elicit a change in trophic status, yet a higher level may not be acceptable in economic terms. Second, the use of riparian buffer zones appears to be gaining acceptance in agricultural quarters as a nutrient reduction strategy. A buffer zone of 20m was applied in most cases, since this is the width used by MAFF in the Habitat Scheme (Section 4.3.3.2). A reduction rate of 53% was used, based on the work of Doyle, Stanton and Wolf [1977], Dillaha *et al.* [1989], Edwards, Owens and White [1983], Mander [1985] and Thompson, Loudon and Gerrish [1978]. For a few sites, buffer zones of 30m were necessary, for which a reduction rate of 63% was applied.

A reduction strategy was also required for phosphorus in urban runoff. From the discussion of options available for stormwater management in Section 2.1.5, the use of detention ponds appeared to be the most widely applicable strategy. A TP reduction rate of about 50% is reported for a pond volume of 200 to 250m^3 per hectare of urban land, although higher removal efficiencies are possible using larger facilities [Toet, Hvitved-Jacobsen and Yousef, 1990]. For most sites where reduction strategies were required for phosphorus in urban runoff, a rate of 50% was applied, but for some a 70% level of removal was necessary.

For a number of sites, known sources of phosphorus could not be quantified. Such sources included effluent from septic tanks serving an unspecified number of people, unconsented, and therefore illegal, discharges, such as those from storage of farm waste, and intermittent consented discharges, normally from combined sewer overflows. The effect of these sources on loadings were not included, although recommendations for reduction strategies were given where applicable.

5.3.3 Evaluation of Results

For each site, predicted present mean inflow TP concentrations, and those produced by the application of reduction strategies, under both low and high export coefficients, were plotted against water residence time. The OECD graph shown in Fig.3.2 was adapted for use with lakes possessing short water residence times by adjusting the range of the x axis. For most sites the graph illustrated in Fig.5.4 was used, but for those with inflow concentrations in excess of 1000μg/l, the y axis was extended to 10000μg/l.

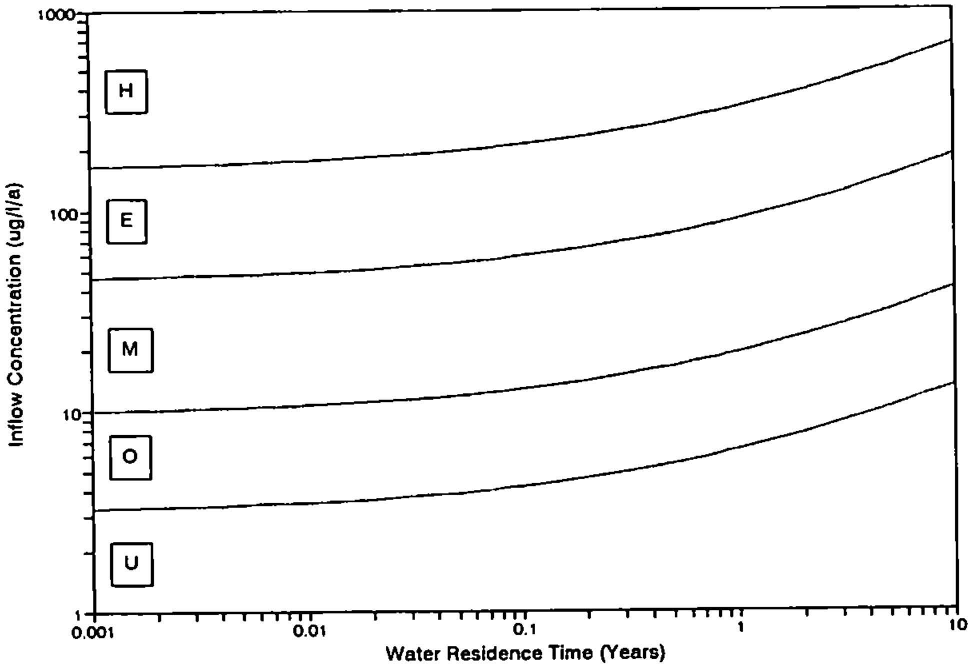

Fig.5.4: OECD Loading Objectives Graph Adapted for Use in this Study

Chapter 6

Site Descriptions

The Sites of Special Scientific Interest (SSSIs) selected for assessment in this investigation may be categorised according both to their geographical location (Fig.6.1) and, to a large extent, their morphological similarity. In each case, a general description of these categories will be followed by more specific information, including the reasons for which the sites were notified as SSSIs by the Nature Conservancy Council (NCC). A brief summary of lake and catchment characteristics will also be given, together with information relating to the nutrient status of the sites, and data supplied by the National Rivers Authority (NRA) on consented discharges.

6.1 Coastal Lakes of South West England

The coastal lakes of South West England (Fig.6.1), which include Loe Pool at Helston, Swanpool near Falmouth, Slapton Ley near Kingsbridge, and Chesil Fleet near Portland, are shallow water bodies, often former estuaries, impounded by sand or shingle bars. While some are tidal or brackish, Slapton Ley and Loe Pool are very rarely affected by marine incursions and may be regarded as freshwater lakes.

6.1.1 Loe Pool

6.1.1.1 Lake and Catchment Characteristics

Loe Pool SSSI (Grid Reference SW 647250), located south of Helston, is the largest natural freshwater lake in Cornwall, extending over an area of about 55 hectares [NCC, 1986b]. Loe Bar and an area of carr at the head of the lake were designated as an SSSI in 1951, but the boundary was not extended to include Loe Pool itself until 1986. The SSSI lies within an Area of Outstanding Natural Beauty, and forms part of the Cornwall Heritage Coast. It is owned mainly by the National Trust (NT).

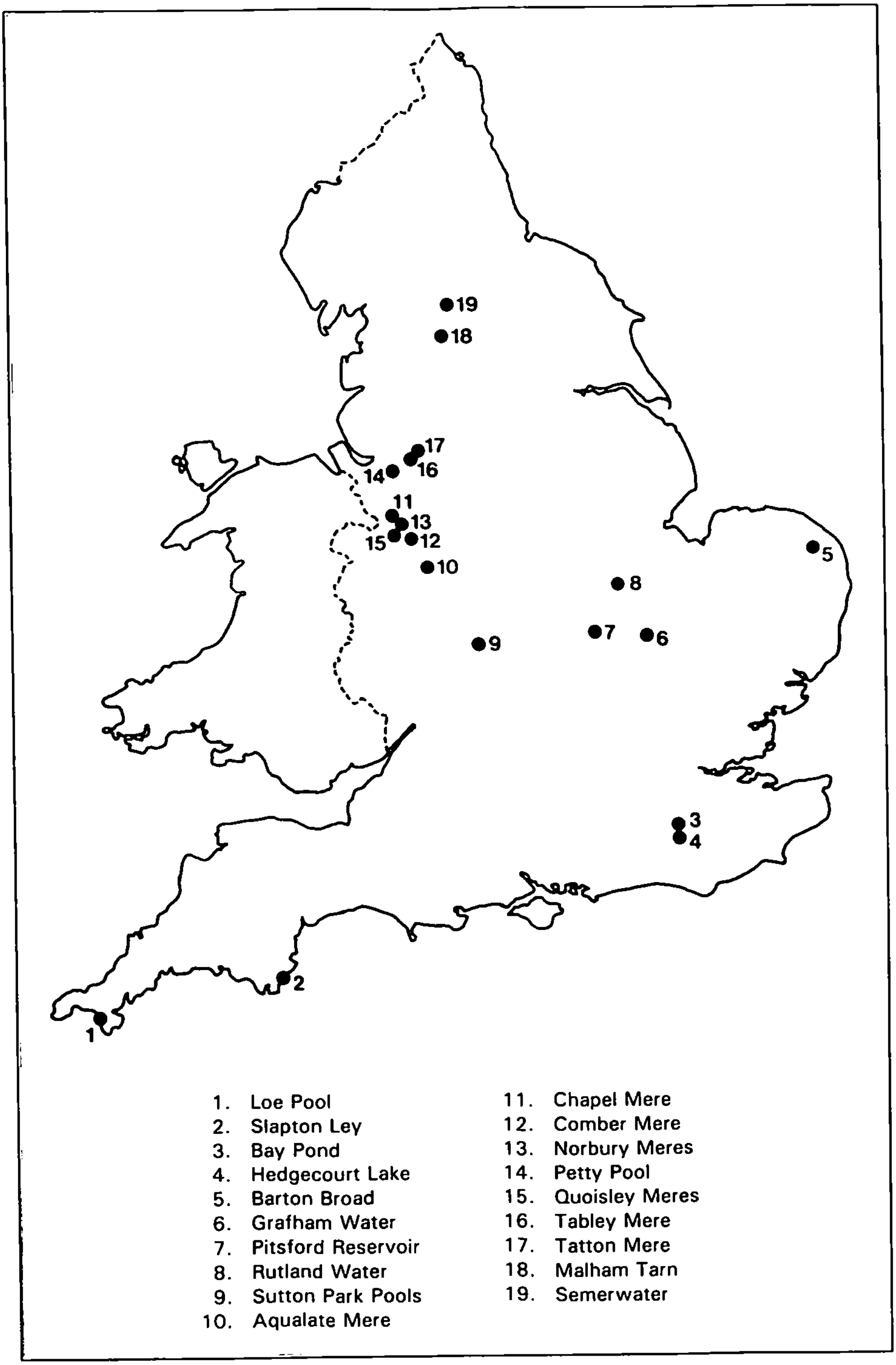

Fig.6.1: Location of Sites Selected for Assessment

Loe Bar, composed mostly of flint shingle and coarse sand, is a classic coastal landform, formed and maintained by predominantly south-west wave regimes [NCC, 1986b]. The annually laminated sediments of the Pool, composed of clastic material, are unique in Great Britain [Coard, 1987; O'Sullivan, 1992]. Both the Pool and the Bar provide scarce habitat not found elsewhere in Cornwall, with rare species of higher plants, bryophytes, and algae, together with many rare and local insect species [NCC, 1986b]. Loe Pool supports nearly 80 species of wintering birds, the numbers of some reaching nationally and regionally important levels. Other habitat types found around the Pool include an extensive area of willow carr, reed beds, ancient woodland and maritime grassland.

Table 6.1: Lake and Catchment Data for Loe Pool

[a] Coard [1987] [b] Johnes, Moss and Phillips [1994]

Catchment Area[a]	54.55km^2
Lake Surface Area[a]	55.6ha
Maximum Depth[a]	10m
Mean Depth[a]	4m
Volume[a]	2.22 x 10^{6}m^3
Mean Hydraulic Residence Time[a]	57 days
Mean pH[a]	7.1
Mean Annual Nitrogen Concentration[b]	8.25mg/l (TN)
Mean Annual Phosphorus Concentration[b]	270μg/l (TP)

Lake and catchment data for Loe Pool are shown in Table 6.1. Metalliferous deposits in the catchment have been extensively exploited over many centuries, significantly affecting sediment yield [Coard, 1987]. The lake is fed by one main river, the Cober, and two large streams. Outflow is *via* a sluice constructed under the cliff at the western end of the shingle bar, which serves to reduce the incidence of flooding in Helston. Prior to 1946 the River Cober between Helston and Loe Pool followed a natural and sinuous course, which was subsequently straightened and widened, again to prevent flooding.

Land use in the catchment is predominantly agricultural, the two largest settlements being the town of Helston and the Royal Naval Air Station (RNAS) at Culdrose. Helston sewage treatment works (STW) was commissioned in 1930, and discharges into the River Cober less than 1 km north of Loe Pool [Coard, 1987]. It serves a population of 8,214 in winter, rising to 9,549 in summer [letter from H. Richards, South West Water Services Ltd., St Austell]. The RNAS has its own STW, with an outfall into Carminowe Stream, which caters for 900 resident and 2,350 daily personnel [data supplied by Captain C.L.L. Quarrie, RNAS, Culdrose]. The many small and scattered settlements in the catchment utilise septic tanks, but the number of people involved is unknown.

6.1.1.2 Current Status

Analysis of sediment chemistry demonstrates a change to a more productive state during the 1930s, which coincides with the opening of Helston STW [O'Sullivan, 1992]. Diatom analysis of sediment cores reveals that, from 1945 onwards, increases occurred in the relative abundance of the planktonic species *Asterionella formosa* Hassall, and in 1973 of the centric *Melosira granulata* Ehrenberg. By 1986, *Stephanodiscus hantzschii* Grunow, which is associated with nutrient enrichment by sewage, became more abundant. Since 1968 blooms of *Microcystis aeruginosa* Kützing have been produced in most years, and every year since 1989 [Flory and Hawley, 1994]. In 1993, however, an extensive bloom of the green alga *Hydrodictyon reticulatum* (L.) Lagerheim was observed, and appeared to replace *Microcystis*. This change was accompanied by clearer water, and lower levels of chlorophyll *a*.

More recently, the NT purchased over forty acres of farmland bordering Loe Pool [NT, 1995]. The acquisition will not only enable the Trust to secure the quality of the overall landscape within an AONB, but will also ensure that appropriate farming practices are carried out on fields adjacent to the lake, in order to protect its conservation interest.

6.1.2 Slapton Ley

6.1.2.1 Lake and Catchment Characteristics

Slapton Ley SSSI (Grid Reference SX 826441), located south of Dartmouth, is the largest natural freshwater lake in South West England [NCC, 1984c]. It was designated an SSSI in 1952, and renotified in 1984. Slapton Ley lies within an AONB and became a National Nature Reserve (NNR) in 1993. The SSSI meets the criteria for designation as a Special Protection Area under the terms of the European Community Directive 79/409/EEC on the Conservation of Wild Birds. The site is owned by the Herbert Whitley Trust and managed by the Field Studies Council.

Slapton Ley is divided by a causeway into two basins, the Lower and the Higher Leys. The Lower Ley is a large (77 ha) shallow lake, while the Higher Ley consists of rich carr and fen vegetation [NCC, 1984c]. The site supports a wide variety of habitats, and a rich and diverse flora and fauna. It is of particular importance for lichens, fungi and invertebrates, and for passage and wintering birds. Slapton Ley is the only native locality in Britain for Strapwort (*Corrigiola litoralis* L.), and the main breeding stronghold of Cetti's Warbler (*Cettia cetti* Temm.). The Higher Ley is notable for its population of European Otter (*Lutra lutra* L.).

Table 6.2: Lake and Catchment Data of Slapton Ley

[a] Johnes and O'Sullivan [1989] [b] Van Vlymen [1980]
[c] Johnes, Moss and Phillips [1994]

Catchment Area[a]	$46km^2$
Lake Surface Area[b]	77ha
Maximum Depth[b]	2.8m
Mean Depth[b]	1.55m
Volume[b]	$1.19 \times 10^6 m^3$
Mean Hydraulic Residence Time[b]	17.5 days
Mean pH[b]	8.7
Mean Annual Nitrogen Concentration[c]	9.21mg/l (TN)
Mean Annual Phosphorus Concentration[c]	$150\mu g/l$ (TP)

Lake and catchment data for Slapton Ley are shown in Table 6.2. The catchment extends over $46km^2$, and consists of a deeply dissected plateau, developed over Lower Devonian slates which weather to shallow silty loams [Trudgill, 1983]. The main tributaries are the River Gara which enters *via* the Higher Ley, and the Start Stream which flows into the north west bay of the Lower Ley. Outflow is through a horizontal weir, culvert and sluice gate system at the southern end.

Land use in the catchment is predominantly agricultural [Johnes and O'Sullivan, 1989]; the majority consists of grassland, while the remainder is used mostly for cereals and field vegetables, with some market gardening. There are two STWs serving three of the four main villages. The larger of these is situated on the edge of the SSSI, into which the effluent is discharged, through an area of marshland. At settlements which are not connected to the STWs, septic tanks are employed, but the number of people involved is unknown. In the case of the village of Stokenham, sewage is piped to Kingsbridge.

6.1.2.2 Current Status

Sequential inorganic chemical analysis of sediments from Slapton Ley records increases in mineral phosphorus inputs to the lake from the 1940s onwards [Heathwaite and O'Sullivan, 1991], which coincides with the intensification of agriculture after the second world war [Acott, 1989]. Higher inputs of mobile phosphorus, dating from *ca* 1953 to 1960, may be attributable to increased sewage outputs [O'Sullivan, 1992].

Diatom analyses of sediment cores show that, prior to *ca* 1960, the most common taxa present were epiphytic or benthic forms, indicating clear water conditions and abundant submerged macrophytes [O'Sullivan, 1992]. Planktonic diatoms then began to proliferate, effecting a decrease in the depth of the euphotic zone. In *ca* 1976, a large increase occurred in the relative abundance of the centric diatom *Cyclostephanos dubius* (Fricke in A. Schmidt), which is associated with advanced eutrophication by inputs of phosphorus derived from sewage [Battarbee, 1986].

Three surveys have been undertaken of the submerged macrophyte community of the Lower Ley: Brookes and Burns [1969], Cole [1984] and Wilson [1991]. A comparison of the results of these surveys appears to concur with conclusions drawn from the paleolimnological analyses described above, in that, prior to the mid 1970s, a reasonably diverse plant community existed, in clear water conditions. Since then there appears to have been a decline in the abundance of submerged macrophytes over extensive areas of the Lower Ley, together with a change in species composition towards those more strongly associated with eutrophic waters [Wilson, 1991]. For instance, the non-rooted, nitrophilous plant, *Ceratophyllum demersum*, which is capable of forming large mats near the water surface [Hutchinson, 1975], now appears to be the most prolific species.

Slapton Ley has been regarded as an important recreational coarse fishery since the late eighteenth century [Bregazzi, Burrough and Kennedy, 1982]. During the early 1970s, perch (*Perca fluviatilis* L.) were abundant in the Lower Ley, and the most common cyprinid was the rudd (*Scardinius erythrophthalmus* (L.)). Roach (*Rutilis rutilis* (L.)) were present, but rare, and the only other species recorded in significant numbers were the pike (*Esox lucius* L.) and the eel (*Anguilla anguilla* (L.)) [Bregazzi and Kennedy, 1982].

Since that time, considerable changes have occurred: roach replaced rudd as the most common cyprinid, with a concurrent decline in the numbers of perch. Pike became smaller and with shorter life-spans [Bregazzi and Kennedy, 1980]. Despite periodic reductions in the roach population, through the parasite *Ligula intestinalis* (L.), this species continues to be the most abundant. Such changes in the fish fauna may be related to eutrophication, not least as a result of the dependence of rudd on submerged macrophytes for food and shelter [Carpenter and Lodge, 1986].

As a result of concern over the possible effects of discharges from Slapton STW on the water quality of the Ley, there were plans to divert the effluent to sea [letter from J. Murray-Bligh, NRA, Exeter, October 1992]. This work has not yet occurred, but secondary treatment at the works has meanwhile been upgraded.

6.2 Surrey Ponds

The sites in the Surrey area selected for assessment (Fig.6.1) are two of several ancient ponds created for various purposes by the damming of rivers. Well-established open water communities such as these are rare in Surrey. They form an integral component of SSSIs noted for their diversity of habitat types.

6.2.1 Bay Pond

6.2.1.1 Lake and Catchment Characteristics

Bay Pond (Grid Reference TQ 353516) is one three lakes known as Godstone Ponds SSSI, designated as such in 1968, and renotified in 1985 [NCC, 1985a]. Bay Pond is managed as a nature reserve by the Surrey Wildlife Trust (SWT). The Pond lies on quartzose sandstones of the Folkstone beds, but is fed by a stream whose source lies in the chalk of the North Downs. The significant calcareous influence of this stream led to considerable variation in the flora of the pond and the alder carr and fen around it. The open water, fen and woodland habitats support a large number of species of breeding birds.

Table 6.3: Lake and Catchment Data for Bay Pond

[a] This study [b] Gilbert [1994]
[c] Supplied by Surrey Wildlife Trust

Catchment Area[a]	$0.031km^2$
Lake Surface Area[b]	2.6ha
Maximum Depth[b]	1.45m
Mean Depth[b]	1.08m
Volume[b]	$0.0281 \times 10^6 m^3$
Mean Hydraulic Residence Time[b]	1.8 days
Mean pH[c]	8.4
Nitrogen Concentration Range[c]	3.7 to 16.0mg/l (TON)
Phosphorus Concentration Range[c]	25 to 4520μg/l (TP)

Lake and catchment data for Bay Pond are presented in Table 6.3. It is a small, shallow waterbody, with a very short mean water residence time. Sluices on the three outflows maintain the water at a constant level. The outflows converge and flow down into Leigh Place Pond, and eventually into the River Eden. The catchment is largely rural, although there are several areas of urban land [Ordnance Survey (OS), 1991a]. The M25, A22 and

A25 roads are close by, and drainage from all three enter the inflow stream [Gilbert, 1994]. Fish were introduced into Bay Pond in 1965, and it has since been used by Godstone Angling Club. There are no STWs within the catchment.

6.2.1.2 Current Status

Prior to 1965, Canadian Pondweed (*Elodea canadensis* Michx.) was extremely abundant, but it has now disappeared [Gilbert,1994]. A decline in the condition of Bay Pond, largely as a result of silting, prompted SWT to dredge the lake in 1985. Using a suction pump, silt was removed to 1.5m. The turbid water was pumped out, and allowed to settle in lagoons prior to flowing back into the lake. The following year a silt trap was constructed, but there remains a potential risk of silt entering the lake during heavy rainfall.

Since 1985 SWT has undertaken monitoring work, including yearly depth surveys and half-yearly water analysis of the inlet and outlet [Gilbert, 1994]. Macrophyte surveys have revealed a complete absence of species, which is attributed to the turbidity of the water. English Nature (EN) considers the water quality to be poor, and the absence of a diverse aquatic flora affects the special interest of the site [EN, 1991b]. However, there are no records of algal blooms occurring at Bay Pond [Gilbert, 1994].

6.2.2 Hedgecourt Lake

6.2.2.1 Lake and Catchment Characteristics

Hedgecourt Lake (Grid Reference TQ 355403), a former mill pond, represents the major part of Hedgecourt SSSI, the most important wetland site remaining in south-east Surrey [NCC, 1986c]. It was designated an SSSI in 1976, and renotified in 1986. The site incorporates a range of habitats, including wet and dry woodland, open heath, neutral grassland and fen, as well as open water. When renotified, these habitats supported a wide variety of fauna, notably a number of locally-distributed beetles, and breeding populations of open water and woodland birds.

Table 6.4: Lake and Catchment Data for Hedgecourt Lake

[a] This study [b] Denley [1994]

Catchment Area[a]	9.73 km^2
Lake Surface Area[b]	17.2 ha
Maximum Depth[b]	1.25 m
Mean Depth[b]	0.84 m
Volume[b]	0.145×10^6 m^3
Mean Hydraulic Residence Time[b]	83.01 days
Mean pH	No data
Mean Annual Nitrogen Concentration	No data
Mean Annual Phosphorus Concentration	No data

Hedgecourt Lake has received little attention in terms of scientific investigation, and limited data are available (Table 6.4). It is a shallow waterbody, but is larger in area than Bay Pond, and with a considerably longer mean water residence time of almost twelve weeks. The Eden Brook is both the inflow and outflow. The catchment is predominantly urban, and includes part of East Grinstead [OS, 1991a]. It is traversed by two major roads, and plans for a further bypass have been proposed [Denley, 1994]. The Green Belt status of the area means, however, that further development is limited. An area of fen and woodland at the west end of the site is owned and managed by Surrey Wildlife Trust. Other landowners are a yacht club, which uses the lake for sailing, and an angling club, which stocks the water with fish [letter from EN, East Sussex, June 1993]. There are no STWs within the catchment, but a total of 310 people are connected to septic tanks or sealed cesspools [Denley, 1994].

6.2.2.2 Current Status

According to EN [1991b], the waters of Hedgecourt Lake are now considered to be eutrophicated, turbid and of poor water quality in general. There has been a corresponding reduction in the diversity of higher aquatic plants, thereby affecting the special interest of the site. In 1990 the abundance of Canadian pondweed (*Elodea* *canadensis*) increased to nuisance levels, and large amounts were manually removed. The following year herbicide was used in an attempt to alleviate the problem [Denley, 1994]. The reasons for the change in water quality are thought to be runoff water quality, bird droppings, overstocking with fish and, possibly, pollution from toilets associated with the sailing club [EN, 1991b]. Silting is a major problem, having reduced the depth of the water to only 25 cm in places [Denley, 1994].

6.3 Norfolk Broads

The Norfolk Broads (Fig.6.1) consists of fifty or so lakes and their associated rivers and wetlands [Moss, 1983]. The lakes were formed during the 12th century through the flooding of medieval peat workings, as climate change caused a rise in sea level. Numerous channels or dykes were subsequently cut between the Broads and the rivers, creating a network of waterways. Around the beginning of the nineteenth century, drainage of the valley marshland began in earnest, establishing the 20,000 hectares of pasture which now occupy the river valleys.

The present Broadland consists of waterways, undrained wetland or fens, drained grazing marshes, arable upland, and some urban areas [Moss, 1983]. River discharges are generally quite low, but since most of the Broads are relatively small and shallow, they are usually rapidly flushed. The waterways are used extensively for sewage effluent disposal, spray irrigation and domestic consumption. As a result of highly agricultural catchments and discharges of sewage effluent, the majority of Broads possess elevated ($>100\mu g/l$) phosphorus concentrations. On average, sewage effluent contributes more than 75% of the total phosphorus concentrations of most of the waters, but only accounts for between 1% and 5% of the flow.

Eutrophication has resulted in the widespread loss of submerged and floating-leaved macrophytes in much of the Broadland waterway, and consequent changes in ecosystem structure [Moss, 1983]. Increased boating activity has also led to decreased species and habitat diversity through bank erosion and sediment resuspension. The Broads thereby lost much of their conservation value by the early 1960s.

Restoration measures began in the mid to late 1970s with isolation of Broads from nutrient-rich inflows, removal of surface sediment, phosphate stripping at STWs, and diversion of sewage effluent. In most cases, external nutrient reduction has not resulted in full restoration [Phillips *et al.*, 1994]. Resistance to recovery is thought to be the result of mechanisms which maintain the fish-zooplankton relationship in homeostasis, and release of phosphorus from the sediment. Biomanipulation is therefore being used in attempts to restore clear water, and encourage the re-establishment of macrophytes.

6.3.1 Barton Broad

6.3.1.1 Lake and Catchment Characteristics

Barton Broad (Grid Reference TG 362213) forms part of the Ant Broads and Marshes SSSI, which also includes Sutton Broad [NCC, 1989a]. It was designated as an SSSI in its own right in 1954, and amalgamated with Sutton Broad and Ant Marshes SSSIs when renotified in 1989. The floodplain of this portion of the River Ant represents one of the most extensive areas of primary fen remaining in Britain, and is considered the finest example of valley fen in Western Europe. The site is included within the Broads Environmentally Sensitive Area.

Table 6.5: Lake and Catchment Data for Barton Broad

[a] Osborne and Moss [1977] [b] Osborne [1981]
[c] Supplied by A.A. Chilvers, NRA, Haddiscoe

Catchment Area[a]	118.1km^2
Lake Surface Area[b]	70.85ha
Maximum Depth	No data
Mean Depth[b]	1.15m
Volume[b]	0.82 x 10^{6}m^3
Mean Hydraulic Residence Time[a]	9.125 days
Mean pH[c]	8.33
Mean Annual Nitrogen Concentration[c]	1.1mg/l (NO_3)
Mean Annual Phosphorus Concentration[c]	150μg/l (TP)

Morphometric and chemical data for Barton Broad are shown in Table 6.5. It is the second largest broad covering over 70 hectares, but with a mean depth of only about a metre, it possesses a short mean water residence time of less than ten days [Osborne, 1981]. The River Ant, which forms a tributary of the River Bure, flows through the Broad. The river is tidal to a point 8 km above Barton Broad at Honing Lock [Phillips, 1984]. A mean annual total phosphorus concentration of 150μg/l is recorded for Barton Broad [data supplied by A.A. Chilvers, NRA, Haddiscoe].

6.3.1.2 Current Status

When first notified as an SSSI, Barton Broad supported a very rich flora and fauna [NCC, 1989a]. By 1968, however, only floating-leaved macrophytes and some *Ceratophyllum demersum* remained, but these species disappeared by 1972 [Osborne and Moss, 1977]. The demise of the macrophyte community, which was accompanied by increases in turbidity and in rates of sedimentation, is attributed to eutrophication [Moss, 1980].

Studies on the paleolimnology reveal that in 1800, prior to installation of main sewers, total phosphorus concentration was about 13μg/l [Moss, 1980]. By 1900, agricultural disturbance, including land drainage, increased this to 52μg/l, but by 1920, the level was still only about 72μg/l. The first STW at North Walsham was constructed in 1924, and by 1940 total phosphorus levels had risen to 120μg/l. It is during this period that abundant macrophytes were recorded, including the bulky floating *Stratoites aloides* L., which derives its nutrients from the water, rather than the sediments. A further STWs was opened at Stalham in 1950.

A feature of the sediment is a long black tongue of sulphide-rich material which extends into the Broad from the inflow [Moss, 1980]. The base of this layer dates back to 1920 to 1930, and coincides both with the beginning of increases in plankton, and with the opening of North Walsham STW. The tongue represents not only movement of labile organic sediment into the Broad from the river, but also decomposition of algal and higher plant organic material.

The catchment of the River Ant contains sixteen STWs, which, during the late 1970s, were thought to contribute approximately 80% of the phosphorus load to Barton Broad [Phillips, 1984]. In 1977, Anglian Water Authority, as it then was, introduced phosphorus stripping with ferric sulphate at Stalham STW. A monitoring programme was initiated to measure the phosphorus budget for the Broad, and to determine the biological response to the reduced phosphorus load. Few changes occurred in the first two years, and further measures were taken to reduce phosphorus loads from STWs. In 1980 effluent from North Walsham STW was diverted to sea, followed in 1982 and 1983 by the installation of phosphorus stripping facilities at Worstead, Horning and Southrepps STWs.

Twelve years after achieving a 90% reduction in external phosphorus load to Barton Broad, there is little evidence that the lake is returning to its previous clear water state [Phillips *et al.*, 1994]. In 1989, NRA Anglian region considered Barton Broad to be experiencing blue-green algal 'problems' [NRA, 1990]. Sampling showed the dominant genera to be *Oscillatoria*, *Anabaena* and *Aphanizomenon*, and toxicity tests proved positive. Mass balance studies have demonstrated that summer total phosphorus concentration is determined by internal, rather than external load [Phillips *et al.*, 1994]. Experiments in Hoveton Great Broad, involving removal of fish, appear to confirm that biomanipulation is capable of reducing sediment phosphorus loss, through both direct and indirect effects. Other investigations include monitoring the effects of suction dredging of upper sediment layers, and the direct addition of ferric chloride. The results of such experiments are extremely important, and, if favourable, may be applied to Barton Broad in order to prevent further phosphorus release, and encourage a return to clear water and a diverse macrophyte community.

6.4 Reservoirs of Eastern England

The three sites selected for assessment, Grafham Water, Pitsford Reservoir and Rutland Water (Fig.6.1), are part of a network of large surface reservoirs created in an area where the geology is unsuitable for the extraction of groundwater [Anglian Water Services Ltd., undated]. Water is pumped into the reservoirs from nearby rivers during periods of high to medium flow. They therefore act primarily as pumped storage impoundments, rather than retaining water only from their catchments [Hayes and Greene, 1984]. The creation of large areas of water attracted populations of breeding, passage migrant and wintering birds. The sites became nationally important for several species, and, consequently, were designated SSSIs. The particular causes, and consequent problems, of eutrophication in these reservoirs are very similar. These aspects of the three sites will therefore be discussed together in Section 6.4.4.

6.4.1 Grafham Water

6.4.1.1 Lake and Catchment Characteristics

The construction of Grafham Water (Grid Reference TL 150680) was completed in 1966 [Anglian Water Services Ltd., undated]. It was designated an SSSI in 1986, on account of its regionally and nationally important numbers of passage migrant and wintering birds, as well as a significant variety of breeding species [NCC, 1986d]. Areas of grassland, scrub, marsh and temporarily inundated shoreline provide additional habitat. The Bedfordshire and Huntingdonshire Wildlife Trust manage part of the site as a Nature Reserve.

Table 6.6: Lake and Catchment Data for Grafham Water

[a] This study [b] Anglian Water Services Ltd [undated]
[c] Supplied by P. Daldorph, Anglian Water Services Ltd., Histon

Catchment Area[a]	8.69km^2
Lake Surface Area[b]	635ha
Maximum Depth	No data
Mean Depth[c]	9.3 m
Volume[c]	59.06 x 10^6m^3
Mean Hydraulic Residence Time[c]	248.2 days
Mean pH	No data
Mean Annual Nitrogen Concentration	No data
Mean Annual Phosphorus Concentration[c]	750μg/l (TP)

Lake and catchment data for Grafham Water are shown in Table 6.6. The lake is fed and drained directly by Diddington Brook, but this flow represents only a minor contribution to the total water budget of the site. As a storage reservoir, the bulk of its water (72800Ml in 1993) is pumped from the River Great Ouse at Offord [data supplied by P. Daldorph, Anglian Water Services Ltd., Histon].The natural catchment of Grafham Water is largely rural, and contains no STWs. In contrast, the basin of the River Great Ouse down to Offord receives the effluent from some 140 STWs, which together treat wastewater from the equivalent of nearly 944,000 people. Additionally, there are a large number of other consented discharges within this catchment area, many of which are from small, privately-owned STWs [data supplied by S. Hopper, NRA, Peterborough]. It has been estimated that, under base flow conditions, about 60% of the water of the River Great Ouse at Offord is derived from sewage effluent [Hayes and Greene, 1984].

6.4.2 Pitsford Reservoir

6.4.2.1 Lake and Catchment Characteristics

Pitsford Reservoir (Grid Reference SP 780708) is the oldest of the three sites, being completed in 1956 [Anglian Water Services Ltd., undated]. The site was designated an SSSI in 1970, and renotified in 1984. It is a major site for passage migrant and wintering waterfowl, and supports a significant number and variety of breeding species [NCC, 1984b]. An area to the north of the reservoir is managed as a nature reserve by the Northamptonshire Trust for Nature Conservation. Additional habitat is provided by areas of grassland, scrub, marsh and temporarily inundated shoreline.

Lake and catchment data for Pitsford Reservoir are shown in Table 6.7. The catchment is drained by several small streams, and the outflow forms a tributary of the River Great Ouse. The main water supply (6032Ml in 1993) is pumped from the River Nene at Duston Mill [data supplied by P. Daldorph, Anglian Water Services Ltd., Histon]. The natural catchment of Pitsford Reservoir contains three STWs, but these together serve the equivalent of fewer than 2,000 people. The River Nene down to Duston Mill receives the effluent from a further 14 STWs, which together treat wastewater from the equivalent of over 41,000 people. There are also a number of other consented discharges within this catchment, many of which are from small, privately-owned STWs [data supplied by S. Hopper, NRA, Peterborough].

Table 6.7: Lake and Catchment Data for Pitsford Reservoir

[a] This study [b] Anglian Water Services Ltd. [undated]
[c] Supplied by P. Daldorph, Anglian Water Services Ltd., Histon

Catchment Area[a]	46.23km^2
Lake Surface Area[b]	300ha
Maximum Depth	No data
Mean Depth[c]	6m
Volume[c]	59 x 10^6m^3
Mean Hydraulic Residence Time[c]	284.7 days
Mean pH	No data
Mean Annual Nitrogen Concentration	No data
Mean Annual Phosphorus Concentration[c]	740μg/l

6.4.3 Rutland Water

6.4.3.1 Lake and Catchment Characteristics

Rutland Water (Grid Reference SK 928070), originally known as Empingham Reservoir, was completed in 1977 [Anglian Water Services Ltd., undated]. It was designated an SSSI in 1981, and renotified in 1984 [NCC, 1987e]. In 1991, Rutland Water was designated as a site of international importance as waterfowl habitat under the Ramsar Convention on Wetlands. In the same year it qualified as a Special Protection Area under the EC Directive 79/409 on the Conservation of Wild Birds. The Leicestershire and Rutland Trust for Nature Conservation manages part of the site as a Nature Reserve, in association with Anglian Water Services Ltd.

The potential of Rutland Water as an important wetland site was recognised at its conception [Appleton, 1982]. In consultation with the Countryside Commission and the

NCC, the conservation requirements for the site were incorporated into the design of the reservoir. Existing habitats, including old meadows, scrub and mature woodland, were preserved, while new ones, such as islands, shelter belts, lagoons and bunds, were provided. In order to ensure that recreational activities do not cause disturbance to wildfowl, zonation of the reservoir was introduced. Within the reserve itself, access is restricted to some areas so that they remain comparatively undisturbed. Educational facilities are provided, as well as a number of birdwatching hides.

The design and management of Rutland Water have led to the creation of a major wetland area, combining large stretches of open water with a mosaic of terrestrial, marsh and aquatic habitats [NCC, 1987e]. The site supports exceptional numbers and diversity of passage migrant and wintering species of birds, while the range of breeding species is of increasing significance.

Morphometric and chemical data for Rutland Water are shown in Table 6.8. The catchment is drained by several small streams, and the outflow forms the River Gwash. The major part of the volume of the reservoir water is pumped from the River Nene at Wansford (42432Ml in 1993), and the River Welland at Tinwell (8216Ml in 1993) [data supplied by P. Daldorph, Anglian Water Services Ltd., Histon].

Table 6.8: Lake and Catchment Data for Rutland Water

[a] This study [b] Anglian Water Services Ltd [undated]
[c] Supplied by P. Daldorph, Anglian Water Services Ltd., Histon

Catchment Area[a]	$60.1km^2$
Lake Surface Area[b]	1255ha
Maximum Depth[b]	34m
Mean Depth[c]	10m
Volume[c]	$124 \times 10^6 m^3$
Mean Hydraulic Residence Time[c]	766.5 days
Mean pH	No data
Mean Annual Nitrogen Concentration	No data
Mean Annual Phosphorus Concentration[c]	$717\mu g/l$

The natural catchment of Rutland Water contains three STWs. One of these, at Oakham, serves the equivalent of over 11,000 people, and discharges into a lagoon in the south 'arm' of the reservoir [Appleton, 1982]. The catchments of the Rivers Nene and Welland, down to the points of abstraction, receive effluent from over 100 STWs, which together treat wastewater from the equivalent of over 773,000 people [data supplied by P. Daldorph, Anglian Water Services Ltd., Histon]. Additionally, there are a number of other consented discharges within these catchments, many of which are from small, privately-owned STWs [data supplied by S. Hopper, NRA, Peterborough].

6.4.4 Current Status

The potential and actual problems caused by eutrophication of public water supply reservoirs in Eastern England have been recognised for many years. Identification of sewage effluent discharges as the major source of phosphorus loadings has long been established by the water industry itself [Hayes and Greene, 1984]. Indeed, even before Rutland Water was constructed, the then Anglian Water Authority anticipated that water quality problems were likely, owing to the high phosphorus concentrations of the input waters, and the propensity of reservoirs of such depth to stratify thermally [Low, 1982]. By 1982, the water quality of all three reservoirs was expected to deteriorate as phosphorus loadings of raw water inputs increased with population expansion [Hayes and Greene, 1984]. Toxicity tests on blue-green algae were carried out on seven waterbodies by Anglian Water between 1983 and 1985. These tests proved positive at Grafham during 1983, and at Pitsford during 1984 [NRA, 1990].

The impact of eutrophication on water treatment and supply, as opposed to its effects on nature conservation interests, are many and varied. They include disruption of treatment owing to the blocking of filters by larger planktonic species [Hayes and Greene, 1984]; increases in turbidity levels in final product water caused by smaller species; tastes and odours caused by algae and fungi; corrosion of iron water-mains through long-term deposition of residual coagulant and seasonal microbiological growth, and discolouration of treated water as a result of elevated iron levels. In order to overcome some of these problems, aluminium sulphate is used in some treatment works as a primary coagulant in algal-laden waters. However, residual aluminium levels must be limited in order to avoid danger to human health. All of these problems have been experienced in the treatment of water from the three reservoirs.

Anticipation of water quality problems at Rutland Water led to the incorporation of a number of features and management options into its design [Harper, 1978]. Yet it was to this reservoir that, in 1989, media attention was drawn, when toxic blue-green algal blooms resulted in the death of sheep and dogs [NRA, 1990]. Blooms occurred also at Pitsford and Grafham, toxicity tests of which proved positive. Problems with substantial growths of blue-green algae during August and early September 1989 were experienced throughout the country as a result, particularly, of long periods of stable weather conditions.

Following the incidents at Rutland, Anglian Water implemented a large scale programme of phosphate control at their major reservoirs, including Grafham, Pitsford and Rutland, commencing in June 1990 [Anglian Water Services Ltd., 1991]. The aim of this programme was to reduce concentrations of dissolved phosphorus by chemical flocculation, through dosing the reservoirs with ferric sulphate. Additionally, helixor airguns installed at Rutland, and bubble curtains in Pitsford and Grafham, were activated, in order to destratify the water, and suppress the growth of blue-green algae.

The ferric sulphate programme has continued from 1991 until the present. It has been estimated that approximately 15,500 tonnes of ferric sulphate was added to Rutland Water between December 1990 and October 1991 [Extence, Brierley and Hine, 1992]. A water chemistry, phytoplankton and zooplankton monitoring programme was instituted by Anglian Water at the three reservoirs, among others, in order to monitor the effects of dosing on algal production [Anglian Water Services Ltd., 1992].

The possible impact of eutrophication on the nature conservation interest of the sites had already caused concern. It was thought to be a likely reason for the decline in bird numbers at Grafham Water, especially in those species such as coot and tufted duck, whose main food source consists of submerged macrophytes [EN, 1991b]. Dosing of the reservoirs with ferric sulphate aroused further concern as to the possible effects on benthic and littoral invertebrates, upon which many species of waterfowl feed. Consequently, a monitoring programme was initiated involving the NRA and Anglian

Water Services Ltd., in order to establish whether or not dosing was producing a deleterious effect.

A first year report, covering Rutland Water and Pitsford and Covenham Reservoirs, concluded that there was clear evidence that dosing operations were damaging invertebrate communities [Extence, Brierley and Hine, 1992]. A further report on Rutland Water produced results which clearly indicated that ferric sulphate additions had led to a qualitative and quantitative loss of benthic macroinvertebrates in areas with elevated iron levels. Most significant was a reduction in both diversity and density of chironomids, an extremely important component of the invertebrate community. Further investigations are being undertaken to confirm these findings.

6.5 West Midlands Lakes

In the industrial and highly urbanised region of the West Midlands, there are several tracts of wild, open country. These include the heathlands of Cannock Chase, Chasewater Heaths and Sutton Park (Fig.6.1). The diversity of habitats provided by these sites is enhanced by the presence of wetlands, including many lakes.

6.5.1 Sutton Park Pools

6.5.1.1 Lake and Catchment Characteristics

Sutton Park SSSI (Grid Reference SP 098974) contains the largest and richest areas of ancient woodland, heath and wetland in the West Midlands [NCC, 1987f]. During the 11th century, Sutton Park was part of a Royal Forest. It remains a fine example of a large medieval park of the compartmented type. Henry VIII presented the Park to the people of Sutton Coldfield in 1528. It was designated an SSSI in 1954, and renotified in 1987.

The well-drained land on the higher ground of Sutton Park is occupied by substantial areas of heath and acidic grassland [NCC, 1987f]. Ancient oak woodlands also cover large parts of the higher ground. The oak-holly-rowan woodland, in particular, is nationally important. The Park encompasses the headwaters of Plants Brook and Longmoor Brook. These streams have been dammed since medieval times to form six main pools. The streams and pools are not only important ecological components of the Park in themselves, but they also enhance the diversity of habitats through the creation of transitional areas between open water and dry ground. The communities represented include wet heath, marshy grassland, valley mire, bog, fen and valley alderwood. Bracebridge Pool, in particular, demonstrates the stages of a succession known as a fen hydrosere.

The catchment consists of the Park itself and an area of urban land surrounding it. There are four consented discharges into the Park of surface water runoff from the latter area [data supplied by A.G. Stanley, NRA, Lichfield]. Four further intermittent consented discharges exist in the form of storm sewage overflows.

6.5.1.2 Current Status

A management plan for Sutton Park prepared for Birmingham City Council (BCC) states that most of the water reaching the wetlands comes from springs and runoff from the Park, and is therefore relatively clean and nutrient poor [Cobham Resource Consultants, 1989]. The document also notes, however, that a golf course situated within the Park may be a source of nutrients, and that excess fertiliser may reach Longmoor Valley mire.

The SSSI status of Sutton Park requires that the NCC is notified of all consented discharges prior to their institution. However, in 1988, an application by Severn Trent Water to use Plants Brook for storm overflow from Thornhill Park Sewage Pumping Station failed to reach the NCC prior to discharges taking place [pers. comm., G. Walker, EN, Shrewsbury]. The discharge in question occurs once or twice a year, and consists of settled sewage, screened to provide safeguard against carry over of gross solids. Both BCC and the NCC objected to this consent being granted. Plants Brook also receives another consented discharge, a mixture of surface water and domestic sewage from 9,300 people, which occurs about once a month.

6.6 North West Midlands Meres

The Meres and Mosses of the north west Midlands (Fig.6.1), also known as the Shropshire-Cheshire Meres, are of national importance as a series of open water and peatland sites [NCC, 1987a]. More than sixty meres and pools over 1 ha in area and over 200 mosses occupy hollows in the glacial drift of the Shropshire-Cheshire Plain [Reynolds, 1979]. Most of the sites occur in distinct local groupings, around the towns of Delamere, Knutsford, Congleton, Ellesmere, Whitchurch and Shrewsbury, in areas of sand and gravel Drift.

Most of the meres may be classified as eutrophic, although their water chemistry is very variable [NCC, 1987a]. Soluble reactive phosphorus concentrations are relatively high, ranging from 20 to 1700μg/l, but these levels have appeared to be natural, derived from apatite in the Drift [Reynolds, 1979]. Reynolds postulated that since phosphorus was relatively abundant plant productivity in the meres is more likely to be controlled by the availability of nitrogen. More recent work by Moss *et al.* [1992] attempted to determine which variables controlled algal productivity in deep (> 3m) and in shallow meres. They established that the former are limited by the availability of nitrogen, whereas algal productivity in the latter is largely determined by the level of zooplankton grazing.

The majority of meres have been isolated from streams for most of their history. Ditches and culverts, mostly constructed during the last two centuries, represent the majority of surface drainage, although direct runoff also contributes to water budgets. It is thought that the meres are in continuity with the groundwater, from which many of them receive their main water supply [Reynolds, 1979]. However, all of the sites under assessment in this study, except Norbury and Quoisley Meres, possess substantial surface inflows.

6.6.1 Aqualate Mere

6.6.1.1 Lake and Catchment Characteristics

Aqualate Mere (Grid Reference SJ 770205) is the largest of the meres, covering some 72 ha. In terms of the distribution of the meres, it lies outside any of the groupings, and is referred to by Reynolds [1979] as isolated. The SSSI, designated in 1956, and renotified in 1987, comprises the Mere and a substantial area of surrounding land [NCC, 1987a]. The scientific interest of the site lies not only in its biological components, but also in its geomorphology. It was declared a National Nature Reserve in March 1992, and is expected to be designated as a site of international importance as waterfowl habitat under the Ramsar Convention on Wetlands in late 1995.

The combination of open water, fen, grassland and woodland found within the SSSI creates a complex of natural features unique in Staffordshire [NCC, 1987a]. As a result, the site supports an outstanding variety of beetles, moths and sawflies. There are nationally important numbers of breeding herons (*Ardea cinerea* L.) and passage shoveler (*Anas clypeata* L.), and the site is regionally significant for breeding waders. The esker formation and related fan, kettleholes and ice-contact slopes on the north side of the mere are of national geomorphological importance.

Lake and catchment data for Aqualate Mere are shown in Table 6.9. It is a relatively large, very shallow waterbody, with a mean water residence time of about seven days. The lake is fed by three streams, the Coley, Humesford and Wood Brooks, and the outflow forms a tributary of the River Meese [OS, 1990c].

Table 6.9: Lake and Catchment Data for Aqualate Mere

[a] This study [b] Reynolds [1979]
[c] Calculated from data supplied by Tim Coleshaw, EN Site Manager, Staffordshire

Catchment Area[a]	50.28km^2
Lake Surface Area[b]	72.5ha
Maximum Depth[c]	0.96m
Mean Depth[c]	0.63m
Volume[c]	0.46 x 10^6m^3
Mean Hydraulic Residence Time[c]	7.55 days
Mean pH	No data
Mean Annual Nitrogen Concentration	No data
Mean Annual Phosphorus Concentration	No data

The catchment is almost entirely rural, consisting of small villages and farmland. There are two small STWs in the catchment, one in the extreme south close to Blymhill and another nearby at Norbury, which discharges into Wood Brook. These STWs together serve about 650 people [data supplied by I. Edwards, NRA, Shrewsbury].

6.6.1.2 Current Status

There has been an occasional problem for many years of pig slurry entering Coley Brook [EN, 1991b]. The other streams are also periodically polluted by farm waste. In March 1992, a major incident occurred, in which 50,000 gallons of slurry overflowed into Coley Brook [pers. comm., G. Walker, EN, Shrewsbury]. Since the piggery has now moved elsewhere, such pollution incidents should no longer occur [letter from T. Coleshaw, EN Site Manager, Staffordshire, 2nd September, 1994].

As a result of problems of access to the site, very little work has been carried out on the limnology of Aqualate Mere, apart from a depth survey in 1984. The Mere is known to be highly eutrophic and subject to siltation [NCC, 1987a]. It contains a grossly impoverished submerged and floating flora, and algal blooms and fish kills have occurred [EN, 1991b]. In 1992 and 1994, the eastern end of the Mere was covered in green algae during warm periods [letter from T. Coleshaw, EN Site Manager, Staffordshire, 2nd September, 1994]. The major influx of nutrients is now thought to enter *via* Wood Brook, which contains total phosphorus concentrations of over $100\mu g/l$.

6.6.2 Chapel Mere

6.6.2.1 Lake and Catchment Characteristics

Chapel Mere (Grid Reference SJ 540519) is situated in the south of Cheshire, in Cholmondeley Park, and is one of the Whitchurch group of meres. It was designated an SSSI as recently as 1987 [NCC, 1987b]. It possesses a variety of vegetation types ranging from submerged and floating leaved plant communities to species rich fen and fen carr. The fringing vegetation is particularly well-developed with a variety of species and communities. More than thirty emergent and fen species occur, making this one of the richest Cheshire meres. The Mere is expected to be designated as a site of international importance as waterfowl habitat under the Ramsar Convention in late 1995.

Lake and catchment data for Chapel Mere are shown in Table 6.10. It is a small, shallow waterbody, with a mean water residence time of about fifty-six days. The Mere appears to be fed mainly by surface flow, since the volume of water entering *via* the three inflows is approximately equal to that of the single outflow [Moss *et al.*, 1992]. The catchment is mainly agricultural, with permanent pasture for dairying being the major land use. Although there are no consented discharges within the catchment of Chapel Mere [letter from G. Hill, NRA, Warrington], Moss *et al.* [1992] are of the opinion that sewage effluent from Cholmondeley Castle enters one of the minor inflows.

Table 6.10: Lake and Catchment Data for Chapel Mere

[a] This study [b] Reynolds [1979]
[c] Smith [1993] [d] Moss *et al.* [1992]

Catchment Area[a]	2.56km^2
Lake Surface Area[b]	6.5ha
Maximum Depth[b]	2.4m
Mean Depth[c]	1.13m
Volume[c]	0.073 x 10^6m^3
Mean Hydraulic Residence Time[d]	56 days
Mean pH[d]	8.39
Mean Annual Nitrogen Concentration[d]	0.46mg/l (NO_3)
Mean Annual Phosphorus Concentration[d]	1267μg/l (TP)

6.6.2.2 Current Status

The water chemistry of Chapel Mere itself is characterised by very high phosphorus concentrations (1267μg/l), and markedly low nitrate and ammonium levels (0.46 and 140μg/l respectively), the latter being attributable to denitrification in the littoral zone and plant uptake [Moss *et al.*, 1992]. Considering the high nutrient loading and relatively long water residence time, dense algal populations and, therefore, a poor submerged macrophyte community, might be expected. When Chapel Mere was designated in 1987, the submerged flora was described as sparse [NCC, 1987b]. By 1992, the community was "of large biomass if of low diversity" [Moss *et al.*, 1992, p.21]. Reduced fish stock levels and the prevalence of large zooplankton species, giving rise to low algal populations and clear water, may account for the presence of an abundant submerged macrophyte community under the high nutrient loadings which this Mere receives.

The agent for the owner of the site recently reported that a faulty septic tank serving one of the houses on the estate has been attended to [letter from C. Hayes, EN, Shrewsbury, 13th September 1994]. However, it is believed that considerable progress could still be made in reducing farm effluents.

6.6.3 Comber Mere

6.6.3.1 Lake and Catchment Characteristics

Comber Mere (Grid Reference SJ 587445) lies within the grounds of Combermere Abbey and Park, and is another of the Whitchurch group of meres. It was designated an SSSI in 1963, and renotified in 1986 [NCC, 1986a]. This site was selected not only as a large wetland habitat, but also for its ornithological interest. A narrow band of swamp and fen vegetation surrounds most of the perimeter of the mere, grading into fen woodland. Comber Mere is the second most important site in Cheshire for wintering wildfowl, and supports one of the largest heronries in the county. Submerged vegetation, which is largely confined to the western end of the mere, consists of substantial beds of macroalgae, with a few species of higher plants. Several patches of floating-leaved species also occur.

Lake and catchment data for Comber Mere are shown in Table 6.11. It is relatively large and deep, compared to many of the other meres, with a correspondingly longer mean water residence time. According to Reynolds [1979], Comber Mere is known to stratify in summer. The Mere is fed by two surface inflows, and is drained by one outflow which forms a tributary of the River Weaver. Moss *et al.* [1992] found, however, that the outflow carries about three times the volume of water as the combined inflows, which suggests the existence of a substantial alternative contribution. Surface runoff is probably the source, rather than groundwater, since the conductivity of the Mere itself is much lower than that of the inflows. The catchment of Comber Mere is largely rural, with a high proportion of permanent pasture. There is one consented discharge from a septic tank serving Combermere Abbey [data supplied by G. Hill, NRA, Warrington].

Table 6.11: Lake and Catchment Data for Comber Mere

[a] This study [b] Reynolds [1979] [c] Moss *et al.* [1992]

Catchment Area[a]	7.97km^2
Lake Surface Area[b]	51.5ha
Maximum Depth[b]	11.8m
Mean Depth[b]	~ 6m
Volume[b]	3.04 x 10^{6}m^3
Mean Hydraulic Residence Time[c]	606 days
Mean pH[c]	8.49
Mean Annual Nitrogen Concentration[c]	0.58mg/l (NO_3)
Mean Annual Phosphorus Concentration[c]	362μg/l (TP)

6.6.3.2 Current Status

The smaller of the two inflows is low in phosphorus and ammonium, with moderate concentrations of nitrate [Moss *et al.*, 1992]. The more substantial inflow exhibits poor water quality, with very high phosphorus, nitrate and ammonium concentrations (1060μg/l, 5.53mg/l and 1142μg/l respectively), indicating a major discharge of an excretory nature along its course. The water quality of Comber Mere itself is characterised by high phosphorus and low nitrogen concentrations (362μg/l and 0.58mg/l respectively), the latter being attributable to denitrification in the littoral zone and plant uptake. The size of the Mere and the presence of moderately large *Daphnia* populations limit the level of phytoplankton production. Water transparency is, therefore, sufficient to support an abundant aquatic macrophyte community. The sparseness of the submerged vegetation may reflect other factors, such as epiphytic burden and wind exposure.

EN [1991b] is of the opinion that the special interest of Comber Mere has been affected. Problems with algal blooms were reported to the NRA in 1989 and 1990. According to Moss *et al.* [1992], however, although blue-green blooms were present in August 1991, they are not a major feature of this Mere. The main sources of nutrients are thought to be the septic tank discharge from Combermere Abbey, and silage effluent from a local farm [EN, 1991b]. It is believed that improvements have recently been made to these point sources, but this has not been confirmed [letter from C. Hayes, EN, Shrewsbury, 13th September 1994].

6.6.4 Norbury Meres

6.6.4.1 Lake and Catchment Characteristics

Norbury Meres (Grid Reference SJ 559492), comprising the Little Mere and the Big Mere, are situated on the Cholmondeley Estate, and belong to the Whitchurch group of meres. This pair of pools, also known as Cholmondeley Meres, was designated an SSSI in 1979, and renotified in 1984 [NCC, 1984a]. Norbury Meres SSSI was selected to represent a mere type consisting of highly eutrophic open water with poorly developed fringing habitats.

Around both the Big Mere and the Little Mere the fen vegetation is restricted to narrow and discontinuous stands which change gradually into alder carr. In addition, the site includes an area of pasture to the east and south of the Meres. This consists of a mosaic of unimproved and semi-improved neutral and acid marshy grassland, the latter being the most species rich. The drainage ditches which traverse the fields and surround the Meres contained notable species of aquatic plants when the site was first designated.

Combined data for the two Norbury Meres are shown in Table 6.12. The Meres are small, shallow pools, about 150m apart, joined by a dyke. Drainage to the Meres is *via* field ditches [Wiggington and Palmer, 1989], although some surface runoff will undoubtedly contribute to the water budget. An outflow leaves the Little Mere on its eastern side, and forms a tributary of the River Weaver. The catchment, which, at 163 hectares, is one of the smallest in this study, consists of agricultural land used mostly for dairying [Smith, 1993]. There are no consented discharges within the catchment of the Norbury Meres [letter from G. Hill, NRA, Warrington].

Table 6.12: Lake and Catchment Data for Norbury Meres

[a] This study [b] Reynolds [1979] [c] Smith [1993]

Catchment Area[a]	$1.63km^2$
Combined Lake Surface Area[b]	3.1ha
Maximum Depth[b]	1.2m
Mean Depth[c]	0.51m
Combined Volume[bc]	$0.016 \times 10^6 m^3$
Mean Hydraulic Residence Time	No data
Mean pH[b]	7.8
Mean Annual Nitrogen Concentration[c]	0.34mg/l (TON)
Mean Annual Phosphorus Concentration[c]	$1632\mu g/l$ (SRP)

6.6.4.2 Current Status

Very little research has been conducted on the Norbury Meres. The NCC carried out surveys of aquatic and marginal vegetation and measurements of conductivity in 1979 and in 1987 [Wiggington and Palmer, 1989]. No submerged or floating-leaved plants were present in either year, and changes in the marginal emergent and fen communities appeared to be minimal. The conductivity of the water, however, increased by 19% in the Big Mere and by 17.5% in the Little Mere, between the two surveys. Water in the Big Mere was reported to be turbid in 1987 and the mud miasmic at the west end, suggesting an ingress of manural waste. The Little Mere exhibited rather clearer water. The existence of a marked conductivity gradient through the Norbury Meres was noted in 1987, whereby the highest levels were to be found in the inflow to the west and the lowest in the Little Mere.

An independent survey of the Norbury Meres, carried out in 1992 [Smith, 1993], revealed that, apart from a few plants of the floating species *Lemna minor*, two macroalgal taxa were present in the Big Mere and three in the Little Mere. Secchi Disc Transparencies of 44.5cm in the Big Mere and 61.5cm in the Little Mere were recorded, which concurs with the observation made in 1987 that the water of the latter is less turbid.

EN [1991b] considers the special interest of Norbury Meres to be affected by eutrophication. The accumulation of black 'ooze' near the inflow to the Big Mere appears to stem from long-term, low-level organic enrichment, probably as a result of cattle slurry entering the drains. The owner's agent reports that agricultural point sources polluting the inflow drain have now been identified and improved, but this has not been confirmed [letter from C. Hayes, EN, Shrewsbury, 13th September 1994].

6.6.5 Petty Pool

6.6.5.1 Lake and Catchment Characteristics

Petty Pool (Grid Reference SJ 620700) lies further north than the other sites, and is included in the Delamere group of meres. It was designated an SSSI as early as 1951, and renotified in 1991 [NCC, 1991b]. The site includes extensive areas of land upstream and downstream of the lake. The wetland communities around Petty Pool represent the most extensive and diverse valley mire system in Cheshire. It is also one of the county's foremost sites for invertebrates. A mosaic of poor and rich fen at the southern end of the site supports a number of plant species of local rarity. Many other plant communities represented around Petty Pool include ancient oak (*Quercus robur* L.) and beech (*Fagus sylvatica* L.) woodland, alder (*Alnus glutinosa* (L.) Gaertner) carr, dry acidic grassland and *Sphagnum* mossland. The open water of Petty Pool supports small patches of floating-leaved macrophytes and a narrow fringe of emergent vegetation.

Lake and catchment data for Petty Pool are shown in Table 6.13. It is small and shallow, with a relatively fast flushing rate of about 60 days. According to Reynolds [1979], the Pool is known to stratify in summer, despite its shallowness. Petty Pool is an artificially enlarged mere, which lies in a deep depression between sand ridges, in an undulating area of glacial Drift [Jones and Savage, 1969]. The Pool receives water from two streams, the larger of which originates from a marsh about 1.5 miles to the west. The outflow runs through a sluice at the southern end and becomes Petty Pool Brook. Moss *et al.* [1992] found that the volume of the outflow was more than twice that of the two inflows combined. This would suggest a significant contribution to the water budget from groundwater. The small catchment is largely rural, with a high proportion of woodland, much of which is owned by the Forestry Commission, but it also includes a golf course, a caravan site and part of a major road, the A556 [OS, 1990d].

Table 6.13: Lake and Catchment Data for Petty Pool

[a] This study [b] Reynolds [1979] [c] Moss *et al.* [1992]

Catchment Area[a]	3.89km^2
Lake Surface Area[b]	11.7ha
Maximum Depth[b]	3.1m
Mean Depth[b]	~ 1.55m
Volume[b]	0.18 x 10^{6}m^3
Mean Hydraulic Residence Time[c]	59 days
Mean pH[c]	8.44
Mean Annual Nitrogen Concentration[c]	0.58mg/l (NO_3)
Mean Annual Phosphorus Concentration[c]	261μg/l (TP)

6.6.5.2 Current Status

Prior to the work carried out by Moss *et al.* [1992], only one published study had been undertaken of Petty Pool, that of Jones and Savage [1969], which described the plant communities. They concluded that the Pool showed early stages in the colonisation of a shallow eutrophic mere. A major factor affecting this process was increased silting around the inlet, allowing colonisation by reeds. This study recorded an algal bloom, lasting a few weeks, which occurred in July 1967.

The work of Moss *et al.* [1992] reveals that submerged plants are very scarce in Petty Pool, while floating species are locally abundant. The main inflow exhibits very high concentrations of phosphorus, nitrate and ammonium (1133μg/l, 6.88mg/l and 3953μg/l respectively), suggesting a major excretal input of nutrients to the stream. However, the minor inflow, which drains woodland and the golf course, contains low phosphorus and moderate nitrate levels (59μg/l and 1.71mg/l respectively). The ecology of the open water of Petty Pool is characterised by relatively high phytoplankton production and small populations of *Daphnia*. Large algal crops, unchecked by grazing, engender turbid conditions during the summer months, which prevent a stable submerged plant community developing. The poor water quality of the Petty Pool Brook inflow has been identified by Moss *et al.* as the likely reason. EN report that for most of the year the water is very clear, but the Pool is still seriously affected by algal blooms [letter from C. Hayes, EN, Shrewsbury, 13th September 1994].

6.6.6 Quoisley Meres

6.6.6.1 Lake and Catchment Characteristics

Quoisley Meres (Grid Reference SJ 549456), comprising the Little Mere and the Big Mere, lie in the south of Cheshire, and are part of the Whitchurch group. They were designated an SSSI in 1963, and renotified in 1987 [NCC, 1987d]. As well as the Meres, the site includes areas of damp grassland. Quoisley Meres SSSI was selected to represent nutrient rich open water with well developed fringing habitats. Around the Big Mere is a continuous narrow reedswamp, which grades into species rich fen, and then into alder (*Alnus glutinosa*) woodland. The surrounding vegetation of the Little Mere is similar in structure but with fewer species. Open water species consist of both white and yellow water-lilies (*Nymphaea alba* L. and *Nuphar lutea* (L.) Sm.), the latter almost completely covering the Little Mere. Aquatic invertebrates include a number of locally and nationally rare species. Quoisley Meres SSSI was designated as a site of international importance as waterfowl habitat under the Ramsar Convention on Wetlands in May, 1994.

Lake and catchment data for Quoisley Meres are shown in Table 6.14. These small, shallow pools lie at the bottom of a basin which also contains Bar Mere [OS, 1990b]. Two small streams enter the Little Mere on its southern side, and an outflow leaves the Big Mere on its south west edge. The Meres are joined by two dykes, along which there is probably a two-way flow, according to local hydrological conditions [Moss *et al.*, 1992]. Direct runoff to the Meres undoubtedly contributes to the water budget, but with an extremely small catchment the amount involved cannot be great. The relatively long water residence time reflects the low rate of inflow from all sources. The catchment is largely agricultural, used primarily for dairying. There are no consented discharges within the catchment [letter from G. Hill, NRA, Warrington].

Table 6.14: Lake and Catchment Data for Quoisley Meres

[a] This study [b] Reynolds [1979]
[c] Smith [1993] [d] Moss *et al.* [1992]

Catchment Area[a]	1.41km^2
Combined Lake Surface Areas[b]	6.2ha
Maximum Depth[b]	2.4m
Mean Depth (Big Mere)[c]	1.2m
Volume	No data
Mean Hydraulic Residence Time[d]	98 days
Mean pH[d]	No data
Mean Annual Nitrogen Concentration[d]	1.09mg/l (NO_3)
Mean Annual Phosphorus Concentration[d]	264μg/l (TP)

6.6.6.2 Current Status

The two inflows into the Little Mere contain moderately high total phosphorus, high nitrate and low ammonium concentrations (123 and 182μg/l, 4.99 and 7.05mg/l, and 65 and 81μg/l respectively), which is consistent with water draining fertile grassland [Moss *et al.*, 1992]. The phosphorus level of the Little Mere (264μg/l) far exceeds those of the inflows, suggesting significant release from the sediments. Ammonium concentrations are also higher in the lakes (261μg/l) than in the inflows, which again is consistent with sediment release. Despite the elevated levels of nutrients, concentrations of chlorophyll *a* are low in both Meres (14.2μg/l), probably as a direct result of abundant *Daphnia*. Fish stocks are unknown, but they are probably low in the Big Mere, where less extensive coverage of lilies provides fewer refuges for zooplankton.

The tenant farmers thought to be responsible for farm effluent pollution of these Meres have now been evicted by the owner [letter from C. Hayes, EN, Shrewsbury, 13th September 1994]. EN is very hopeful that the owner will adopt ideas to safeguard water quality following the preparation of a Whole Farm Conservation Plan by the Farming and Wildlife Advisory Group.

6.6.7 Tabley Mere

6.6.7.1 Lake and Catchment Characteristics

Tabley Mere (Grid Reference SJ 723769) lies towards the north of Cheshire, and is part of the Knutsford group of meres. It was designated an SSSI in 1963, and renotified in 1985 [NCC, 1985b]. The site consists of the Mere itself, a smaller pool to the north, known as Tabley Moat, and an area of acidic, marshy grassland and woodland. Tabley Mere was selected to represent a nutrient rich mere with a well developed aquatic flora. The Mere supports extensive stands of submerged plants, including autumnal starwort (*Callitriche hermaphroditica* L.), horned pondweed (*Zannichellia palustris* L.) and three species of *Potamogeton*. Around the Mere are wide bands of reed swamp which, in places, grades into tall fen. Along the north east shore lies an area of acidic marshy grassland.

Lake and catchment data for Tabley Mere only are given in Table 6.15. The main inflow, Serpentine Water, comes from the north, and a minor one drains land to the south. The Mere has been subject to some degree of landscaping in the past, which may have included raising of water levels [Moss *et al.*, 1992]. There are two outflows from the Mere, the smaller of which feeds Tabley Moat. The volume of the combined outflows exceeds that of the inflows by about 30%, but considering the likely contribution of direct runoff, the groundwater component of the water balance may be relatively insignificant.

The catchment is mainly agricultural, but includes an area of urban land covering south west Knutsford, and part of the M6 motorway [OS, 1990d]. There are three consented discharges within the catchment, two for intermittent storm overflows from sewage pumping stations near Knutsford, which combine with runoff from the M6 discharging into Serpentine Water, and the other for sewage effluent from Tabley House Nursing Home [letter from G. Hill, NRA, Warrington].

Table 6.15: Lake and Catchment Data for Tabley Mere

[a] This study [b] Reynolds [1979] [c] Moss *et al.* [1992]

Catchment Area[a]	11.56km^2
Lake Surface Area[b]	19.4ha
Maximum Depth[b]	4.4m
Mean Depth[b]	~ 2.2m
Volume[b]	0.43 x 10^6m^3
Mean Hydraulic Residence Time[c]	119 days
Mean pH[c]	8.77
Mean Annual Nitrogen Concentration[c]	2.18mg/l (NO_3)
Mean Annual Phosphorus Concentration[c]	323μg/l (TP)

6.6.7.2 Current Status

The inflow from the north of the catchment carries a high chloride concentration (122mg/l), the source of which may be road salt [Moss *et al.*, 1992]. Both inflows are very rich in soluble reactive and total phosphorus (332 and 496μg/l, and 630 and

911μg/l respectively) and in nitrate (9.8 and 4.65mg/l). The stream from the south also contains high concentrations of ammonium (2427μg/l), suggesting contamination by farm waste. Within the Mere itself phosphorus concentrations are high (323μg/l), and nitrate levels (2.18mg/l) generally higher than in most other meres. Phytoplankton populations are low for most of the year, apart from a spring peak, probably as a result of high numbers of *Daphnia*. Occasional blooms of blue-green algae do occur, but they are not a significant component of the community. At present, the Mere maintains its rich aquatic flora, but Moss *et al.* [1992] are of the opinion that the nutrient loading is high enough that only a small change would cause additional switch mechanisms to operate, leading to loss of submerged macrophytes and institution of phytoplankton proliferation. The authors maintain that the source of nutrient enrichment is undoubtedly farm effluent.

EN does not think that agricultural inputs to the Mere are a serious problem, and is far more concerned about runoff from the M6 [letter from C. Hayes, EN, Shrewsbury, 13th September 1994]. The two consented discharges pose serious pollution problems, since they carry accidental spillages of oil and other chemicals from the M6 and Knutsford Services, and raw sewage during storm events. Serpentine Water and the area around its entry into Tabley Mere require dredging on a regular basis in order to remove silt carried by the runoff. A strong smell of sewage persists around one of the pipes, even after a period of dry weather [pers. obs.]. A culverted inflow which feeds the Moat from the north-east apparently contains polluted water, the source of which is thought by the agent for the estate to be agricultural animal waste [letter from C. Hayes, EN, Shrewsbury, 13th September 1994]. However, the smell and appearance of the ditch lead EN to favour enrichment by sewage effluent, possibly as a result of the consented discharge for Tabley House Nursing Home.

6.6.8 Tatton Mere

6.6.8.1 Lake and Catchment Characteristics

Tatton Mere (Grid Reference SJ 755802) lies within the grounds of Tatton Park, and is part of the Knutsford group of meres. It was designated an SSSI in 1963, and renotified in 1986 [NCC, 1986f]. As well as Tatton Mere, the site includes Melchett Mere, and a large area of fen, flushed acidic grassland and woodland. The SSSI was designated as a site of international importance as waterfowl habitat under the Ramsar Convention in May, 1994. Tatton Park, which extends well beyond the boundary of the SSSI, is managed and financed by Cheshire County Council on behalf of the National Trust.

The site was selected to represent meres of moderate fertility, with rich and well developed aquatic flora [NCC, 1986f]. Tatton Mere itself possesses an extensive community of submerged macrophytes, including autumnal pondweed (*Callitriche hermaphroditica*), spiked water milfoil (*Myriophyllum spicatum*) and fennel pondweed (*Potemogeton pectinatus*), among others. A narrow disjunct fen is present around the lake. At the southern end of the Mere is Knutsford Moor, one of the largest areas of fen and reedswamp in Cheshire.

Table 6.16: Lake and Catchment Data for Tatton Mere

[a] This study [b] Reynolds [1979] [c] Moss *et al.* [1992]

Catchment Area[a]	3.9km^2
Lake Surface Area[b]	31.7ha
Maximum Depth[b]	3.5m
Mean Depth[b]	~ 1.75m
Volume[b]	1.11 x 10^6m^3
Mean Hydraulic Residence Time[c]	322 days
Mean pH[c]	8.59
Mean Annual Nitrogen Concentration[c]	0.33mg/l (NO_3)
Mean Annual Phosphorus Concentration[c]	233μg/l (TP)

Lake and catchment data for Tatton Mere only are shown in Table 6.16. Inflows to this relatively large, shallow mere originate in Knutsford Moor to the south, and there is a single outflow to the north. Melchett Mere is bypassed by this outflow, and appears to be fed entirely by groundwater and direct runoff [Moss *et al.*, 1992]. The mean hydraulic residence time of Tatton Mere is over 300 days, and varies little throughout the year. The catchment, which is small in relation to the size of the lake, drains part of

Knutsford to the south and parkland to the north [OS, 1990d]. There is one consented discharge within the catchment for intermittent storm overflow from a sewage pumping station on Knutsford Moor.

6.6.8.2 Current Status

The water quality of Tatton Mere is characterised by rather high soluble reactive and total phosphorus concentrations (183 and 233μg/l respectively), but low nitrate levels (0.33mg/l) [Moss *et al.*, 1992]. Algal productivity appears to be limited by the low nitrogen availability, rather than grazing, since *Daphnia* populations are modest. It might be expected that nitrogen fixing blue-green algae would be abundant in such conditions, yet these are also scarce. Moss *et al.* therefore conclude that Tatton Mere is in a reasonable state. The authors postulate that the source of high phosphorus levels may be septic tank effluent, but that it is more likely to be natural.

EN [1991b], however, identified nutrient enrichment of Tatton Mere as "a serious example of the not infrequent pollution of [Knutsford] Moor resulting from overflows of storm-forced sewage en route for Knutsford STW", that is, the consented discharge referred to above. In 1991, this problem was thought to be in the process of being remedied by the provision of increased storm capacity, and by screening of the envisaged, very infrequent, overflow. However, no work has yet been undertaken to upgrade the facility, and EN is still of the opinion that this source is the primary cause of eutrophication [letter from C. Hayes, EN, Shrewsbury, 13th September 1994].

6.7 Lakes of the Yorkshire Dales

The Yorkshire Dales National Park (YDNP) encompasses an area in which occurs only two natural lakes, Malham Tarn and Semerwater (Fig.6.1). The weathering of the Great Scar Limestone in the west and south has produced a series of features unique in Britain of limestone pavements, cliffs and scars, cave systems and underground rivers [YDNP Committee, 1984]. It is within this landscape that Malham Tarn has evolved. Millstone Grit of the Central Watershed separates this area from the limestone outcrops of

Wensleydale to the north. Here glacial activity has produced a wide valley with a system of narrow sided dales, one of which contains Semerwater. The significance of the lakes of North Yorkshire lies not only in their scientific interest and scenic attraction, but also in their scarcity value [Squance, 1980].

6.7.1 Malham Tarn

6.7.1.1 Lake and Catchment Characteristics

Malham-Arncliffe SSSI (Grid Reference SD 920676) is an extensive (nearly 5000 ha) site of outstanding geological and biological interest [NCC, 1988]. The SSSI was designated in 1955, extended in 1975, and renotified in 1988. Malham Tarn, and the wetland areas associated with it, although covering only a small part of the site, contribute significantly to the interest. The major part of the SSSI is predominantly calcareous grassland, including nationally important limestone pavement. In 1993, Malham Tarn and the surrounding wetlands were designated a site of international importance as waterfowl habitat under the Ramsar Convention on Wetlands. The site is used extensively for courses run by Malham Tarn Field Studies Centre, which has been operating here since about 1950. Most of the SSSI is owned by the National Trust.

Malham Tarn itself, a highly flushed, calcium-rich system, is the highest marl lake in Britain [NCC, 1988]. It is rich in submerged plants, the most abundant species being the macroalga *Chara globularis* Thuill. Six fish species occur here, as well as a population of European crayfish (*Austropotamobius pallipes* Lereboullet). Around the Tarn is a range of wetland vegetation associated with calcicolous fen, willow carr, acidophilous raised mire and soligenous mire that is unique in Britain. The peat deposit of Tarn Moss, adjacent to the lake, contains a continuous pollen record from the late Glacial period to the present day [Pigott and Pigott, 1959]. Malham Tarn is of importance for wildfowl such as coot (*Fulica atra* L.), tufted duck (*Aythya fuligula* (L.)) and great crested grebe (*Podiceps cristatus* L.). The wetland habitats support a large number of notable invertebrates, including a species of caddis (*Agrypnia crassicornis* (McLachlan)) which is confined in Britain to Malham Tarn.

Lake and catchment data for Malham Tarn are shown in Table 6.17. It is a large, shallow lake, with a mean hydraulic residence time of about 80 days [Talling, 1987]. The lake lies in a small basin, with steep limestone cliffs and slopes to the north [Holmes, 1965]. The Tarn is fed by a short stream which arises from springs to the west. The outflow to the south flows overland for about 500m before disappearing underground.

Table 6.17: Lake and Catchment Data for Malham Tarn

[a] This study [b] Holmes [1965] [c] Talling [1987]
[d] Cottrill *et al.* [1991], cited by Hinton [1991a]

Catchment Area[a]	$5.59km^2$
Lake Surface Area[b]	62ha
Maximum Depth[b]	4.25m
Mean Depth[b]	2.32m
Volume[b]	$1.44 \times 10^6 m^3$
Mean Hydraulic Residence Time[c]	77 days
Mean pH	No data
Mean Annual Nitrogen Concentration[d]	0.7mg/l (NO_3)
Mean Annual Phosphorus Concentration[d]	3-17μg/l (TP)

In an area of underground water courses and numerous springs, the water budget of Malham Tarn may not be completely accounted for by surface flows. Indeed, the concept of a definable catchment in such geology may be erroneous. However, the surface water catchment of the Tarn consists mainly of unimproved pasture, used for beef suckler cattle and hill sheep [pers. comm., B. Mercer, EN, Leyburn]. There are no STWs within the catchment, but septic tank systems serve a farm and the various residences associated with the Field Centre.

6.7.1.2 Current Status

Several assessments of the nutrient status of Malham Tarn have been undertaken, with apparently conflicting conclusions. Ratcliffe [1977] noted that, while phytoplankton production is low, the species recorded are those associated with eutrophic conditions. Macrophyte and zooplankton productivity is high, although the latter lacks diversity. Talling [1987] undertook a review of previous findings, and a re-assessment of the nutrient and algal status, drawn from analysis of samples taken over a two year period. Levels of chlorophyll *a* are rarely more than 8μg/l, but peak values exceed 20μg/l, which may affect water transparency. Phytoplankton productivity is thought to be limited by, among other factors, low water temperature (which exceeds 12°C for only nine to ten weeks during the year), low water retention times and winter-spring phosphorus concentrations of only about 8μg/l. There is no evidence from the phosphorus data that the loading of this element has increased. Nitrate levels appear to have doubled over forty years, but are still relatively low (0.7mg/l). Other long-term changes include the loss of the diatom *Asterionella formosa*, indicating some instability, and a recent increase in *Chara* species and other macrophytes.

Comments on the work of Talling [1987] by Hinton [1991b] concluded that, from the evidence currently available, Malham Tarn is not suffering from eutrophication. Despite the mild enrichment which has occurred over the past forty years, the Tarn is still mesotrophic. However, he did maintain that, with regard to nutrient inputs, developments within the catchment should be closely monitored, and attempts made to reduce anthropogenic nutrient sources.

In its capacity as landowner, the NT undertook a survey of the efficacy of sewage facilities during 1988 and 1989, as part of a nationwide environmental audit of its properties [pers. comm., R. Jarman, NT, Cirencester]. The results of this survey found that the sewage system serving the Field Centre was inadequate and in need of redevelopment. However, tracing studies revealed that effluent did not enter the Tarn, except during periods of high rainfall. It was felt that a sewage system involving the use of constructed reedbeds would be appropriate. Consequently, a meeting was held with a representative of Camphill Water in November 1992. By September 1994, such a system had not been installed, although the design is near completion [letter from B. Mercer, EN, Leyburn, 6th September 1994].

6.7.2 Semerwater

6.7.2.1 Lake and Catchment Characteristics

Semerwater SSSI (Grid Reference SD 913865) consists of the lake itself together with its inflow streams and marginal wetland and meadow habitats [NCC, 1986e]. The site was designated in 1975, and renotified in 1986. Part of the SSSI is owned and managed by the Yorkshire Wildlife Trust as a nature reserve. The recreational uses of the lake include water skiing, sailing and angling.

Semerwater, like Malham Tarn, is noted both for its geological and biological interest. The broad and deep lakehead, the terminal moraine which retains the lake and the presence of erratic boulders, notably the Carlow Stone, indicate its glacial origins [NCC, 1986e]. In the centre, the lake was further deepened by the glacier to a depth of 10m.

The vegetation surrounding the lake provides a good example of hydroseral development from aquatic to terrestrial communities. Stands of emergent aquatic vegetation are established along the western and eastern shoreline, whereas willow carr has developed on the northwest edge. There are also areas of reedbed, sedge fen, swamp, marsh and rush pasture. The wetland vegetation continues for some distance upstream along the inflow. Other biological features of the site are the large number of species of mayfly, a considerable population of crayfish (*Austropotamobius pallipes*), and the presence of the cladoceron *Leptodora kindti* (Focke), which is rare in Yorkshire.

Lake and catchment data for Semerwater are given in Table 6.18. It is a large, shallow lake with a short water residence time, fed by several fast-flowing streams. The River Bain drains the lake, before joining the River Ure. The outflow was deepened by the local council in 1937, in order to assist drainage in fields beyond the head of the lake. This resulted in a lowering of the water level by about 0.5 m, which, added to progressive silting, has caused the emergence of fen around the inflows. Compared to that of Malham Tarn, the catchment of Semerwater is large. It consists mainly of rough grazing, but with some improved grassland in the valley bottoms [Barlow, 1994]. A privately-owned forestry plantation of nearly 170ha is located to the south. There is one consented discharge within the catchment, which provides for a septic tank serving two dwellings and six to ten people [pers. comm., P. D'Arcy, NRA, Thirsk]. There are many other septic tanks in the catchment which do not require consents, serving about 75 p.e. [Barlow, 1994].

Table 6.18: Lake and Catchment Data for Semerwater

[a] This study [b] Barlow [1994]
[c] Yorkshire Water Authority, cited in Squance [1980]

Catchment Area[a]	$43.61 km^2$
Lake Surface Area[b]	26.65ha
Maximum Depth[b]	10.5m
Mean Depth[b]	1.95m
Volume[b]	$0.52 \times 10^6 m^3$
Mean Hydraulic Residence Time[b]	15.8 days
Mean pH[c]	7.7
Mean Annual Nitrogen Concentration[c]	<0.01mg/l (NO_3)
Mean Annual Phosphorus Concentration[c]	10μg/l (PO^4)

6.7.2.2 Current Status

Semerwater was considered by Ratcliffe [1977] to be moderately eutrophic for an upland lake, with a phytoplankton assemblage indicative of rich waters. However, the results from surveys carried out by Squance [1980] indicate that the lake is mesotrophic. He considered that high turbidity is probably one of the most important factors affecting the aquatic flora and fauna, but that this is caused by suspended sediment, rather than phytoplankton productivity. The shallowness of the lake, exposure to wind, and the loose, silty nature of the sediments were the factors which contribute to poor water clarity. Additionally, the use of power boats on the lake not only increase turbidity, but also cause disturbance and physical damage to plants.

In July 1989, fish mortalities occurred at Semerwater, which prompted an investigation by Yorkshire Water Authority (YWA) [letter from YWA, York to NCC, Leyburn, 1st

September, 1989]. Water samples were found to contain large quantities of algae (32000 cells/ml), with the diatom *Asterionella formosa* and very small green flagellates constituting 40% and 47% of the total count respectively. Since none of the algae present were known to produce toxins, it was concluded that the fish mortalities may have been the result of depleted levels of dissolved oxygen and elevated pH, caused by the high level of phytoplankton productivity. However, a further loss of fish in August 1989 was attributed to the influence of algal blooms. In September 1990, monitoring of Semerwater by the NRA showed that *Anabaena* had developed to levels with the potential for surface scum formation [letter from NRA, York to NCC, Leyburn, 24th September 1990], and were just above the NRA's national limit of 12 colonies or filaments per ml [letter from G. Hawley, NRA, Exeter, 11th November 1991].

Chapter 7

Results and Recommendations

Results obtained from the modelling of external phosphorus loadings are discussed initially on a site specific basis in Section 7.1. Section 7.2 provides a general discussion of these results, while Section 7.3 aims to draw together the site specific recommendations, and to integrate them into conclusions drawn from discussion of the legislative framework (Chapter 4).

The degree of error inherent in the export coefficient approach (Sections 3.1 and 3.2), coupled with the lack of available data to calibrate the results (Section 5.3.1), means that the loadings calculated for each site may not be taken as absolute. Information cited which does not originate with the author is credited in the appropriate Sections of Chapter 6. Only data used for the first time will be referenced here. All maps are from Ordnance Survey Landranger 1:50000 Series. In graphs used to illustrate loading levels, trophic categories are labelled U, O, M, E and H, denoting ultra-oligotrophic, oligotrophic, mesotrophic, eutrophic and hypertrophic, respectively.

7.1 Site Specific Results and Recommendations

7.1.1 Loe Pool

7.1.1.1 Results

A description of this site may be found in Section 6.1.1. Land use results for the catchment of Loe Pool (Fig.7.1), calculated from data obtained at the Institute of Terrestrial Ecology (ITE), are shown in Table 7.1. The catchment consists mainly of grassland, with nearly 53% of the area under permanent pasture, a further 23% temporary grassland, and arable land contributing 10%. Information in the parish returns on the types of holdings reveals that nearly 65% of farms operate on a part-time basis. Of the remainder, two thirds are either specialist or mainly dairy farms.

Table 7.1: Land Use in the Catchment of Loe Pool

Land Use	Area (ha)	Percentage
Temporary Grassland	1253.70	23.20
Permanent Grassland	2852.45	52.79
Arable	544.42	10.08
Rough Grazing	168.69	3.12
Settlements and Roads	431.32	7.98
Woodland	87.22	1.61
Rivers	65.18	1.21
Total	5402.97	100.00

Total livestock numbers from parish returns required reduction by 50.68% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. The results of this exercise are shown in Table 7.2, and confirm the predominance of cattle in this catchment. Two specialist poultry units exist, supporting up to 36,000 birds, one of which lies within the catchment, but the other is probably situated outside.

Table 7.2: Livestock Levels in the Catchment of Loe Pool

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	9118	1704	1236	19286	530

External phosphorus loading results for Loe Pool are detailed in Table 7.3. Total crude load is predicted to be between 7.27 and 9.44t/a, which, on an areal basis, converts to 13.07 and 16.98g/m^2/a. These results give a potential mean annual total phosphorus (TP) concentration from external sources of between 366 and 417μg/l, and an inflow TP concentration within the range of 1092 and 1283μg/l, which places Loe Pool in the hypertrophic category (Fig.7.2).

Table 7.3: Predicted External Phosphorus Loadings on Loe Pool

Parameters	Low Export	High Export
Total Load (t/a)	7.27	9.44
Areal Load ($g/m^2/a$)	13.07	16.98
Lake Concentration ($\mu g/l$)	366	417
Inflow Concentration ($\mu g/l$)	1092	1283

Table 7.4 provides a division of external phosphorus loading according to source. Sewage effluent may contribute between 62% and 81% of the load. A measured mean annual TP concentration of 270$\mu g/l$ is recorded for the lake. Although this value cannot be compared directly with the predicted results, it is likely that biological uptake of soluble phosphate in sewage effluent may occur in the River Cober and around the inlet, which would not be reflected in results obtained from in-lake water sampling.

Table 7.4: Export of Phosphorus to Loe Pool According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.58	7.92	1.43	15.12
Organic Fertiliser	0.57	7.91	1.72	18.27
Sewage Effluent	5.89	81.03	5.89	62.38
Settlements and Roads	0.22	2.97	0.37	3.88
Woodland	0.00	0.01	0.00	0.05
Direct Precipitation	0.01	0.15	0.03	0.29

7.1.1.2 Recommendations

Loe Pool is one of several sites proposed by the National Rivers Authority (NRA) for designation as a 'sensitive area' under the Directive on Urban Waste Water Treatment (UWWT), but rejected by the Department of the Environment (DoE). The combined population equivalent (p.e.) of the two STWs discharging into rivers leading to Loe Pool lies above the required 10,000 during the summer, but falls below it in the winter.

Fig.7.2 illustrates the effect of possible measures on the predicted inflow concentration for Loe Pool. Designation of the site as a 'sensitive area' would require 80% reduction in effluent phosphorus levels, lowering the inflow concentration to between 380 and 541μg/l. Alternatively, complete diversion of sewage effluent to sea would produce an inflow concentration of 144 to 283μg/l. A cost-benefit analysis may be necessary in order to establish which of these two options may be economically and ecologically more viable, taking into account the effect of diverting the proportion of inflow volume contributed by the effluent.

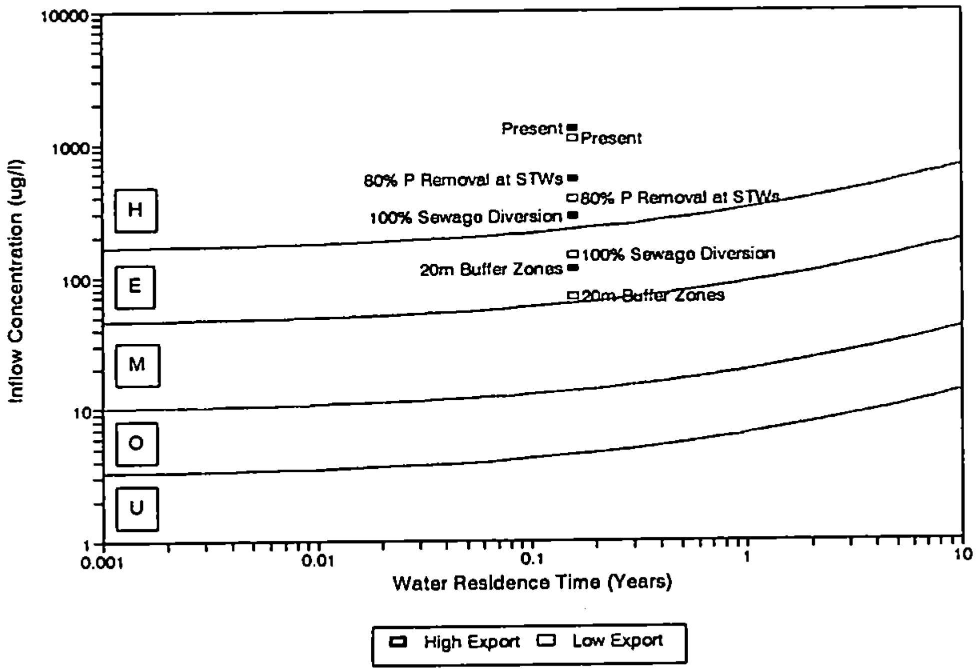

Fig.7.2: Effect of Reductions in External Phosphorus Loadings on Loe Pool

Sewage diversion alone would place Loe Pool in the eutrophic to hypertrophic category, so that it is also necessary to lower contributions from diffuse agricultural sources. Since organic and inorganic sources contribute roughly equal amounts of phosphorus, this may be achieved through the establishment of 20m riparian buffer zones. This measure would theoretically produce an inflow concentration of between 71 and 114μg/l, placing Loe Pool in the eutrophic category, as illustrated in Fig.7.2.

Since 1946, the River Cober between Helston and Loe Pool has been confined to a straight, channelised course in order to prevent flooding upstream. The restoration of the lower reach of this river to a meandering course along its floodplain could be achieved without increasing the incidence of flooding. This measure would not only re-establish floodplain functions of reducing nutrient and sediment loads, but could also enhance the nature conservation value of the site. The benefits of such work are impossible to quantify, but if the problem of nutrient-rich sewage effluent from Helston STW is not dealt with by tertiary treatment or diversion to sea, restoration of the floodplain could contribute to a reduction in loading on Loe Pool. Funding for the work could be contributed by South West Water Services Ltd., in conjunction with NRA South West.

7.1.2 Slapton Ley

7.1.2.1 Results

A description of this site may be found in Section 6.1.2. Land use results for the catchment of Slapton Ley (Fig.7.3), based on data obtained at the ITE, are shown in Table 7.5. They indicate that the land use is very similar to that of the catchment of Loe Pool, in that it consists mainly of grassland. Nearly 52% of the land is under permanent pasture, a further 19% is temporary grassland, and arable land forms about 14%. Information held in the agricultural returns on the types of holdings in each parish indicate that 44% of farms operate on a part-time basis. Of the remainder, half are either specialist or mainly dairy farms, and over 20% rear sheep, or cattle and sheep. Cereals and general crops are grown on only 8% of farms run full-time.

Table 7.5: Land Use in the Catchment of Slapton Ley

Land Use	Area (ha)	Percentage
Temporary Grassland	906.62	19.29
Permanent Grassland	2418.96	51.46
Arable	674.02	14.34
Rough Grazing	310.78	6.61
Settlements and Roads	323.65	6.89
Woodland	29.69	0.63
Rivers	36.62	0.78
Total	4700.35	100.00

Fig.7.3: The Catchment of Slapton Ley

Data on livestock include all figures for Slapton parish which lies wholly within the catchment, and 56% of those for the remaining parishes. The results, shown in Table 7.6, indicate the importance of sheep and cattle.

Table 7.6: Livestock Levels in the Catchment of Slapton Ley

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	6153	16235	1140	916	290

Phosphorus loading results for Slapton Ley are detailed in Table 7.7. Total crude load is predicted to be between 1.81 and 3.70t/a, which, on an areal basis, converts to 2.35 and 4.81g/m^2/a. These results produce a potential mean annual TP concentration from external sources of between 60 and 122μg/l, and an inflow TP concentration within the range of 104 to 250μg/l, placing Slapton Ley in the eutrophic to hypertrophic category. A measured inflow mean for TP of 170μg/l correlates well with the predicted figures. Previous modelling work on this catchment predicted a higher crude total phosphorus loading of 4.8t/a, but a hydrologically and biologically relevant load of only 1.58t/a [Johnes and O'Sullivan, 1989].

Table 7.7: Predicted External Phosphorus Loadings on Slapton Ley

Parameters	Low Export	High Export
Total Load (t/a)	1.81	3.70
Areal Load (g/m^2/a)	2.35	4.81
Lake Concentration (μg/l)	60	122
Inflow Concentration (μg/l)	104	250

Table 7.8 identifies the proportion of external phosphorus loading attributable to each source. Sewage effluent contributes between 15% and 31% of the load, while diffuse agricultural sources account for 59% to 76%, depending on whether low or high export coefficients are applied. Urban and woodland sources and direct precipitation do not contribute significant amounts of phosphorus.

Table 7.8: Export of Phosphorus to Slapton Ley According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.56	30.99	1.31	35.40
Organic Fertiliser	0.50	27.80	1.51	40.75
Sewage Effluent	0.57	31.39	0.57	15.34
Settlements and Roads	0.16	8.95	0.28	7.43
Woodland	0.00	0.02	0.00	0.04
Direct Precipitation	0.02	0.85	0.04	1.04

7.1.2.2 Recommendations

Two STWs are located in the catchment of Slapton Ley, one inland to the north, and the other, near the coast at Slapton. Diversion of effluent from Slapton STW has been mooted as a possible measure for reduction of nutrient inputs. However, the case for diversion does not appear to be justified by the results of this study. The contribution to TP loading from Slapton STW lies between 5.6% and 11.5%, which is far less than the 20% minimum reduction necessary to produce an improvement in water quality.

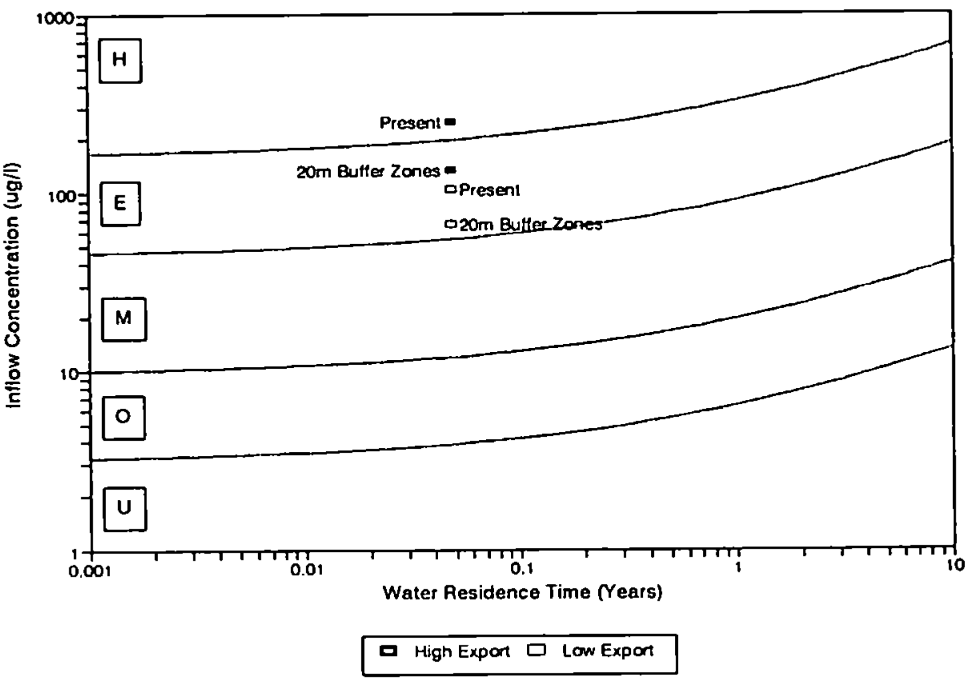

Fig.7.4: Effect of Reductions in External Phosphorus Loading on Slapton Ley

It follows therefore that, in order to reduce external loadings on Slapton Ley, diffuse agricultural sources of phosphorus need to be addressed. Since neither organic nor inorganic sources predominate, this may be achieved by the establishment of 20m riparian buffer zones. This measure, for which compensation to farmers is available under the pilot Habitat Scheme, would theoretically produce a mean annual TP concentration of between 41 to 73μg/l, and an inflow concentration within the range of 66 to 133μg/l, placing Slapton Ley in the eutrophic category (Fig.7.4). This level of reduction would, however, require the interception of field drainage flow as an integral part of the Habitat Scheme, rather than as an additional option.

7.1.3 Bay Pond

7.1.3.1 Results

A site description of Bay Pond may be found in Section 6.2.1. Land use results for the catchment (Fig.7.5) are shown in Table 7.9. It extends over 313 hectares, about 75% of which consists of agricultural land, divided more or less equally between permanent grassland, temporary pasture, and arable land. Roads and settlements cover a further 18% of the catchment. Parish data reveal that 80% of farms are operated on a part-time basis. The remainder are either dairy, or cattle and/or sheep rearing farms, although several horticultural units also exist.

Table 7.9: Land Use in the Catchment of Bay Pond

Land Use	Area (ha)	Percentage
Temporary Grassland	78.03	24.92
Permanent Grassland	87.67	28.00
Arable	67.44	21.54
Rough Grazing	4.00	1.23
Settlements and Roads	55.88	17.85
Woodland	20.23	6.46
Rivers	0.00	0.00
Total	313.10	100.00

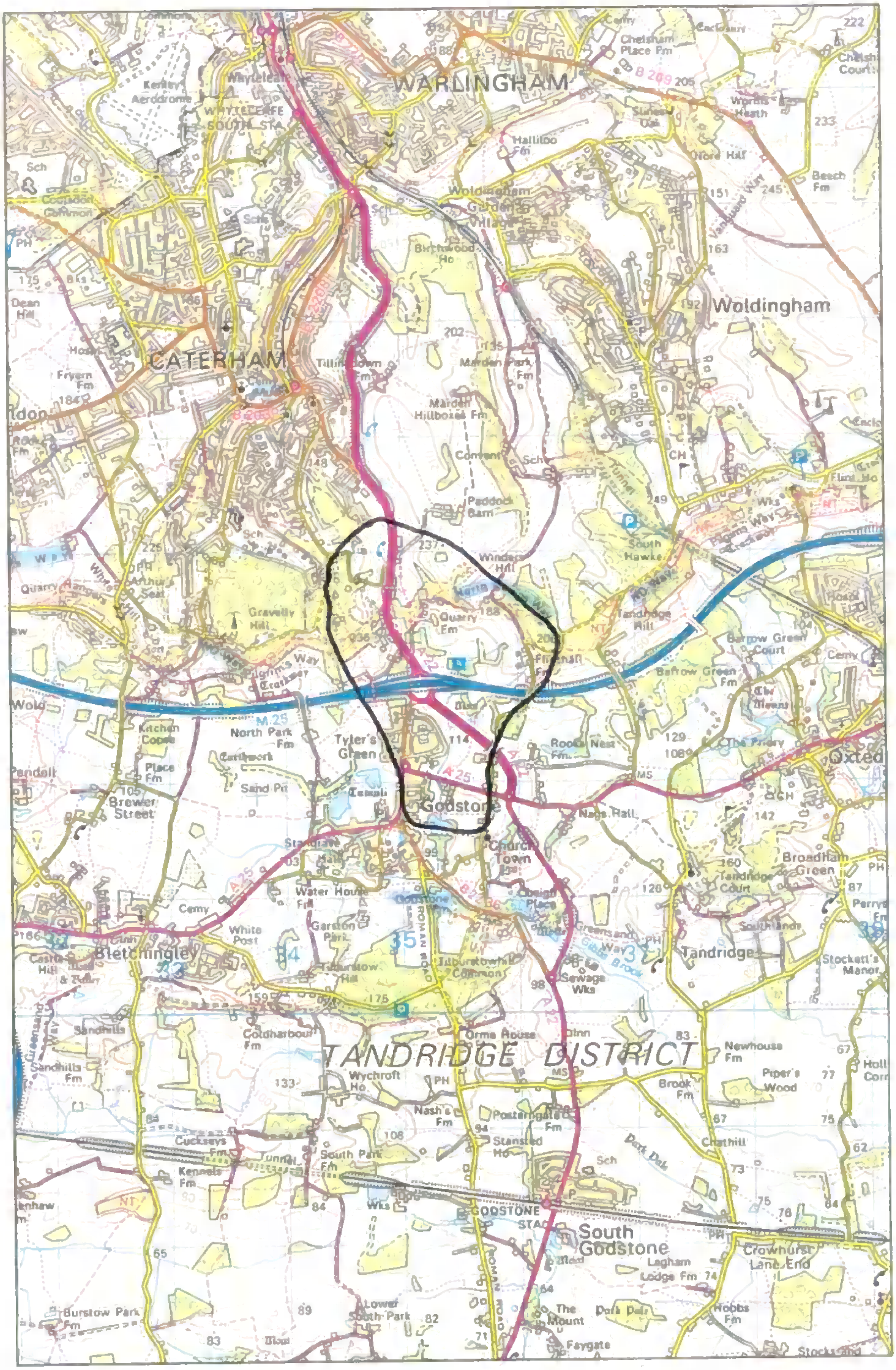

Fig.7.5: The Catchment of Bay Pond

Total livestock numbers for the parishes required reduction by 78%. Such disparity between parish and catchment areas renders the results for livestock numbers, shown in Table 7.10, open to question.

Table 7.10: Livestock Levels in the Catchment of Bay Pond

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	255	340	9	133	35

Predicted phosphorus loadings for Bay Pond are detailed in Table 7.11. Total crude load lies between 0.09 and 0.20t/a, which gives an areal loading within the range of 3.56 and 7.87g/m^2/a. These figures convert to a mean annual TP concentration from external sources of 15 to 34μg/l, and an inflow TP concentration between 17 and 46μg/l, placing Bay Pond in the mesotrophic category (Fig.7.7). The predicted results do not compare well with measured values for phosphate which range from 25 to 4520μg/l, as illustrated in Fig.7.6.

Table 7.11: Predicted External Phosphorus Loadings on Bay Pond

Parameters	Low Export	High Export
Total Load (t/a)	0.09	0.20
Areal Load (g/m^2/a)	3.56	7.87
Lake Concentration (μg/l)	15	34
Inflow Concentration (μg/l)	17	46

Table 7.12 provides a division of external phosphorus loading according to source. Inorganic fertilisers contribute about 50%, while animal waste varies between 19% and 26%. Settlements and roads confer a further 23% to 30%, reflecting the more urbanised nature of the catchment. There are no STWs, but it is likely that septic tanks which do not require consents to discharge are present. Woodland and direct precipitation do not contribute insignificant amounts of phosphorus to the loading.

Table 7.12: Export of Phosphorus to Bay Pond According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.04	50.11	0.10	49.93
Organic Fertiliser	0.02	18.69	0.05	25.75
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.03	30.15	0.05	23.20
Woodland	0.00	0.22	0.00	0.49
Direct Precipitation	0.00	0.56	0.00	0.63

7.1.3.2 Recommendations

As noted above, it appears that external inputs of phosphorus are not the prime source of high phosphate concentrations of up to 4520μg/l recorded for this lake. Fig.7.6 illustrates the disparity between measured phosphate concentrations and predicted mean TP levels.

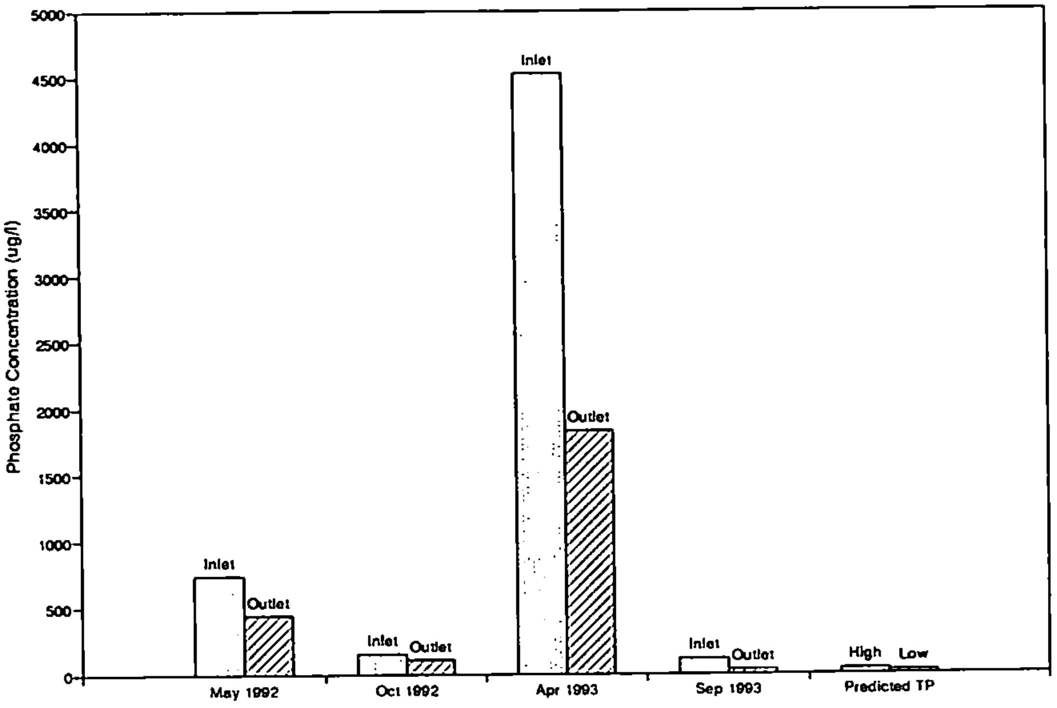

Fig.7.6: Comparison of Measured Phosphate and Predicted Mean Total Phosphorus Concentrations for Bay Pond

The problem of high phosphate levels in Bay Pond may stem from one of two factors. First, ponds such as this, created by the damming of rivers, are prone to siltation. Work carried out in 1985 to remove sediment in order to increase the depth was followed by the construction of a silt trap the next year. The effectiveness of such devices in accumulating particulate phosphorus may be negated if transformation to more soluble forms occurs. In such circumstances, and during heavy rainfall, the sediment trap may be acting as a source of phosphorus. Second, there may be sources within the catchment which are not detected by the methods used in this study. However, there is evidence to suggest that the first hypothesis is more likely. Nitrate levels recorded at the same time as particularly high phosphate concentrations are quite moderate, ranging from 6 to 16mg/l. This suggests the presence of a source of phosphorus internal to the hydrological system, since an external source would be expected to generate high levels of nitrogen as well.

External phosphorus loadings on Bay Pond do not appear to be excessive, as illustrated in Fig.7.7. It may therefore be the case that internal sources are the cause of reduced water transparency, and of the consequent demise of submerged macrophytes. On this evidence, no reduction strategies may be recommended.

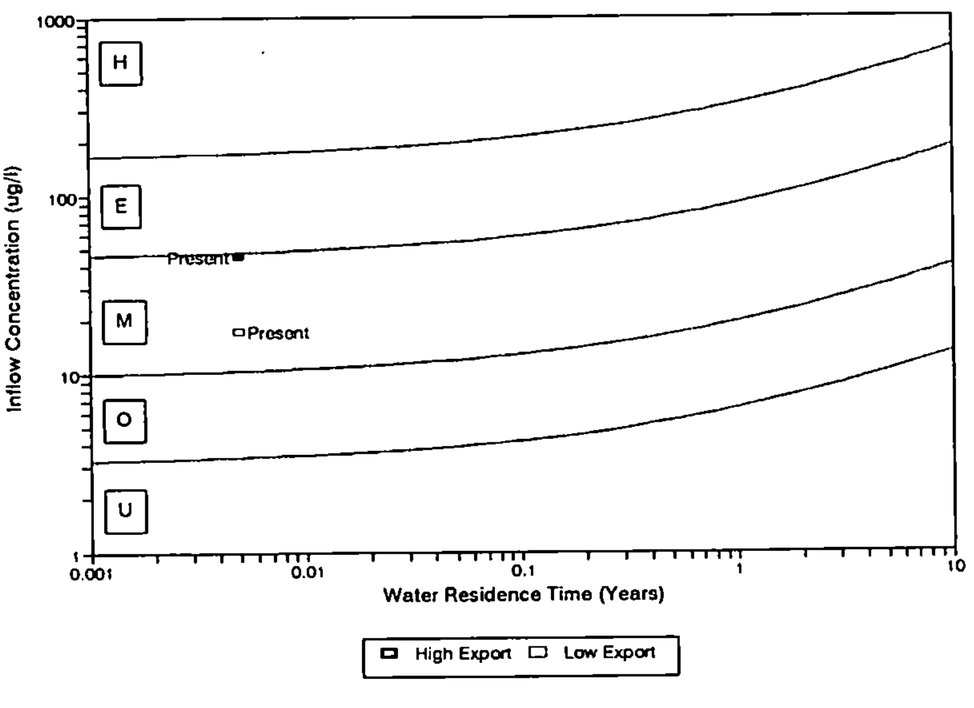

Fig.7.7: Predicted External Phosphorus Loadings on Bay Pond

7.1.4 Hedgecourt Lake

7.1.4.1 Results

A description of this site may be found in Section 6.2.2. Land use results for the catchment of Hedgecourt Lake (Fig.7.8), based on ITE data, are shown in Table 7.13. Nearly 60% of the catchment consists of agricultural land, half of which is under arable production. Settlements and roads account for 23% of the land use, while woodland covers over 16%. Data from the parish returns reveals that two thirds of farms are operated on a part-time basis. Of the remainder, a third are specialist dairy farms, while the rest are mostly horticultural units.

Table 7.13: Land Use in the Catchment of Hedgecourt Lake

Land Use	Area (ha)	Percentage
Temporary Grassland	127.54	13.11
Permanent Grassland	163.25	28.00
Arable	284.83	29.28
Rough Grazing	5.10	0.52
Settlements and Roads	221.92	22.81
Woodland	159.00	16.35
Rivers	11.05	1.14
Total	972.69	100.00

Total livestock numbers from parish returns required reduction by over 77% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. The results are shown in Table 7.14, but the difference between parish and catchment areas means that they are not completely reliable.

Table 7.14: Livestock Levels in the Catchment of Hedgecourt Lake

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	710	946	194	1010	93

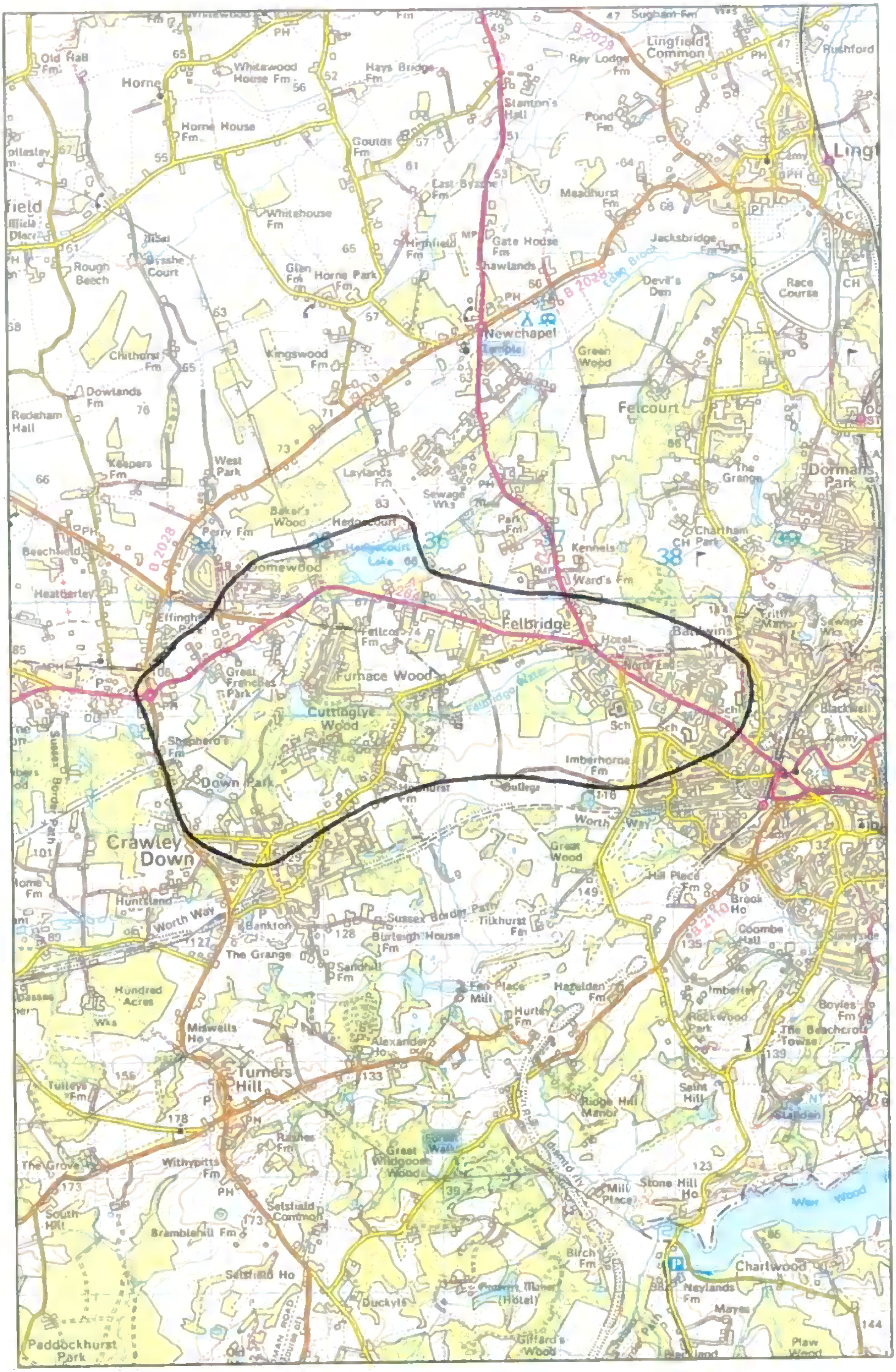

Fig.7.8: The Catchment of Hedgecourt Lake

Predicted external phosphorus loadings for Hedgecourt Lake are listed in Table 7.15. Total crude load is predicted to be between 0.40 and 0.75t/a, which, on an areal basis, converts to 2.35 and 4.38g/m^2/a. These results yield a potential mean annual TP concentration from external sources of between 276 and 514μg/l, and an inflow TP concentration ranging from 758 to 1619μg/l, placing Hedgecourt Lake well into the hypertrophic category. Actual concentrations may be higher, allowing for the possibility that internal sources of phosphorus may exist, but there are no data with which to assess the results.

Table 7.15: Predicted External Phosphorus Loadings on Hedgecourt Lake

Parameters	Low Export	High Export
Total Load (t/a)	0.40	0.75
Areal Load (g/m^2/a)	2.35	4.38
Lake Concentration (μg/l)	276	514
Inflow Concentration (μg/l)	758	1619

Table 7.16 identifies the proportion of external phosphorus loading which may be attributed to each source. Diffuse agricultural sources account for 52% to 62% of the load, while runoff from settlements and roads contribute 25% to 27%. Sewage sources, in the form of septic tanks, provide between 10% and 20% of the phosphorus load.

Table 7.16: Export of Phosphorus to Hedgecourt Lake According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.16	38.47	0.31	41.13
Organic Fertiliser	0.05	13.04	0.16	21.00
Sewage Effluent	0.08	19.79	0.08	10.62
Settlements and Roads	0.11	27.45	0.19	25.05
Woodland	0.00	0.39	0.01	1.06
Direct Precipitation	0.00	0.85	0.01	1.14

7.1.4.2 Recommendations

The catchment of Hedgecourt Lake includes extensive road networks and urban development, as well as areas of intensive agriculture. The lake itself is prone to silting, since it is formed by the damming of a river. These factors combined appear to have led to the waters of Hedgecourt Lake becoming hypertrophic. The extremely high predicted inflow TP concentrations (758 to 1619μg/l) have proved difficult to reduce on the basis of lowering inputs from existing sources, as illustrated in Fig.7.9.

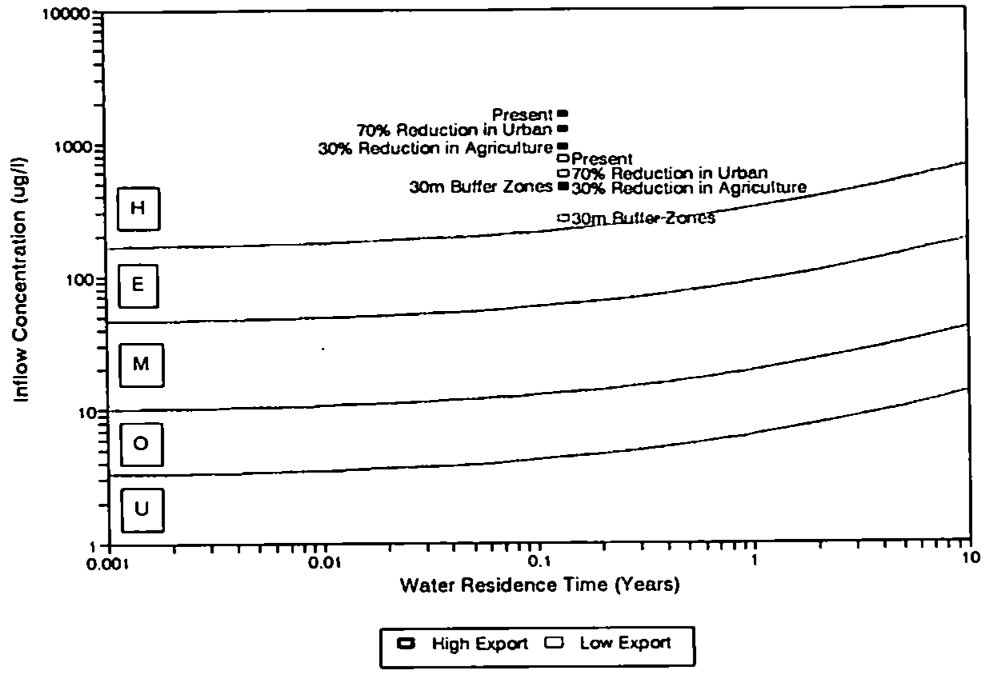

Fig.7.9: Effect of Reductions in External Phosphorus Loadings on Hedgecourt Lake

The application of stringent reduction strategies of lowering inputs from urban runoff by 70%, from diffuse agricultural sources by 30%, and by the establishment of 30m riparian buffer zones, would theoretically produce an inflow TP concentration of 268 to 473μg/l. This would result in Hedgecourt Lake remaining in the hypertrophic category, albeit reducing phosphorus concentrations to about one third of their previous level.

Since it is unlikely that land use in the catchment will change sufficiently to alter the trophic status of Hedgecourt Lake, the problem needs to be addressed within the hydrological system. If the objective of restoration, from a nature conservation viewpoint, is to re-establish submerged macrophytes, water transparency must be enhanced. The lake is used for sailing and angling, and is stocked with fish on a periodic basis. Providing cooperation from owners and users is forthcoming, the possibility exists of applying biomanipulation to establish a trophic structure capable of maintaining an appropriate level of water clarity under high nutrient loadings. The introduction of piscivorous fish and large *Daphnia* may effectively control total algal biomass sufficiently to achieve this [Carpenter *et al.*, 1995].

7.1.5 Barton Broad

7.1.5.1 Results

A description of this site may be found in Section 6.3.1. Land use results for the catchment of Barton Broad (Fig.7.10), calculated from ITE data, are shown in Table 7.17. The catchment covers some 13300 hectares, of which over 55% is devoted to arable land. The rest of the agricultural land is mostly permanent pasture, while settlements and roads cover nearly 10%. Data contained in parish returns reveal that 36% of farms operate on a part-time basis. Of the remainder, 69% are units growing general crops, and 14% are specialist poultry, or pig and poultry enterprises.

Table 7.17: Land Use in the Catchment of Barton Broad

Land Use	Area (ha)	Percentage
Temporary Grassland	793.72	5.97
Permanent Grassland	3633.60	27.32
Arable	7386.60	55.54
Rough Grazing	30.92	0.23
Settlements and Roads	1295.38	9.74
Woodland	91.05	0.68
Rivers	68.72	0.52
Total	13300.00	100.00


Frogshall
Dams
Bridge Fm
Lodge Fm
The Grove
Home Fm
38

Data on livestock include all figures for twelve parishes which lie wholly within the catchment, and those for a further fourteen reduced by 46%. The similarity between parish and catchment areas suggests that the results (Table 7.18) are reasonably reliable. However, whether the specialist poultry or pig and poultry units are, first, located inside the catchment, and second, dispose of their waste within it, remains unknown.

Table 7.18: Livestock Levels in the Catchment of Barton Broad

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	3264	1217	14007	644751	148

Phosphorus loading results for Barton Broad are shown in Table 7.19. Total crude load is predicted to be between 7.29 and 13.03t/a, which, on an areal basis, converts to 10.28 and 18.40g/m^2/a. These results yield a potential mean annual TP concentration from external sources of between 193 and 345μg/l, and an inflow TP concentration within the range of 416 and 845μg/l, which would place Barton Broad in the hypertrophic category. A measured mean annual TP concentration for Barton Broad of 150μg/l is recorded for 1993. Considering this value includes substantial phosphorus loading from the sediments, the model appears to overpredict considerably for this lake.

Table 7.19: Predicted External Phosphorus Loadings on Barton Broad

Parameters	Low Export	High Export
Total Load (t/a)	7.29	13.03
Areal Load (g/m^2/a)	10.28	18.40
Lake Concentration (μg/l)	62	111
Inflow Concentration (μg/l)	104	212

Contributions to the phosphorus loading by source are shown in Table 7.20. Under both high and low coefficients, the proportion of phosphorus from inorganic fertilisers varies little at around 50%, whereas organic sources range between 14% and 24%. Sewage effluent contributes between 15% and 27% of the load, although there is evidence to suggest that the amount calculated for this component is overestimated. Data on measured mean annual TP loads from three STWs above Barton Broad, which operate tertiary treatment and serve more than 98% of the population, provide totals for 1991 of 1.23t/a, and for 1992 of 0.58t/a. These figures are far less than the 1.96t/a calculated by the model. A partial explanation for this discrepancy might be that the phosphorus removal rates achieved by STWs discharging into the River Ant may be higher than the mean of 82% calculated from data on STWs on the River Bure.

Table 7.20: Export of Phosphorus to Barton Broad According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	3.63	49.75	6.81	52.26
Organic Fertiliser	1.04	14.29	3.12	23.96
Sewage Effluent	1.96	26.86	1.96	15.01
Settlements and Roads	0.65	8.89	1.10	8.45
Woodland	0.00	0.01	0.01	0.04
Direct Precipitation	0.01	0.19	0.03	0.27

7.1.5.2 Recommendations

The installation of phosphate stripping capability at Stalham, Worstead and South Repps STWs, situated above Barton Broad, has reduced significantly the loading from sewage effluent. Under the UWWT Directive, the River Ant has been designated as a 'sensitive area', which will require a minimum of 80% removal at Stalham STW. Data supplied on STWs discharging into the River Bure reveal that removal rates fluctuate from 0% to 99%. The effect of designation will therefore be that the removal rate will need to be stabilised so that a mean of at least 80% reduction is assured.

Given that the contribution of phosphorus from sewage effluent is, or will be, reduced as much as costs and technology allow, it is necessary to address the problem of agricultural sources. This may be achieved through a lessening in the intensity of agriculture throughout the catchment. A reduction of 20% both in stocking levels and in inorganic fertiliser applications, and the establishment of 20m riparian buffer zones, would, according to the model, produce a mean annual TP concentration from external sources of 116 to 160μg/l, and an inflow TP concentration within the range of 141 to 331μg/l, which would leave Barton Broad in the hypertrophic category, but at much reduced levels. Considering the model seems to overpredict for this lake, the inflow levels would probably be substantially lower, placing Barton Broad in the eutrophic category.

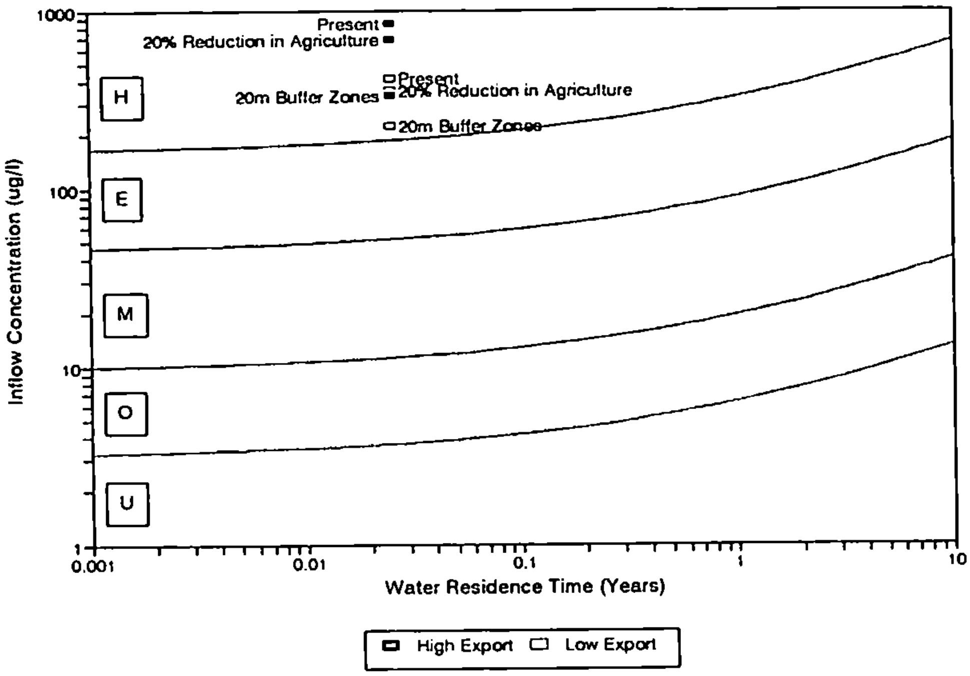

Fig.7.11: Effect of Reductions in External Phosphorus Loadings on Barton Broad

Given the lack of evidence that Barton Broad is returning to its former clear water state, despite substantial reductions in external loadings from sewage effluent, it is unlikely that recommendations made above, with respect to agricultural sources, will elicit a change in trophic status. Nevertheless, efforts to establish means of reducing sediment phosphorus loss may be obstructed in the long-term if agricultural sources are not reduced.

7.1.6 Grafham Water

The results reported below for land use and livestock relate to the immediate catchment of Grafham Water. Data on phosphorus inputs from water abstracted from the Great River Ouse will be discussed separately.

7.1.6.1 Results

A description of this site may be found in Section 6.4.1. Land use results for the catchment of Grafham Water (Fig.7.12), based on ITE data, are shown in Table 7.21. They indicate that two thirds of the catchment is under arable cultivation, with the remainder devoted mainly to permanent grassland (17%).

Table 7.21: Land Use in the Catchment of Grafham Water

Land Use	Area (ha)	Percentage
Temporary Grassland	68.37	7.87
Permanent Grassland	149.90	17.25
Arable	570.65	65.66
Rough Grazing	1.75	0.20
Settlements and Roads	34.19	3.93
Woodland	41.37	4.76
Rivers	2.89	0.33
Total	869.13	100.00

Data on livestock numbers for the two parishes adjacent to Grafham Water, which occupy about two thirds of the catchment, were unavailable, since they contain fewer than three holdings. Total livestock numbers from four other parishes were therefore used, but these required reduction by 73% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. The results shown in Table 7.22 are therefore likely to be distorted by the lack of appropriate data. In addition, the existence of a poultry unit with 350,000 birds was noted, but whether or not it lies within the catchment could not be ascertained.

Table 7.22: Livestock Levels in the Catchment of Grafham Water

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	161	1	207	95468	0

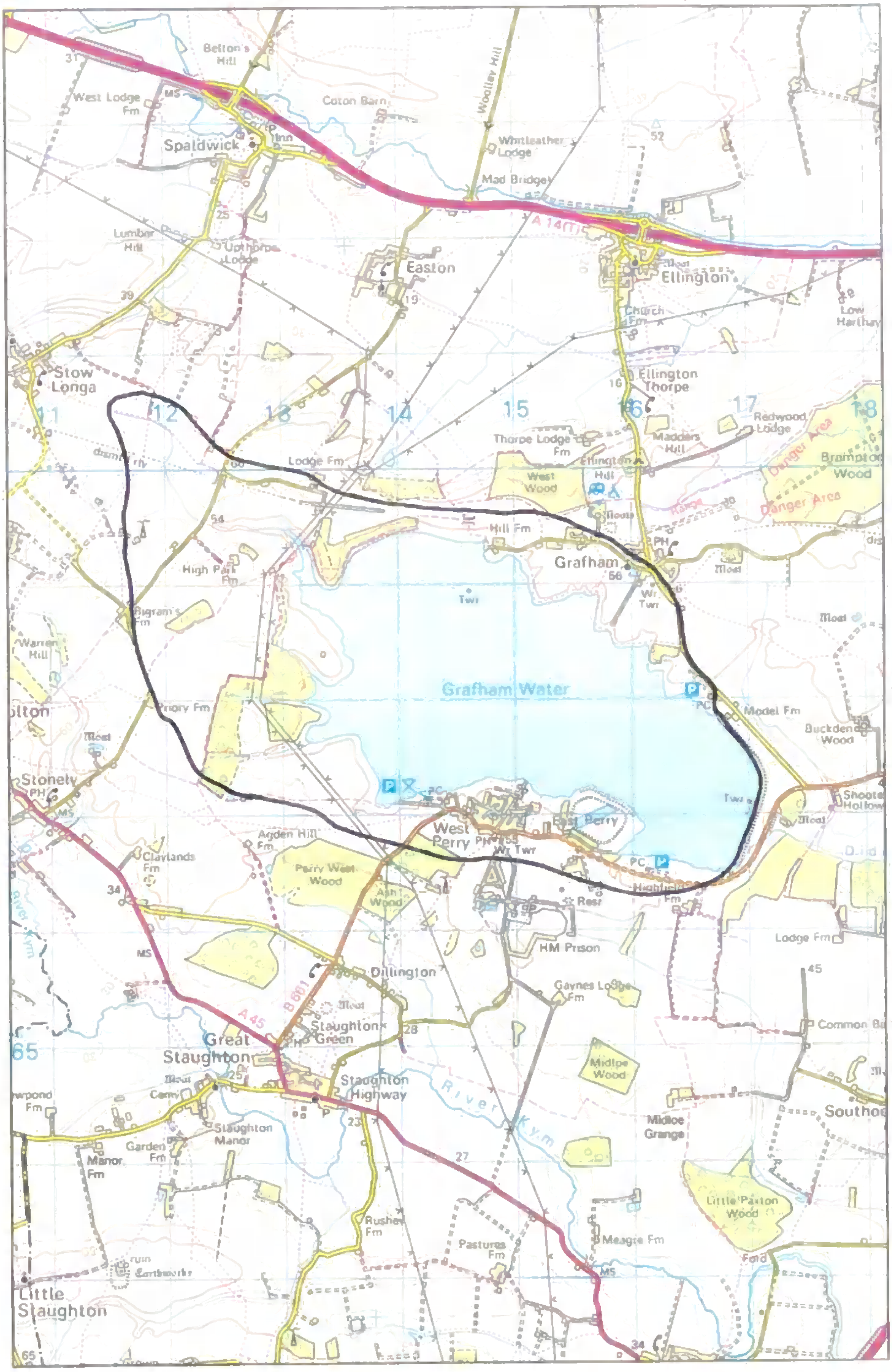

Fig.7.12: The Catchment of Grafham Water

Phosphorus loading results for Grafham Water are shown in Table 7.23. Total crude load is predicted to be between 0.52 and 1.16t/a, which, on an areal basis, converts to 0.08 and 0.18g/m^2/a. These results yield a potential mean annual TP concentration from external sources of between 3 and 7μg/l, and an inflow TP concentration within the range of 5 to 12μg/l, placing Grafham Water in the ultra-oligotrophic to oligotrophic category without pumped water inputs (Fig.7.13).

Table 7.23: Predicted External Phosphorus Loadings on Grafham Water

Parameters	Low Export	High Export
Total Load (t/a)	0.52	1.16
Areal Load (g/m^2/a)	0.08	0.18
Lake Concentration (μg/l)	3	7
Inflow Concentration (μg/l)	5	12

As might be expected in an area of arable production, between 44% and 53% of external loading is attributable to inorganic fertiliser (Table 7.24). Organic sources account for less than direct precipitation, but this is more a reflection of the extent of the reservoir.

Table 7.24: Export of Phosphorus to Grafham Water According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.27	52.99	0.51	44.12
Organic Fertiliser	0.10	19.11	0.30	25.71
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.02	3.30	0.03	2.51
Woodland	0.00	0.08	0.00	0.18
Direct Precipitation	0.13	24.51	0.32	27.48

Data on pumped water inputs to Grafham Water indicate that a further 54.60t TP was received in 1993. This loading produces a mean annual TP concentration of 348μg/l, and an inflow TP concentration of 757μg/l, placing the reservoir in the hypertrophic category. Phosphorus in pumped inputs is derived largely from sewage effluent, but diffuse agricultural sources in the catchment of the River Great Ouse may contribute.

7.1.6.2 Recommendations

Under the UWWT Directive, Grafham Water will be designated as a 'sensitive area'. Within the catchment of the River Great Ouse, down to the point of abstraction, there are a total of fifteen STWs serving over 10,000 p.e., but the DoE is allowing Anglian Water to install phosphate stripping facilities at only three (Table 7.25).

Table 7.25: The Effect of the UWWT Directive on Sewage Inputs to Grafham Water

River Great Ouse Catchment	No. of STWs	Population Equivalent (p.e.)	Percentage of p.e.
STWs >10,000 p.e.	15	816,901	86.5
STWs <10,000 p.e.	125	126,985	13.5
UWWT Directive	3	491,831	52.1

The designation of Grafham Water as a 'sensitive area' will reduce the level of TP from pumped input to 31.84t. This amount translates to an inflow TP concentration of 437μg/l, as illustrated in Fig.7.13.

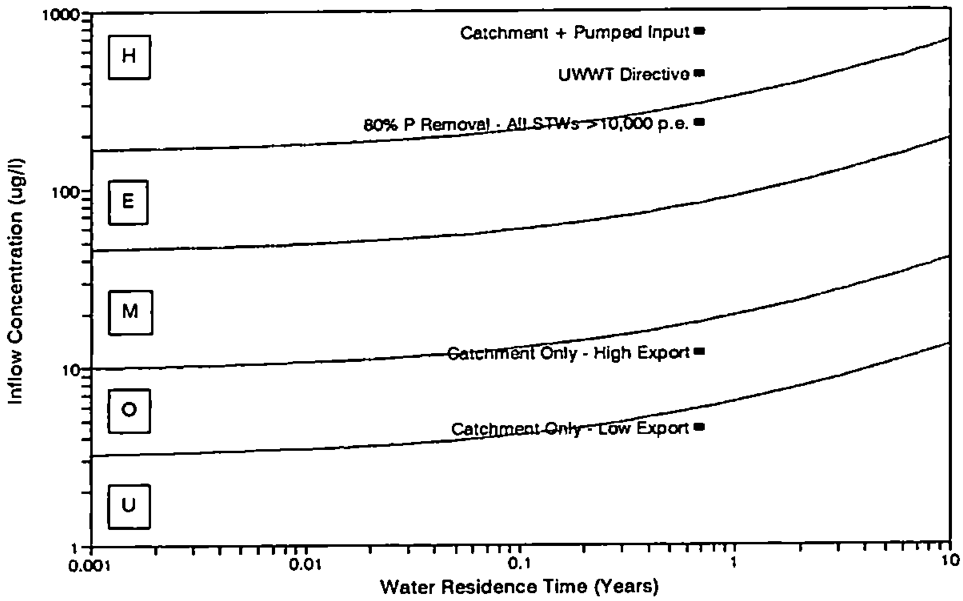

Fig.7.13: Effect of Reductions in External Phosphorus Loadings on Grafham Water

The installation of phosphate stripping facilities at all STWs serving over 10,000 p.e. would produce an inflow TP concentration of 231μg/l, placing Grafham Water in the eutrophic category. Further significant reductions may be achieved through addressing phosphorus loads from STWs serving less than 10,000 p.e., and by reducing diffuse agricultural sources of phosphorus in the catchment of the River Great Ouse.

7.1.7 Pitsford Reservoir

The results for land use and livestock relate to the immediate catchment of Pitsford Reservoir. Phosphorus inputs from water abstracted from the River Nene will be discussed separately.

7.1.7.1 Results

A description of this site may be found in Section 6.4.2. Land use results for the catchment of Pitsford Reservoir (Fig.7.14), based on ITE data, are shown in Table 7.26. They indicate that almost two thirds of the catchment is under arable cultivation, with most of the remainder devoted to grassland (28%).

Table 7.26: Land Use in the Catchment of Pitsford Reservoir

Land Use	Area (ha)	Percentage
Temporary Grassland	471.95	10.21
Permanent Grassland	833.87	18.04
Arable	2909.92	62.94
Rough Grazing	29.10	0.63
Settlements and Roads	327.37	7.08
Woodland	0.91	0.02
Rivers	50.01	1.08
Total	4623.14	100.00

Data on livestock were calculated from figures for five parishes which lie wholly within the catchment, and a further four which required reduction by 68% (Table 7.27).

Table 7.27: Livestock Levels in the Catchment of Pitsford Reservoir

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	2502	8028	2300	2031	28

Total phosphorus loading is predicted to be between 3.06 and 5.02t/a, which, on an areal basis, converts to 1.02 and 1.67g/m^2/a (Table 7.28). These results yield a potential mean annual TP concentration from external sources of between 71 and 115μg/l, and an inflow TP concentration within the range of 198 to 361μg/l, placing Pitsford Reservoir in the eutrophic to hypertrophic category even without pumped water inputs.

Table 7.28: Predicted External Phosphorus Loadings on Pitsford Reservoir

Parameters	Low Export	High Export
Total Load (t/a)	3.06	5.02
Areal Load (g/m^2/a)	1.02	1.67
Lake Concentration (μg/l)	71	115
Inflow Concentration (μg/l)	198	361

Table 7.29 provides a division of external phosphorus loading according to source. The greatest proportion (46% to 53%), using both low and high export coefficients, is derived from inorganic fertiliser, with sewage sources contributing 23% to 38%.

Table 7.29: Export of Phosphorus to Pitsford Reservoir According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	1.42	46.41	2.67	53.14
Organic Fertiliser	0.25	8.22	0.75	15.06
Sewage Effluent	1.17	38.08	1.17	23.26
Settlements and Roads	0.16	5.34	0.28	5.55
Woodland	0.00	0.00	0.00	0.00
Direct Precipitation	0.06	1.96	0.15	2.99

Data on pumped water inputs to Pitsford Reservoir show that a further 4.46t TP was received in 1993. This loading, added to that from the immediate catchment, gives a mean annual TP concentration of between 173 and 218μg/l, and an inflow TP concentration within the range of 593 to 785μg/l, placing the reservoir in the hypertrophic category. The source of phosphorus in pumped inputs is largely sewage effluent discharged into the River Nene. However, it is likely that diffuse agricultural sources in the catchment of the Nene also contribute.

7.1.7.2 Recommendations

Pitsford Reservoir has been designated as a 'sensitive area' under the UWWT Directive. There are three STWs located in the immediate catchment, but none of these serve the necessary 10,000 p.e. Within the catchment of the River Nene, down to the point of abstraction, there are a total of fourteen STWs, one of which serves over 10,000 p.e. and will be subject to the requirements of the Directive. The total p.e. for the remaining STWs amounts to slightly less than that for the single works (Table 7.30).

Table 7.30: The Effect of the UWWT Directive on Sewage Inputs to Pitsford Reservoir

Catchment	No. of STWs	Population Equivalent (p.e.)	Percentage of p.e.
Pitsford Reservoir	3	1,938	4.5
River Nene >10,000 p.e.	1	21,167	49.0
River Nene <10,000 p.e.	13	20,192	46.5
UWWT Directive	1	21,167	49.0

The designation of Pitsford Reservoir as a 'sensitive area' will reduce the level of TP from pumped input to 2.72t. However, this amount, coupled with the loading from the immediate catchment, translates to a mean annual TP concentration of between 133 and 178μg/l. The effect of this measures is illustrated in Fig.7.15. Further significant reductions may be achieved through addressing phosphorus loads from STWs serving less than 10,000 p.e., and by reducing diffuse agricultural sources of phosphorus in the catchment of the River Nene.

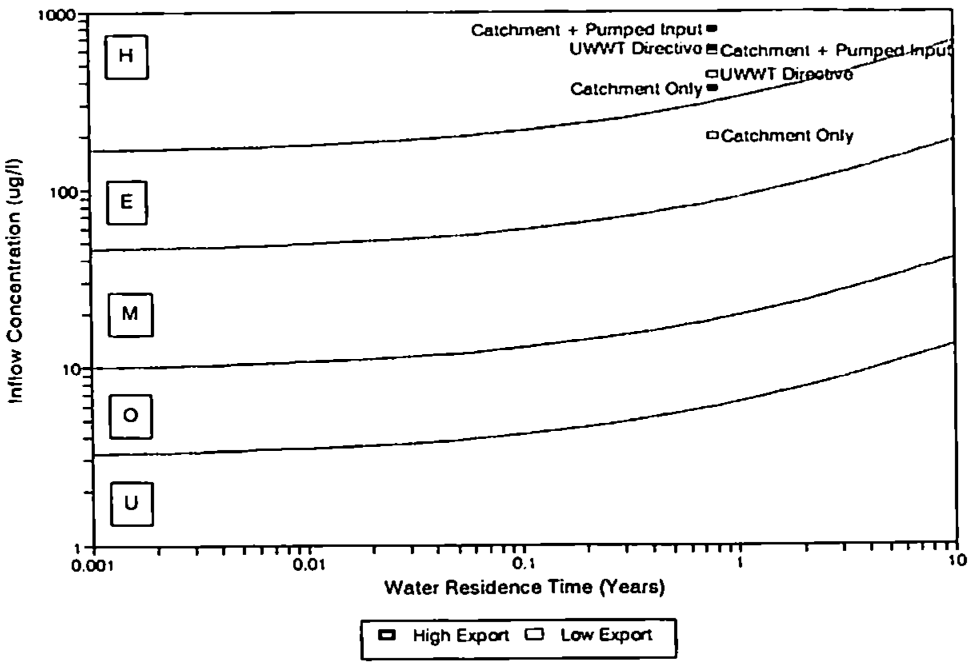

Fig.7.15: Effect of Reductions in External Phosphorus Loadings on Pitsford Reservoir

7.1.8 Rutland Water

The results reported below for land use and livestock numbers relate to the immediate catchment of Rutland Water. Data on phosphorus inputs from water abstracted from the Rivers Nene and Welland will be discussed separately.

7.1.8.1 Results

A description of this site may be found in Section 6.4.3. Land use results for the catchment of Rutland Water (Fig.7.16), based on ITE data, are shown in Table 7.31. They indicate that about half the catchment is under arable cultivation, with a further 33% devoted to grassland, either temporary or permanent. Settlements and roads occupy about 9% of the catchment, and woodland nearly 5%.

Data from parish returns on livestock numbers required reduction by 68% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. Data for five of the seven parishes which border the reservoir were unavailable, since fewer than three holdings are present in each. The results are shown in Table 7.32.

Table 7.31: Land Use in the Catchment of Rutland Water

Land Use	Area (ha)	Percentage
Temporary Grassland	639.20	10.64
Permanent Grassland	1369.46	22.79
Arable	2975.86	49.51
Rough Grazing	129.49	2.15
Settlements and Roads	549.02	9.14
Woodland	278.49	4.63
Rivers	68.96	1.15
Total	6010.08	100.00

Table 7.32: Livestock Levels in the Catchment of Rutland Water

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	2098	10436	1612	1755	13

Predicted phosphorus loadings for Rutland Water are listed in Table 7.28. Total crude load is predicted to be between 9.49 and 11.91t/a, which, on an areal basis, converts to 0.76 and 0.95g/m^2/a. Potential mean annual TP concentration from external sources is calculated to be between 65 and 81μg/l, and an inflow TP concentration within the range of 233 to 307μg/l, placing Rutland Water in the eutrophic category even without pumped water inputs.

Table 7.33: Predicted External Phosphorus Loadings on Rutland Water

Parameters	Low Export	High Export
Total Load (t/a)	9.49	11.91
Areal Load (g/m^2/a)	0.76	0.95
Lake Concentration (μg/l)	65	81
Inflow Concentration (μg/l)	233	307

Table 7.34 lists the division of external phosphorus loading according to source. The greatest proportion (61% to 76%) is derived from sewage sources, with inorganic fertilisers providing a further 16% to 24%. The contribution from animal wastes is on a par with that from direct precipitation.

Table 7.34: Export of Phosphorus to Rutland Water According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	1.50	15.84	2.87	24.13
Organic Fertiliser	0.24	2.47	0.71	5.91
Sewage Effluent	7.22	76.12	7.22	60.65
Settlements and Roads	0.28	2.89	0.47	3.92
Woodland	0.00	0.03	0.01	0.12
Direct Precipitation	0.25	2.64	0.63	5.27

Data on pumped water inputs to Rutland Water indicate that a further 37.2t TP was received in 1993. This loading, added to that from the immediate catchment, gives a mean annual TP concentration of between 319 and 335μg/l, and an inflow TP concentration within the range of 1622 to 1725μg/l, placing the reservoir in the hypertrophic category. The source of phosphorus in pumped inputs is largely sewage effluent discharged into the Rivers Nene and Welland. However, it is likely that diffuse agricultural sources in the catchments of these rivers also contribute.

7.1.8.2 Recommendations

Rutland Water and the River Nene will be designated as 'sensitive areas' under the UWWT Directive. Three STWs are located in the immediate catchment of Rutland Water, one of which serves the necessary 10,000 p.e. to qualify for phosphate removal. Within the catchments of the Rivers Nene and Welland, down to the point of abstraction, there are a further 109 STWs. Seven serve over 10,000 p.e., five of which will be subject to the requirements of the Directive, but these represent about 84% of the total p.e. (Table 7.35).

Table 7.35: The Effect of the UWWT Directive on Sewage Inputs to Rutland Water

Catchment	STWs	Population Equivalent (p.e.)	Percentage of p.e.
Rutland Water >10,000 p.e.	1	11,999	1.5
Rutland Water <10,000 p.e.	2	653	0.1
Nene and Welland >10,000 p.e.	7	691,391	88.0
Nene and Welland <10,000 p.e.	102	81,987	10.4
UWWT Directive	5	657,106	83.6

The designation of Rutland Water and the River Nene as 'sensitive areas' will reduce the level of phosphorus from pumped input to 11.91t TP, based on data for 1993. However, this amount, coupled with the loading from the immediate catchment, still converts to a mean annual TP concentration of between 111 and 127μg/l, and an inflow TP concentration within the range of 448 to 530μg/l, leaving Rutland Water in the hypertrophic category, as illustrated in Fig.7.17.

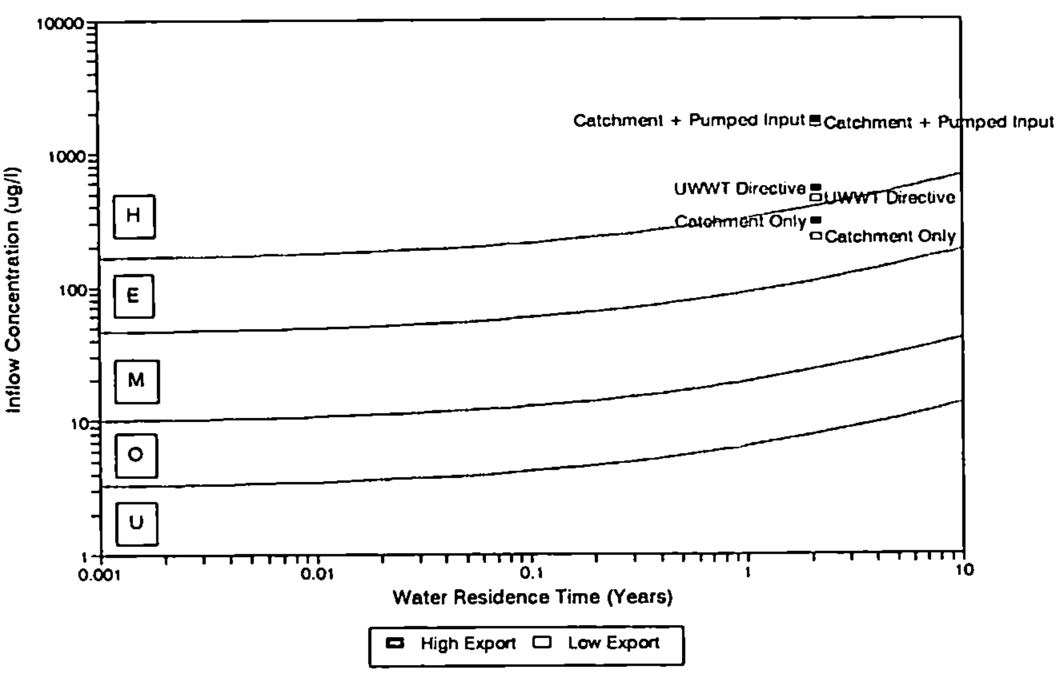

Fig.7.17: Effect of Reductions in External Phosphorus Loadings on Rutland Water

As in the case of Grafham Water and Pitsford Reservoir, further significant reductions may be achieved through addressing phosphorus loads from STWs serving less than 10,000 p.e., and by reducing diffuse agricultural sources of phosphorus in the catchments of the Rivers Nene and Welland.

7.1.9 Sutton Park Pools

As discussed in Section 5.1, a full evaluation of Sutton Park Pools cannot be carried out, since a series of lakes is involved. However, sufficient information is available to indicate the major sources of nutrient inputs to the lakes.

7.1.9.1 Results

A description of this site may be found in Section 6.5.1. Land use results for the catchment of Sutton Park Pools (Fig.7.18), calculated from ITE data, are shown in Table 7.36. The catchment covers nearly 1800 hectares, with settlements and roads accounting for 42%. Permanent pasture provides a further 29%, and woodland 22%.

Table 7.36: Land Use in the Catchment of Sutton Park Pools

Land Use	Area (ha)	Percentage
Temporary Grassland	9.98	0.56
Permanent Grassland	512.01	28.50
Arable	6.99	0.39
Rough Grazing	119.77	6.67
Settlements and Roads	748.56	41.67
Woodland	394.24	21.94
Rivers	4.99	0.28
Total	1796.54	100.00

Information on the types of holdings reveal that over two thirds are operated on a part-time basis. Livestock numbers for the catchment (Table 7.37) are appropriately low in this urbanised catchment, but some cattle and sheep are kept within the Park.

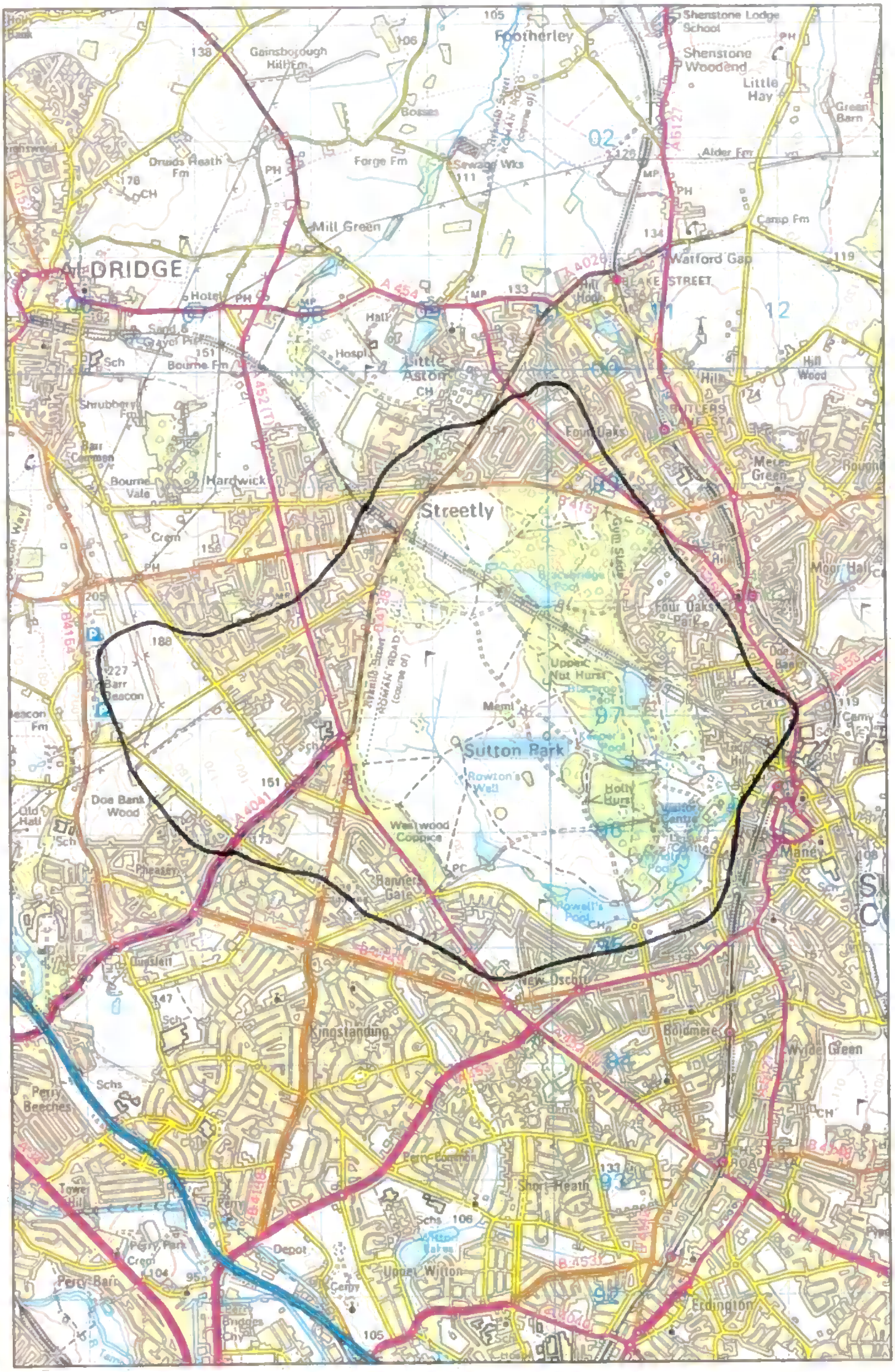

Fig.7.18: The Catchment of Sutton Park

Table 7.37: Livestock Levels in the Catchment of Sutton Park Pools

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	39	55	8	145	4

Phosphorus loading results for Sutton Park Pools are detailed in Table 7.38. Total crude load is predicted to be between 0.42 and 0.77t/a, which, on an areal basis, converts to 1.49 and 2.73g/m^2/a.

Table 7.38: Predicted External Phosphorus Loadings on Sutton Park Pools

Parameters	Low Export	High Export
Total Load (t/a)	0.42	0.77
Areal Load (g/m^2/a)	1.49	2.73
Lake Concentration (μg/l)	Not Calculable	Not Calculable
Inflow Concentration (μg/l)	Not Calculable	Not Calculable

Table 7.39 provides a breakdown of external phosphorus loadings according to source. By far the greatest contribution (83% to 90%) is derived from urban runoff. The only other source of any significance relates to inorganic fertilisers which may supply between 7% and 11%.

Table 7.39: Export of Phosphorus to Sutton Park Pools According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.03	7.25	0.09	11.35
Organic Fertiliser	0.00	0.70	0.01	1.17
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.37	89.76	0.64	83.10
Woodland	0.00	0.95	0.02	2.57
Direct Precipitation	0.01	1.34	0.01	1.92

7.1.9.2 Recommendations

Surface water from surrounding urban land enters the Park *via* four sewers, for which consents are granted. Phosphorus in these discharges probably correlates to the estimated 0.4 to 0.6t TP contributed by settlements and roads. However, the methodology adopted does not lend itself to accounting for intermittent discharges in the form of combined sewer overflows (CSOs). There are consents for four CSOs into Sutton Park, which must affect not only the nutrient status of streams and lakes, but also levels of BOD. All consented discharges into Sutton Park need to be reviewed, and their effects upon the Park assessed fully. Provision of increased storm capacity would reduce the frequency of overflows, but it is debatable whether such discharges should occur at all.

7.1.10 Aqualate Mere

7.1.10.1 Results

A description of this site may be found in Section 6.6.1. Land use results for the catchment of Aqualate Mere (Fig.7.19), calculated from ITE data, are shown in Table 7.40. Approximately 90% of the catchment, which covers over 5000 hectares, is devoted to agriculture, with arable land and permanent pasture accounting for the majority. Temporary pasture occupies about 14% of the catchment.

Table 7.40: Land Use in the Catchment of Aqualate Mere

Land Use	Area (ha)	Percentage
Temporary Grassland	683.61	13.59
Permanent Grassland	1638.95	32.59
Arable	1915.44	38.09
Rough Grazing	290.80	5.78
Settlements and Roads	382.33	7.60
Woodland	82.95	1.65
Rivers	34.32	0.68
Total	5028.40	100.00

Information on the types of holdings in each parish indicate that 27% of farms are run on a part-time basis. Of the remainder, 53% are either specialist or mainly dairy farms, and 13% are involved in the cultivation of general crops. Six specialist poultry, or pig and poultry units are also located in the parishes, but it was not ascertained whether or not these lay within the catchment. Total livestock numbers required reduction by 63%, so that the results, shown in Table 7.41, are not as reliable as they might be.

Table 7.41: Livestock Levels in the Catchment of Aqualate Mere

Livestock	**Cattle**	**Sheep**	**Pigs**	**Poultry**	**Other Fowl**
Adjusted Total	5547	5720	8887	72786	129

Phosphorus loading results for Aqualate Mere are listed in Table 7.42. Total crude load is predicted to lie between 2.24 and 4.63t/a, which, on an areal basis, converts to 3.09 and 6.38g/m^2/a. These figures yield a mean annual TP concentration from external sources of between 89 and 184μg/l, and an inflow TP concentration within the range of 160 and 388μg/l, placing Aqualate Mere in the eutrophic to hypertrophic category. There are no measured data for direct comparison with these results, but one of the main inflows is reported to contain in excess of 100μg/l TP on a regular basis.

Table 7.42: Predicted External Phosphorus Loadings on Aqualate Mere

Parameters	**Low Export**	**High Export**
Total Load (t/a)	2.24	4.63
Areal Load (g/m^2/a)	3.09	6.38
Lake Concentration (μg/l)	89	184
Inflow Concentration (μg/l)	160	388

The phosphorus loading on Aqualate Mere is divided according to source in Table 7.43. The majority (45% to 47%) is derived from inorganic fertiliser, whether low or high export coefficients are applied, and animal waste supplies a further 27% to 39%. Between 8% and 17% may be attributed to sewage sources, while urban runoff generates

7% to 9%. Woodland and direct precipitation do not appear to contribute significant amounts to the loading.

Table 7.43: Export of Phosphorus to Aqualate Mere According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	1.05	46.77	2.09	45.09
Organic Fertiliser	0.60	26.61	1.79	38.91
Sewage Effluent	0.39	17.38	0.39	8.41
Settlements and Roads	0.19	8.54	0.32	7.02
Woodland	0.00	0.04	0.00	0.09
Direct Precipitation	0.01	0.65	0.04	0.78

7.1.10.2 Recommendations

Diffuse agricultural sources appear to be the primary target for phosphorus control strategies in this catchment. The contribution of sewage, while not insignificant in total, is, on the basis of individual STWs, less important. One small STW (168 p.e.) is located about two and a half miles upstream to the north of Aqualate Mere, while the other (478 p.e.) lies about five miles away in the extreme south of the catchment. It is therefore unlikely, considering the distances involved, that sewage sources influence the lake to a significant degree.

Diffuse sources of phosphorus could be reduced by a lessening in the intensity of agriculture throughout the catchment. A reduction of 20% both in stocking density and in inorganic fertiliser applications, and the establishment of 20m buffer zones along watercourses, would theoretically produce a mean annual TP concentration from external sources within the range of 48 to 88μg/l, and an inflow TP concentration between 76 to 158μg/l, placing Aqualate Mere in the eutrophic category, as illustrated in Fig.7.20. The use of buffer zones is likely to reduce also the level of silting to which the Mere is subject.

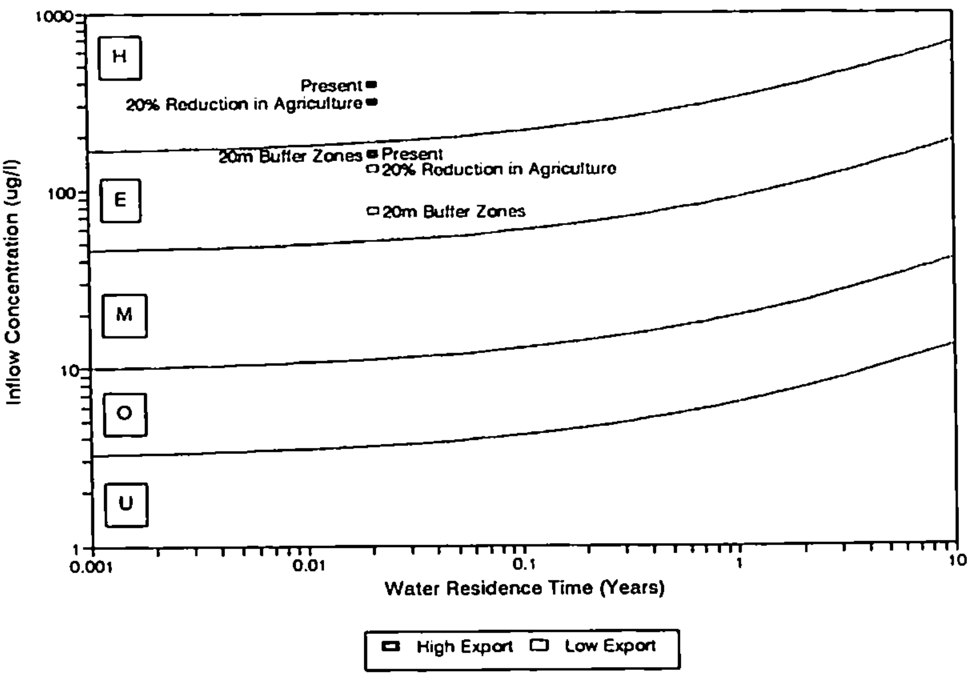

Fig.7.20: Effect of Reductions in External Phosphorus Loading on Aqualate Mere

7.1.11 Chapel Mere

7.1.11.1 Results

A description of this site may be found in Section 6.6.2. Land use results for the catchment of Chapel Mere (Fig.7.21), calculated from ITE data, are shown in Table 7.44. The catchment extends over 256 hectares, of which 90% is devoted to agricultural land. Permanent pasture provides 41%, arable land 29%, and temporary grassland a further 13%. The remaining 10% of the catchment consists of settlements and roads. Information on the type of holdings in the relevant parishes indicates that 30% of farms operate on a part-time basis. Of the remainder, nearly 80% are specialist or mainly dairy, with the rest involved in cattle rearing, pigs and poultry, fruit growing, or mixed farming. The location of a pig and poultry unit, which involves 15,000 birds, was not established.

Total livestock numbers from parish returns required reduction by 87% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. The results, shown in Table 7.44, are therefore of doubtful reliability.

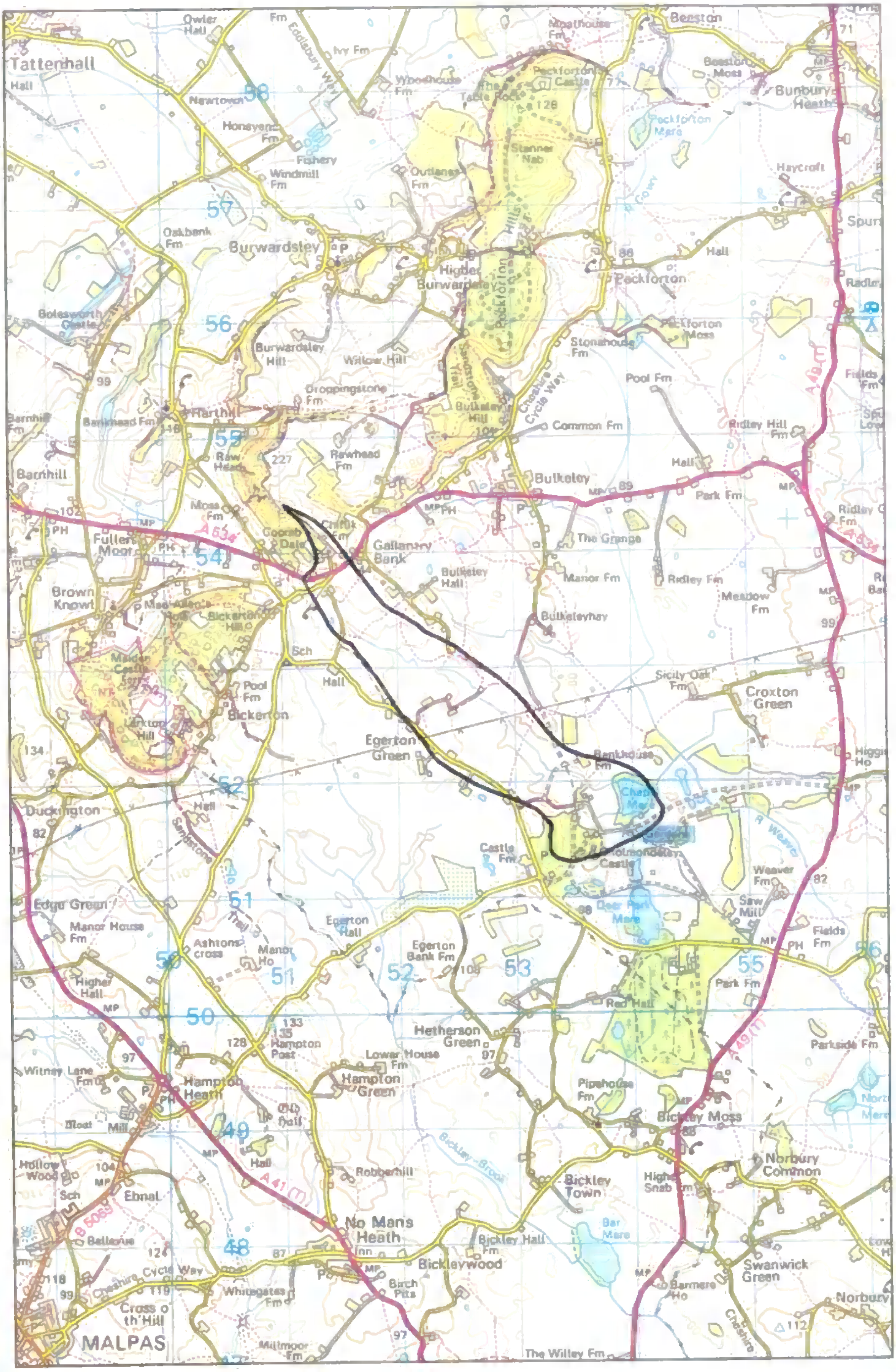

Fig.7.21: The Catchment of Chapel Mere

Table 7.44: Land Use in the Catchment of Chapel Mere

Land Use	Area (ha)	Percentage
Temporary Grassland	34.52	13.49
Permanent Grassland	105.33	41.18
Arable	73.47	28.72
Rough Grazing	15.93	6.23
Settlements and Roads	24.78	9.69
Woodland	0.00	0.00
Rivers	1.77	0.69
Total	255.81	100.00

Table 7.45: Livestock Levels in the Catchment of Chapel Mere

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	388	16	4	2039	0

Phosphorus loading results for Chapel Mere are detailed in Table 7.46. Total crude load is predicted to be between 0.08 and 0.19t/a, which, on an areal basis, converts to 1.25 and 2.87g/m^2/a. These results produce a mean annual TP concentration from external sources of between 122 and 281μg/l, and an inflow TP concentration within the range of 286 and 789μg/l, placing Chapel Mere in the hypertrophic category. A mean inflow TP concentration of 727μg/l for 1991/2, calculated from measured data in Moss *et al.* [1992], correlates well with the figure produced by the use of high export coefficients.

Table 7.46: Predicted External Phosphorus Loadings on Chapel Mere

Parameters	Low Export	High Export
Total Load (t/a)	0.08	0.19
Areal Load (g/m^2/a)	1.25	2.87
Lake Concentration (μg/l)	122	281
Inflow Concentration (μg/l)	286	789

Table 7.46 provides an analysis of external phosphorus loading according to source. Inorganic fertilisers account for the greatest proportion (48% to 54%), under both low and high export levels. Animal waste contributes a further 30% to 39%, while settlements and roads present between 11% and 15%.

Table 7.47: Export of Phosphorus to Chapel Mere According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.04	53.50	0.09	48.24
Organic Fertiliser	0.02	29.67	0.07	38.75
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.01	15.24	0.02	11.28
Woodland	0.00	0.00	0.00	0.00
Direct Precipitation	0.00	1.60	0.00	1.74

7.1.11.2 Recommendations

Since there are no consented discharges within the catchment of Chapel Mere, the sewage and farm effluents which are thought to contribute nutrients may be seen as contraventions of the Water Resources Act 1991, and should be dealt with on that level. It is within the power of the NRA to investigate and sample such effluents, and to take action against those responsible. Leakages from storage of animal waste, in particular, may be addressed through the NRA's farm campaign.

In order to lessen contributions from diffuse sources of phosphorus it may be necessary to lower the intensity of agriculture throughout the catchment. This may be achieved through a reduction of 20%, both in stocking levels and in inorganic fertiliser application, and the establishment of 20m riparian buffer zones. For Chapel Mere, such measures produce a mean annual TP concentration from external sources ranging from 59 to 128μg/l, and an inflow TP concentration between 117 and 304μg/l, moving Chapel Mere towards the eutrophic category, as illustrated in Fig.7.22.

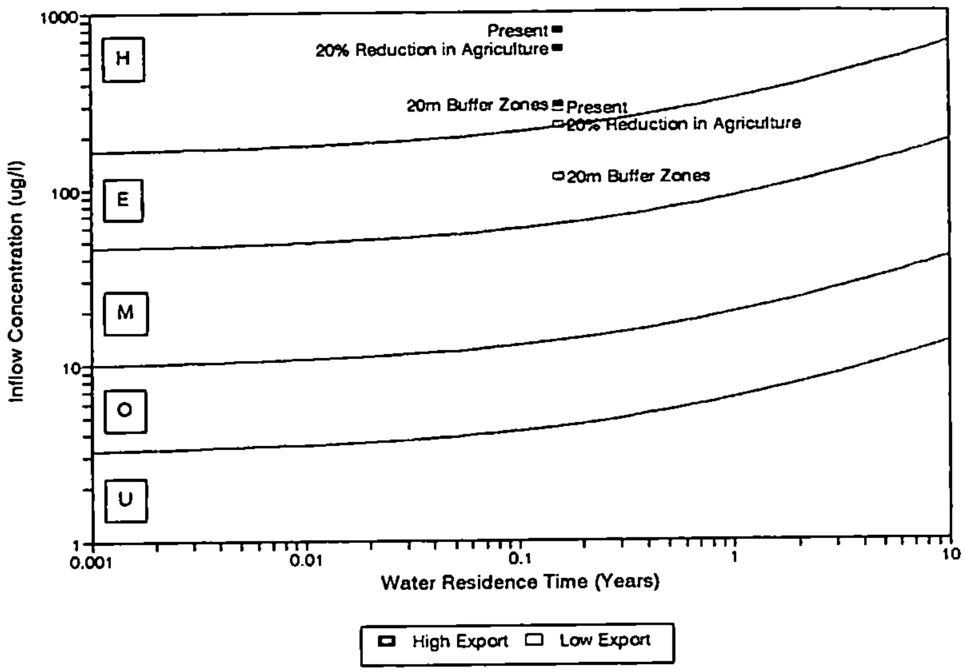

Fig.7.22: Effect of Reductions in External Phosphorus Loadings on Chapel Mere

7.1.12 Comber Mere

7.1.12.1 Results

A description of this site may be found in Section 6.6.3. Land use results for the catchment of Comber Mere (Fig.7.23), calculated from ITE data, are shown in Table 7.48. Over 90% of the catchment is devoted to agriculture. Arable land forms the greater part at 40%, and permanent pasture constitutes a further 32%. Information in the parish returns on the types of holdings indicates that nearly half the farms are operated on a part-time basis. Of the remainder, 80% are specialist or mainly dairy farms. Two specialist poultry and four pig and poultry units exist. Four of these are not situated within the catchment, but the location of the other two is unknown.

The catchment of Comber Mere straddles the border between Shropshire and Cheshire, at a point where the boundaries of four large parishes meet. Consequently, the data for total livestock numbers required reduction by 91% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. The results are shown in Table 7.49, and confirm the importance of cattle in this catchment.

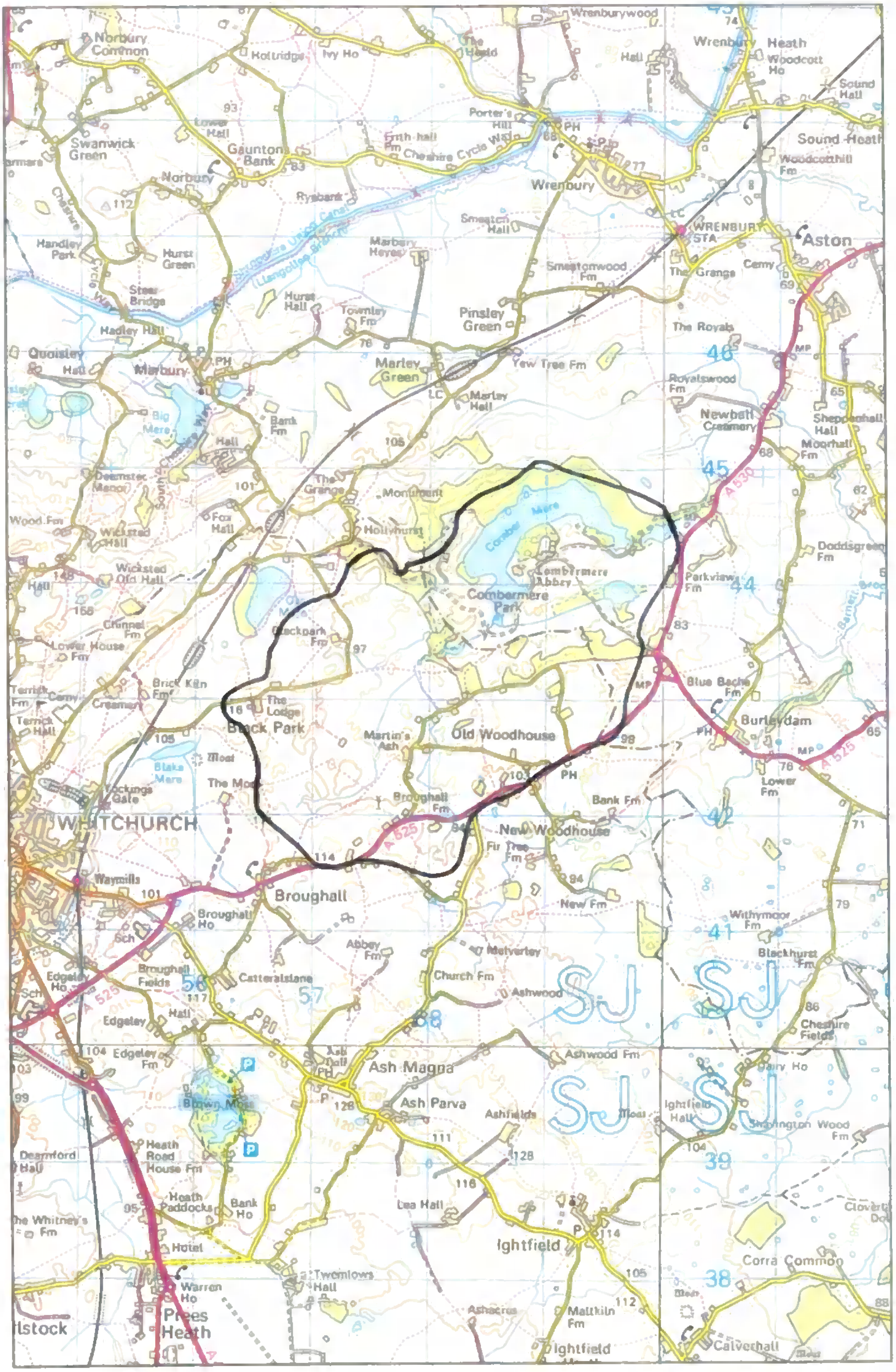

Fig.7.23: The Catchment of Comber Mere

Table 7.48: Land Use in the Catchment of Comber Mere

Land Use	Area (ha)	Percentage
Temporary Grassland	110.52	13.87
Permanent Grassland	258.20	32.40
Arable	313.92	39.39
Rough Grazing	42.72	5.36
Settlements and Roads	19.50	2.45
Woodland	47.37	5.94
Rivers	4.64	0.58
Total	796.88	100.00

Table 7.49: Livestock Levels in the Catchment of Comber Mere

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	1403	422	538	3529	9

Total crude load for Comber Mere is predicted to be between 0.29t/a and 0.67t/a, which, on an areal basis, converts to 0.56 and 1.31g/m^2/a (Table 7.50). Mean annual TP concentration from external sources is calculated to lie between 69 and 161μg/l, and inflow TP concentration to be between 233 and 659μg/l, placing Comber Mere in the eutrophic to hypertrophic category. A mean inflow TP concentration of 1060μg/l for 1991/2, calculated from measured data in Moss *et al.* [1992], suggests that the use of high export coefficients is more appropriate in this catchment.

Table 7.50: Predicted External Phosphorus Loadings on Comber Mere

Parameters	Low Export	High Export
Total Load (t/a)	0.29	0.67
Areal Load (g/m^2/a)	0.56	1.31
Lake Concentration (μg/l)	69	161
Inflow Concentration (μg/l)	233	659

Table 7.51 details the external phosphorus loading according to source. Inorganic fertiliser contributes the largest proportion (50% to 60%), under both low and high export coefficients. Animal waste accounts for between 34% and 43%, but other sources are not of significance.

Table 7.51: Export of Phosphorus to Comber Mere According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.17	59.30	0.34	50.37
Organic Fertiliser	0.10	33.57	0.29	43.00
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.01	3.39	0.02	2.46
Woodland	0.00	0.16	0.00	0.35
Direct Precipitation	0.01	3.58	0.02	3.82

7.1.12.2 Recommendations

In addition to the loadings detailed above, Comber Mere is thought to be receiving inputs from septic tank discharge, for which a consent is granted, and silage effluent from a local farm. Both of these sources require the attention of the NRA, since it is within its power to investigate and to take appropriate action.

Loading results indicate that diffuse agricultural sources are the primary targets for phosphorus control strategies in this catchment. A lessening in the intensity of agriculture throughout the catchment could be achieved through a reduction of 20% both in stocking density and in organic fertiliser application, and the establishment of 20m riparian buffer zones. These measures would produce a mean annual TP concentration within the range of 29 to 55μg/l, and an inflow TP concentration of between 81 and 178μg/l. Since the use of high export coefficients appears to predict measured inflow concentration better, the higher figure of these two figures is more likely. This would place Comber Mere in the eutrophic, rather than mesotrophic, category.

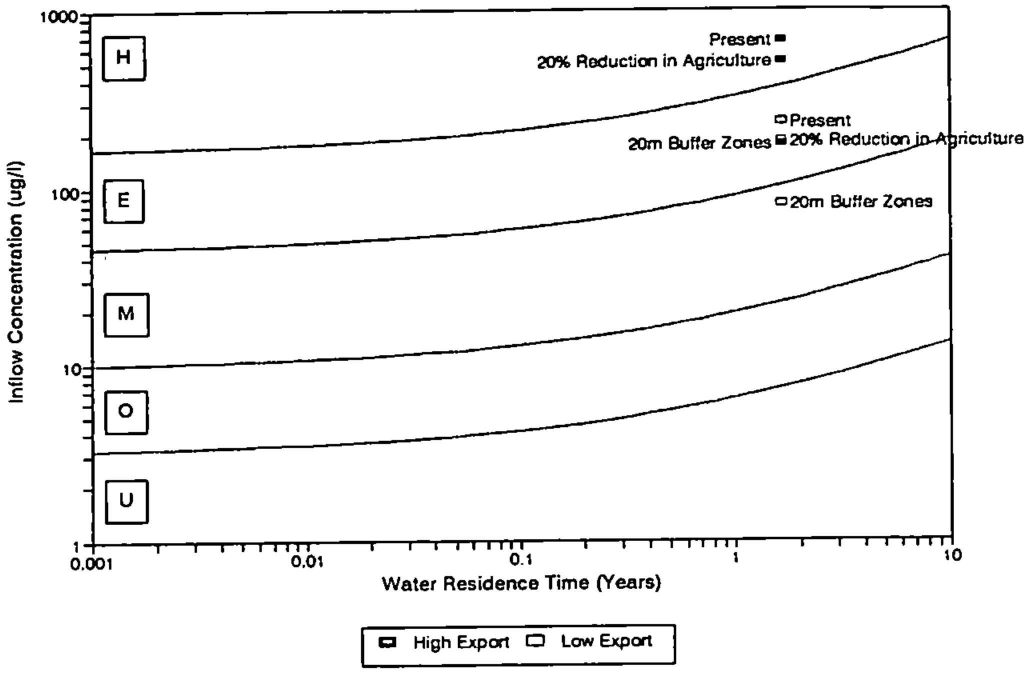

Fig.7.24: Effect of Reductions in External Phosphorus Loadings on Comber Mere

7.1.13 Norbury Meres

As explained in Section 5.2.5, lack of data on water residence times for Norbury Meres means that a full evaluation cannot be carried. However, sufficient information is available to indicate the sources of nutrient inputs.

7.1.13.1 Results

A description of this site may be found in Section 6.6.4. Land use results for the catchment of Norbury Meres (Fig.7.25), calculated from ITE data, are shown in Table 7.52. Over 97% of the catchment is devoted to agricultural, with 43% in arable cultivation, and 54% under grass.

The catchment is situated in an area where four parishes meet. As a consequence, data for total livestock numbers required reduction by 95% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. The results, shown in Table 7.53, are therefore less than reliable. Four pig and poultry units exist in the parishes, but whether they are located in the catchment is unknown.

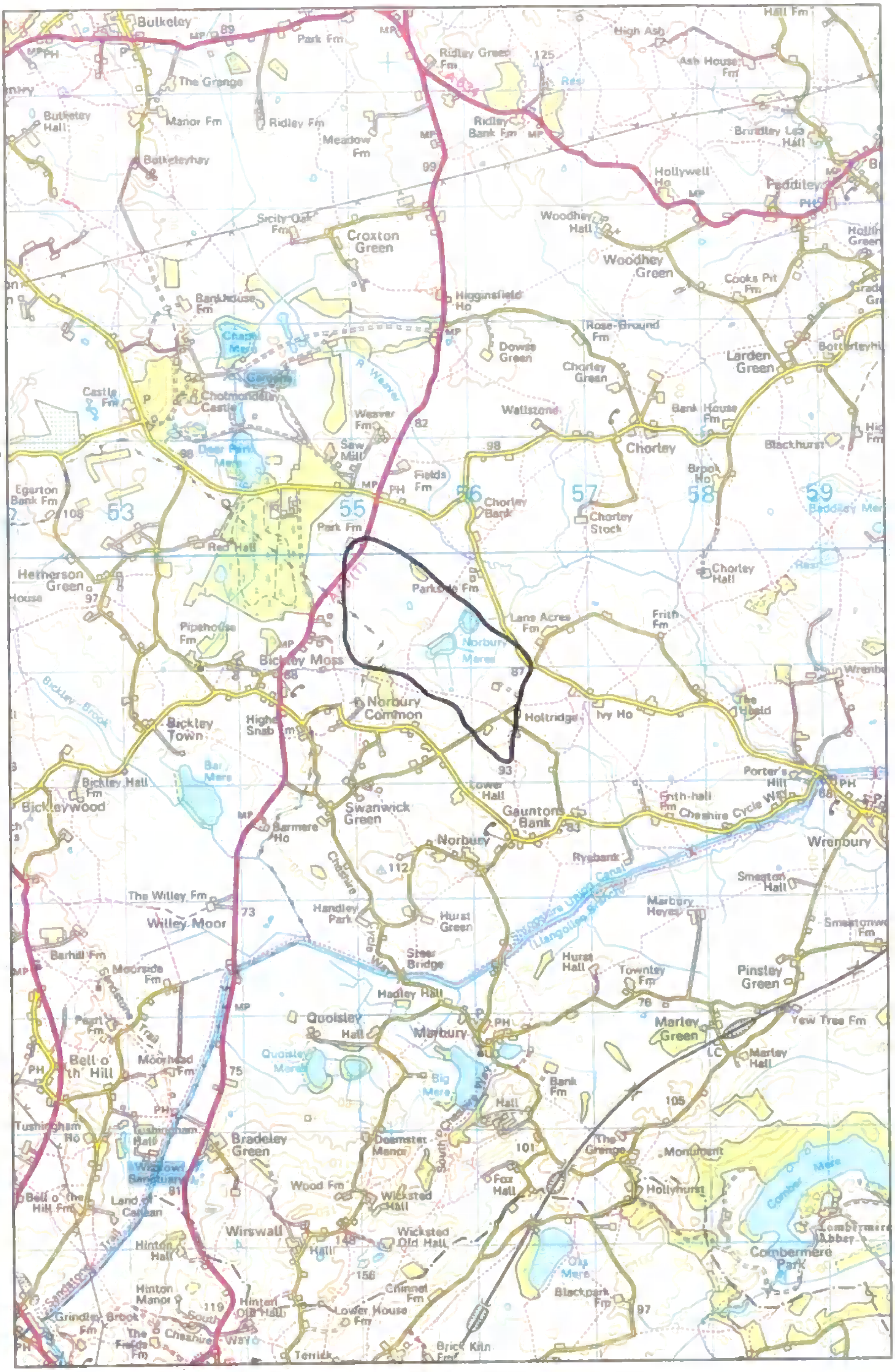

Fig.7.25: The Catchment of Norbury Meres

Table 7.52: Land Use in the Catchment of Norbury Meres

Land Use	Area (ha)	Percentage
Temporary Grassland	24.26	14.90
Permanent Grassland	54.79	33.65
Arable	70.44	43.27
Rough Grazing	8.61	5.29
Settlements and Roads	3.91	2.40
Woodland	0.00	0.00
Rivers	0.78	0.48
Total	162.79	100.00

Table 7.53: Livestock Levels in the Catchment of Norbury Meres

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	318	28	278	1549	5

Phosphorus loadings for the Norbury Meres are listed in Table 7.54. Total crude load is predicted to be between 0.07 and 0.16t/a, which, on an areal basis, converts to 2.12 and 5.01g/m^2/a. An SRP value of 1632μg/l has been recorded.

Table 7.54: Predicted External Phosphorus Loadings on Norbury Meres

Parameters	Low Export	High Export
Total Load (t/a)	0.07	0.16
Areal Load (g/m^2/a)	2.12	5.01
Lake Concentration (μg/l)	Not Calculable	Not Calculable
Inflow Concentration (μg/l)	Not Calculable	Not Calculable

Table 7.55 details the external phosphorus loading according to source. Altogether, diffuse agricultural sources account for 96% to 97% of phosphorus export, while urban runoff contributes only 2% to 3%.

Table 7.55: Export of Phosphorus to Norbury Meres According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.04	57.99	0.08	48.58
Organic Fertiliser	0.03	38.08	0.08	48.28
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.00	2.98	0.00	2.14
Woodland	0.00	0.00	0.00	0.00
Direct Precipitation	0.00	0.94	0.00	1.00

7.1.13.2 Recommendations

As in the case of Chapel Mere and Comber Mere, Norbury Meres appear to be suffering from enrichment as a result of agricultural point sources, for which consents do not exist. Although improvements to farm drains have reportedly been made, there is still cause for the NRA to investigate and to take appropriate action.

As specified above, diffuse agricultural sources contribute by far the greatest proportion of phosphorus loading. This level could be reduced by a lessening in the intensity of agriculture throughout the catchment. However, Norbury Meres may also benefit from the removal of drainage ditches for some distance around their perimeter. This would allow fringing vegetation to act as a nutrient buffer, and may also prevent the direct entry of farm effluent into the Meres.

7.1.14 Petty Pool

7.1.14.1 Results

A description of this site may be found in Section 6.6.5. Land use results for the catchment of Petty Pool (Fig.7.26), calculated from ITE data, are shown in Table 7.56. Nearly 70% consists of agricultural land, and, compared to most catchments in this study, a high proportion of land is occupied by woodland (20%).

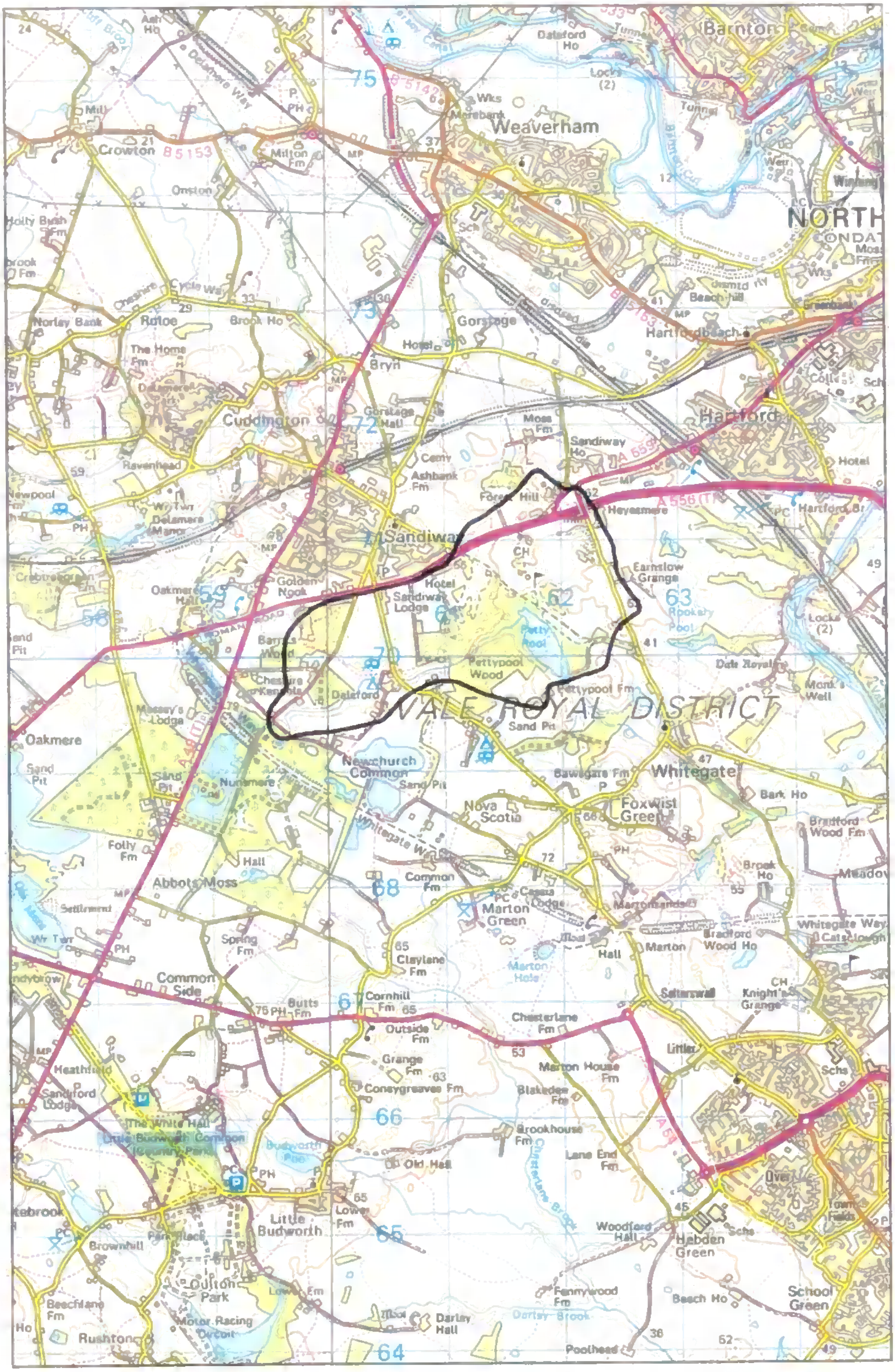

Fig.7.26: The Catchment of Petty Pool

Table 7.56: Land Use in the Catchment of Petty Pool

Land Use	Area (ha)	Percentage
Temporary Grassland	58.87	14.89
Permanent Grassland	106.94	27.05
Arable	86.34	21.84
Rough Grazing	22.57	5.71
Settlements and Roads	41.21	10.42
Woodland	79.47	20.10
Rivers	0.00	0.00
Total	395.39	100.00

Information contained in the parish returns indicates that over half of farms are run on a part-time basis. Of the remainder, 65% are specialist or mainly dairy farms, and a further 24% cultivate general crops. Data on livestock numbers required reduction by 66% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. The results, shown in Table 7.57, are therefore less than reliable.

Table 7.57: Livestock Levels in the Catchment of Petty Pool

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	461	498	147	121	25

Phosphorus loading results for Petty Pool are detailed in Table 7.58. Total crude load is predicted to be between 0.11 and 0.26t/a, which, on an areal basis, converts to 0.94 and 2.19g/m^2/a. Mean annual TP concentrations from external sources are calculated to lie between 70 and 162μg/l, and inflow TP concentration to be between 145 and 407μg/l, placing Petty Pool in the eutrophic to hypertrophic category. A mean inflow TP concentration of 999μg/l for 1991/2, calculated from measured data in Moss *et al.* [1992], suggests that the use of high export coefficients is more appropriate in this catchment.

Table 7.58: Predicted External Phosphorus Loadings on Petty Pool

Parameters	Low Export	High Export
Total Load (t/a)	0.11	0.26
Areal Load ($g/m^2/a$)	0.94	2.19
Lake Concentration ($\mu g/l$)	70	162
Inflow Concentration ($\mu g/l$)	145	407

Table 7.59 details the external phosphorus loading according to source. The largest contribution comes from inorganic fertilisers, whether under low or high export levels. Animal waste supplies between 29% and 37%, and settlements and roads also generate a significant amount, within a range of 14% to 19%.

Table 7.59: Export of Phosphorus to Petty Pool According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.05	49.36	0.11	44.92
Organic Fertiliser	0.03	28.54	0.09	37.02
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.02	19.18	0.04	14.10
Woodland	0.00	0.74	0.00	1.60
Direct Precipitation	0.00	2.18	0.01	2.35

7.1.14.2 Recommendations

The main inflow to Petty Pool appears to receive a major excretal input of nutrients, the origin of which is unknown. This requires investigation by the NRA, so that appropriate action may be taken. The model indicates that other significant sources of phosphorus are diffuse agricultural losses and urban runoff.

A 20% reduction both in inorganic and in organic fertiliser, and the establishment of 20m riparian buffer zones, as well as a 50% decrease in phosphorus from urban runoff through the use of detention ponds, would theoretically produce a mean annual TP concentration from external sources of between 26 and 68μg/l, and an inflow TP concentration within the range of 45 to 140μg/l. Since high export coefficients appear to predict inflow TP concentrations better, these measures would place Petty Pool in the eutrophic, rather than mesotrophic category, as illustrated in Fig.7.27.

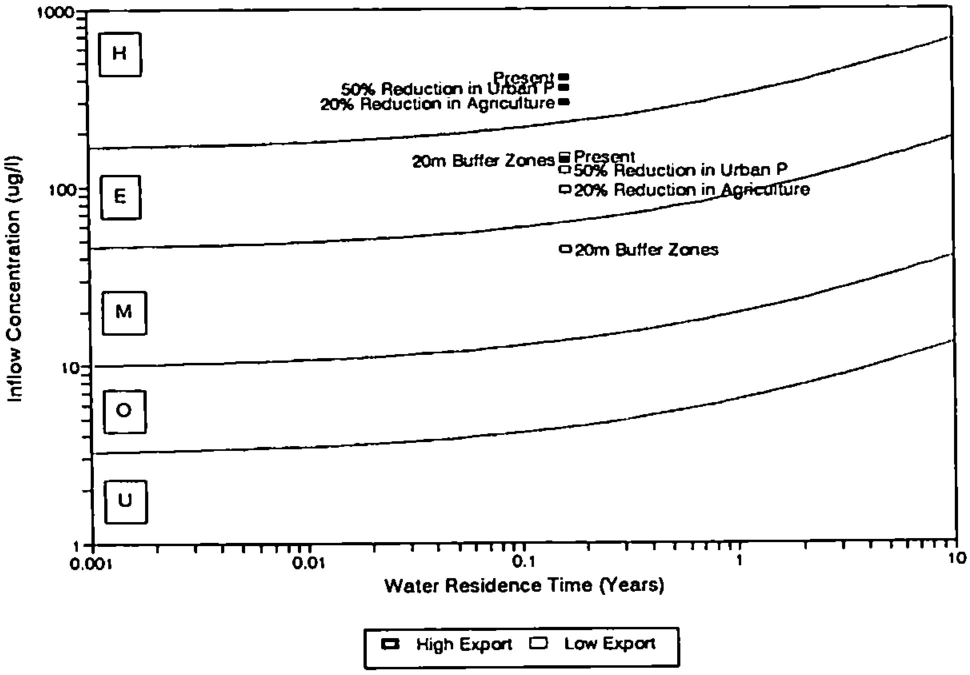

Fig.7.27: Effect of Reductions in External Phosphorus Loading on Petty Pool

7.1.15 Quoisley Meres

7.1.15.1 Results

A description of this site may be found in Section 6.6.6. Land use results for the catchment of Quoisley Meres (Fig.7.28), calculated from ITE data, are shown in Table 7.60. In this small catchment of about 140 hectares more than 95% of the land is devoted to agricultural production, with over half of it as grassland of one type or another.

Information contained in the parish returns indicates that about 70% of holding are specialist or mainly dairy farms. The remainder are either specialist poultry units or mixed farms. The former contain nearly 80,000 birds, but it could not be established whether they are situated within the catchment.

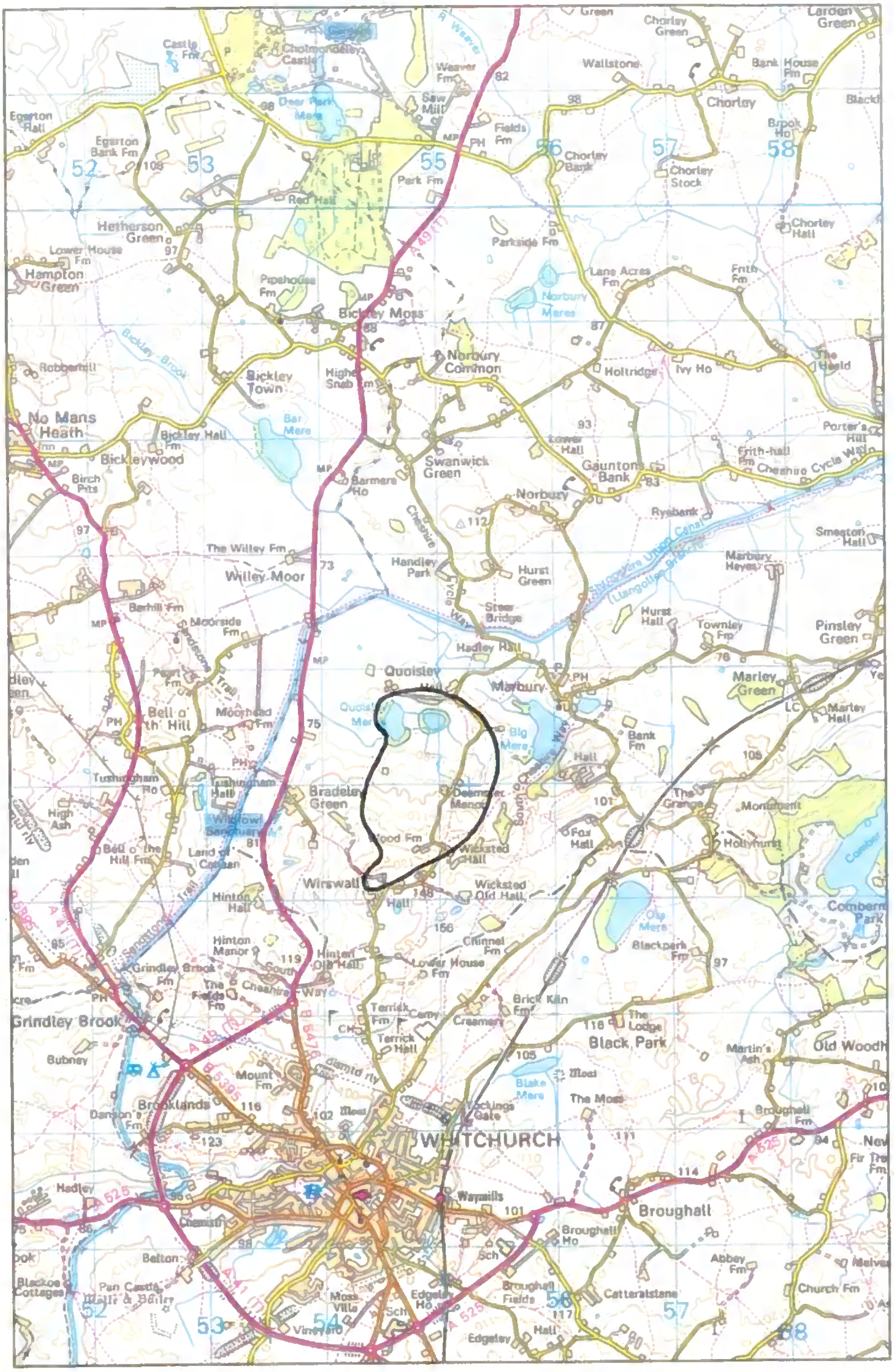

Fig.7.28: The Catchment of Quoisley Meres

Table 7.60: Land Use in the Catchment of Quoisley Meres

Land Use	Area (ha)	Percentage
Temporary Grassland	20.86	14.75
Permanent Grassland	45.36	32.08
Arable	63.24	44.73
Rough Grazing	5.63	3.98
Settlements and Roads	5.96	4.22
Woodland	0.00	0.00
Rivers	0.33	0.23
Total	141.39	100.00

Data on total livestock numbers required reduction by 88% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parishes. The results, shown in Table 7.61, are therefore less than reliable.

Table 7.61: Livestock Levels in the Catchment of Quoisley Meres

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	356	48	80	9154	14

Phosphorus loading results for Quoisley Meres are detailed in Table 7.62. Total crude load is predicted to be between 0.07 and 0.17t/a, which, on an areal basis, converts to 1.11 and 2.68g/m^2/a. These results produce a mean annual TP concentration from external sources of between 168 and 407μg/l, and an inflow TP concentration within the range of 460 and 1356μg/l, placing Quoisley Meres in the hypertrophic category. A mean inflow TP concentration of 167μg/l for 1991/2, calculated from measured data in Moss *et al.* [1992], suggests that the use of low export coefficients is more appropriate in this catchment.

Table 7.62: Predicted External Phosphorus Loadings on Quoisley Meres

Parameters	Low Export	High Export
Total Load (t/a)	0.07	0.17
Areal Load ($g/m^2/a$)	1.11	2.68
Lake Concentration ($\mu g/l$)	168	407
Inflow Concentration ($\mu g/l$)	460	1356

Table 7.63 lists the external phosphorus loading according to source. Diffuse agricultural sources account for between 94% and 95% of the total loading, with most of the remainder emanating from settlements and roads.

Table 7.63: Export of Phosphorus to Quoisley Meres According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.03	49.41	0.07	40.17
Organic Fertiliser	0.03	44.43	0.09	54.92
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.00	4.35	0.01	3.05
Woodland	0.00	0.00	0.00	0.00
Direct Precipitation	0.00	1.81	0.00	1.86

7.1.15.2 Recommendations

Quoisley Meres have been affected in the past by farm effluent pollution similar to that occurring in the catchments of several of the Meres previously discussed. This problem may now be alleviated by the preparation of a Whole Farm Conservation Plan. However, since diffuse agricultural sources account for such a large proportion of phosphorus inputs, this aspect of farming needs to be addressed in addition to point sources. A general reduction in the intensity of agriculture throughout the catchment may therefore be necessary.

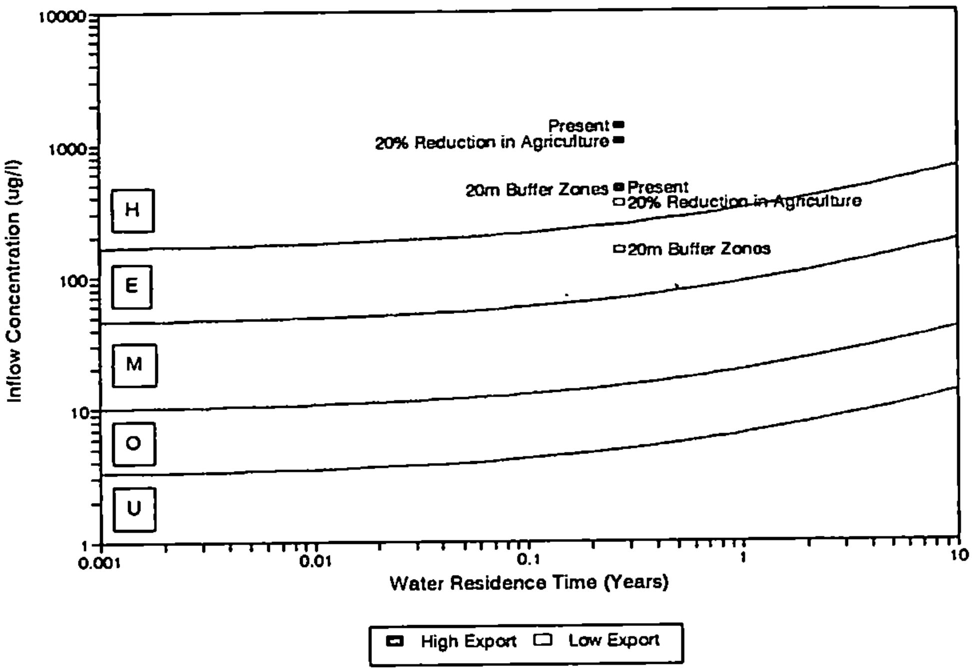

Fig.7.29: Effect of Reductions in External Phosphorus Loading on Quoisley Meres

A 20% reduction both in stocking levels and in organic fertiliser, and the establishment of 20m riparian buffer zones would produce a mean annual TP concentration from external sources of between 70 and 166μg/l, and an inflow TP concentration within the range of 157 to 453μg/l. Considering that the lower export level appears to predict more closely the inflow TP concentration, adoption of the above strategies may place Quoisley Meres in the eutrophic, rather than hypertrophic, category. These Meres may also benefit from the removal of drainage ditches for some distance around their perimeter. This would allow fringing vegetation to act as a nutrient buffer, and may also prevent the direct entry of farm effluent into the Meres.

7.1.16 Tabley Mere

7.1.16.1 Results

A description of this site may be found in Section 6.6.7. Land use results for the catchment of Tabley Mere (Fig.7.30), calculated from ITE data, are shown in Table 7.64. The catchment, which extends over about 1,200 hectares, is composed of 85% agricultural land and 15% urban areas.

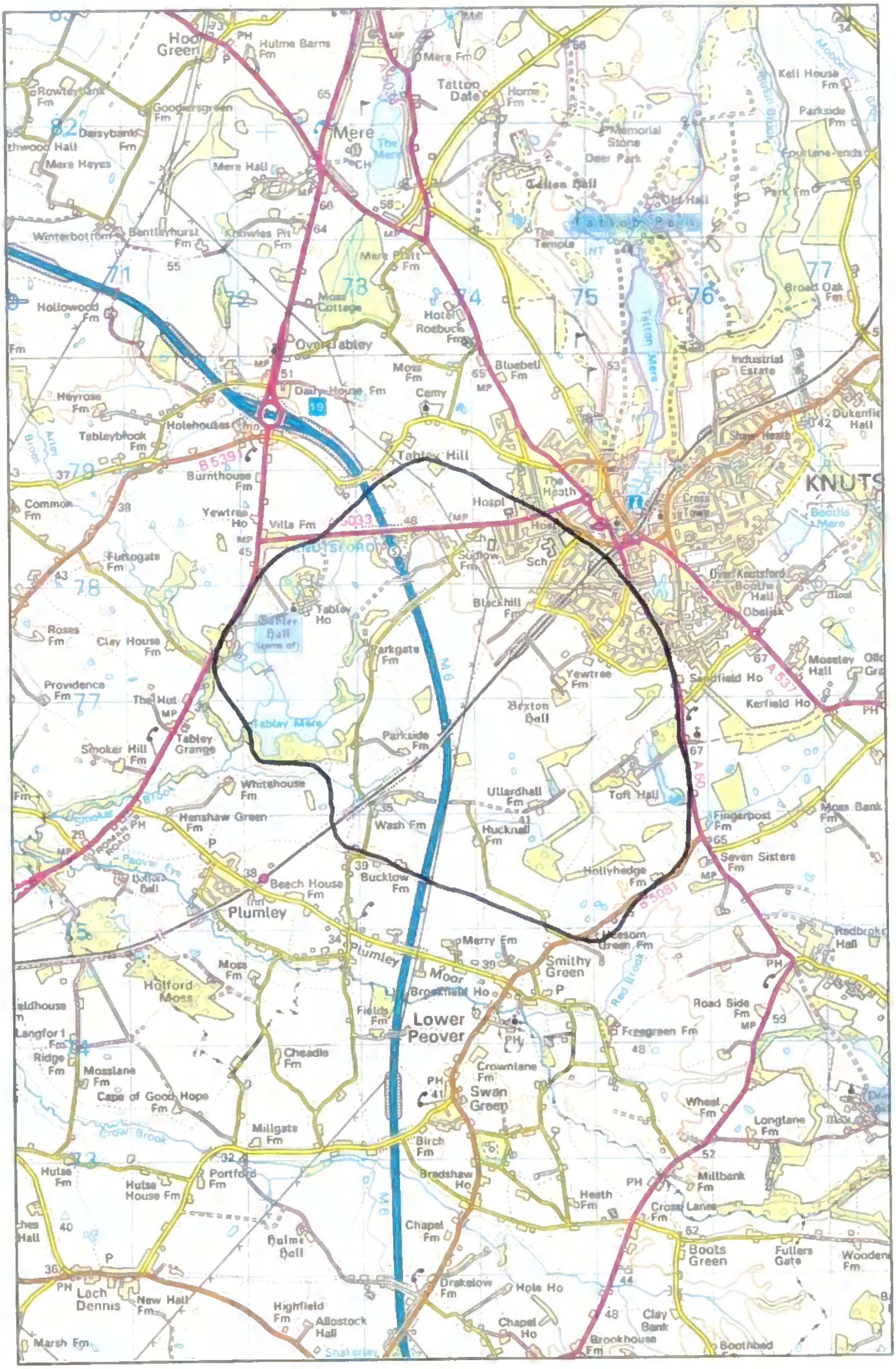

Fig.7.30: The Catchment of Tabley Mere

Information contained in the parish returns reveals that about 35% of the holdings are operated on a part-time basis. Of the remainder, 62% are specialist or mainly dairy, while 14% are devoted to general crops, and a further 14% are horticultural units.

Table 7.64: Land Use in the Catchment of Tabley Mere

Land Use	Area (ha)	Percentage
Temporary Grassland	179.07	15.50
Permanent Grassland	417.21	36.10
Arable	305.52	26.44
Rough Grazing	78.46	6.79
Settlements and Roads	168.91	14.62
Woodland	0.00	0.00
Rivers	6.46	0.56
Total	1155.63	100.00

Data on total livestock numbers required reduction by 65%. The results, shown in Table 7.65, are therefore somewhat unreliable.

Table 7.65: Livestock Levels in the Catchment of Tabley Mere

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	1193	509	15	2990	18

Phosphorus loading results for Tabley Mere are detailed in Table 7.66. Total crude load is predicted to be between 0.36 and 0.78t/a, which, on an areal basis, converts to 1.88 and 4.03g/m^2/a. These results produced a mean annual TP concentration from external sources of between 354 and 760μg/l, and an inflow TP concentration within the range of 1182 and 3001μg/l, placing Tabley Mere in the hypertrophic category. A mean inflow TP concentration of 759μg/l for 1991/2, calculated from measured data in Moss *et al.* [1992], suggests that the use of low export coefficients is more appropriate in this catchment.

Table 7.66: Predicted External Phosphorus Loadings on Tabley Mere

Parameters	Low Export	High Export
Total Load (t/a)	0.36	0.78
Areal Load ($g/m^2/a$)	1.88	4.03
Lake Concentration ($\mu g/l$)	354	760
Inflow Concentration ($\mu g/l$)	1182	3001

Table 7.67 provides a breakdown of the external phosphorus loading according to source. Diffuse agricultural sources contribute the major part, ranging between 72% and 78% of the total. Settlements and roads represent a significant source of 18% to 23%. Sewage sources, from Tabley House Nursing Home, contribute 2% to 4%.

Table 7.67: Export of Phosphorus to Tabley Mere According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.19	50.87	0.39	49.59
Organic Fertiliser	0.08	20.62	0.22	28.82
Sewage Effluent	0.02	4.25	0.02	1.98
Settlements and Roads	0.08	23.30	0.14	18.37
Woodland	0.00	0.00	0.00	0.00
Direct Precipitation	0.00	1.07	0.01	1.24

7.1.16.2 Recommendations

Diffuse agricultural sources appear to be the main target for phosphorus reduction strategies, according to the model. However, two other sources are of significance and require attention. First, it is reported that effluent, possibly from Tabley House Nursing Home, is causing a culverted ditch to smell of sewage. This matter should be investigated by the NRA, and appropriate action taken. Second, the consented discharges for runoff from the M6 and intermittent sewage overflow may be contributing significant amounts of phosphorus to Tabley Mere, as well as causing considerable pollution and silting. The sewage component of these discharges needs to be addressed through upgrading and expanding the capacity of the pumping stations from which they originate. The creation of detention ponds incorporating oil traps for runoff from the M6 is likely to reduce both the nutrient and pollution components considerably, and to obviate the need to dredge Serpentine Water. The NRA appears satisfied that such works will be carried out when upgrading of the M6 and Knutsford Service Area occurs, but this may be several years away [pers. comm. C. Hayes, EN, Shrewsbury]. Meanwhile, Tabley Mere is in danger of losing the rich aquatic flora for which it was notified.

A significant decrease in loading levels from diffuse agricultural sources may be achieved through a 20% reduction both in stocking levels and in inorganic fertilisers, and the establishment of 20m riparian buffer zones, as well as a 50% reduction in phosphorus from urban runoff through the use of detention ponds. These measures would produce a mean annual TP concentration from external sources in the range of 118 and 243μg/l, and an inflow TP concentration of between 311 and 746μg/l. These results mean that Tabley Mere remains in the hypertrophic category, but at a much reduced level. The Mere may also benefit from the removal of field drains for some distance around its perimeter, which would prevent the direct entry of runoff from surrounding farmland.

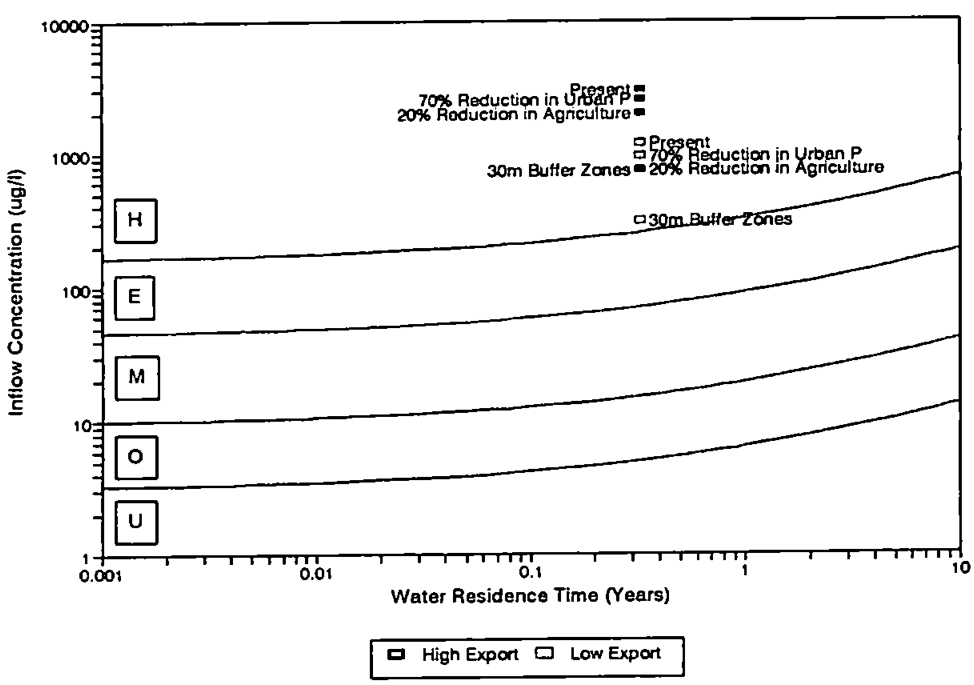

Fig.7.31: Effect of Reduction in External Phosphorus Loadings on Tabley Mere

7.1.17 Tatton Mere

7.1.17.1 Results

A description of this site may be found in Section 6.6.8. Land use results for the catchment of Tatton Mere (Fig.7.32), calculated from ITE data, are shown in Table 7.68. The catchment, covering some 400 hectares, encompasses a high proportion of agricultural land (55%), as well as a significant amount of urban area (30%). Information on the types of holdings present indicates that 42% of farms are run on a part-time basis. Of the remainder, 43% are specialist dairy units.

Table 7.68: Land Use in the Catchment of Tatton Mere

Land Use	Area (ha)	Percentage
Temporary Grassland	35.05	8.99
Permanent Grassland	103.49	26.55
Arable	60.93	15.63
Rough Grazing	14.19	3.64
Settlements and Roads	117.68	30.19
Woodland	55.92	14.35
Rivers	2.50	0.64
Total	389.76	100.00

Agricultural data for the parish of Tatton were unavailable, since fewer than three holding are present. Data on livestock numbers for Knutsford parish required reduction by 58% to allow for the difference in the area of agricultural land in the catchment, compared to that of the parish. The results are shown in Table 7.69.

Table 7.69: Livestock Levels in the Catchment of Tatton Mere

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	206	133	0	10	0

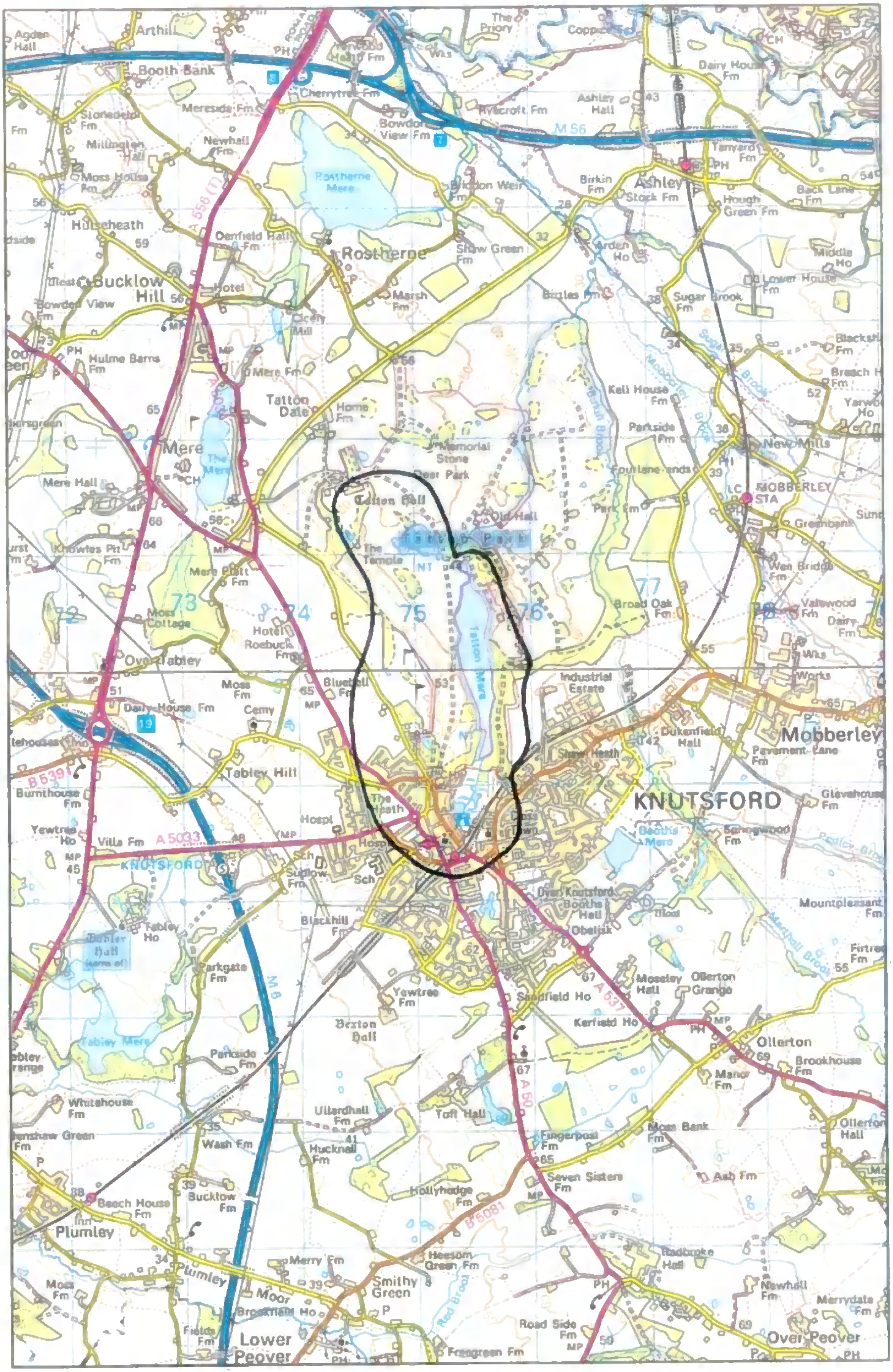

Fig.7.32: The Catchment of Tatton Mere

Phosphorus loading results for Tatton Mere are detailed in Table 7.70. Total crude load is predicted to be between 0.12 and 0.24t/a, which, on an areal basis, converts to 0.37 and 0.75g/m^2/a. These results produce a mean annual TP concentration from external sources within the range of 96 to 195μg/l, and an inflow TP concentration of between 295 and 704μg/l, placing Tatton Mere in the eutrophic to hypertrophic category. There are no measured inflow data with which to compare these results, but a mean TP concentration of 233μg/l for 1991/2 has been recorded for the Mere itself.

Table 7.70: Predicted External Phosphorus Loadings on Tatton Mere

Parameters	Low Export	High Export
Total Load (t/a)	0.12	0.24
Areal Load (g/m^2/a)	0.37	0.75
Lake Concentration (μg/l)	96	195
Inflow Concentration (μg/l)	295	704

Table 7.71 provides a division of the external phosphorus loading according to source. Diffuse agricultural sources account for between 43% and 50% of the total load, while settlements and roads contribute most of the remainder at 42% to 50%. Although a CSO discharges near the headwaters of the inflow, such intermittent sources are not accounted for in the model.

Table 7.71: Export of Phosphorus to Tatton Mere According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.04	32.49	0.08	33.68
Organic Fertiliser	0.01	10.99	0.04	16.16
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.06	50.51	0.10	42.09
Woodland	0.00	0.58	0.00	1.41
Direct Precipitation	0.01	5.44	0.02	6.67

7.1.17.2 Recommendations

Since urban runoff and diffuse agricultural losses account for the majority of the external loading, these sources are the main targets for phosphorus reduction strategies. A significant decrease in loading levels may be achieved through a 20% reduction both in inorganic and in organic fertiliser, and the establishment of 20m riparian buffer zones, as well as a 50% reduction in phosphorus from urban runoff through the use of detention ponds. These measures would produce a mean annual TP concentration from external sources within the range of 41 to 93μg/l, and an inflow TP concentration of between 104 and 287μg/l, placing Tatton Mere in the eutrophic category.

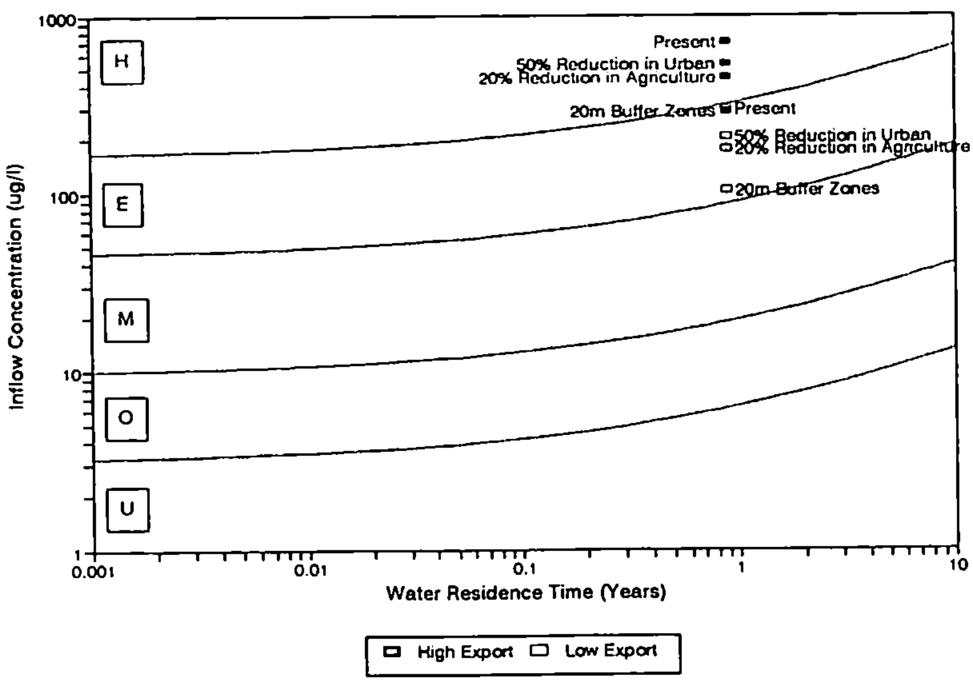

Fig.7.33: Effect of Reductions in External Phosphorus Loading on Tatton Mere

There is, however, a source of nutrients which is not accounted for by the methodology, that is, the consented discharge for a CSO located near the headwaters of the stream feeding Tatton Mere. Although plans to upgrade this facility have been mooted, the work has not yet occurred. It seems apparent that until improvements are made the nature conservation interest of Tatton Mere will continue to decline.

7.1.18 Malham Tarn

7.1.18.1 Results

A description of this site may be found in Section 6.7.1. Land use results for the catchment of Malham Tarn (Fig.7.34), calculated from ITE data, are shown in Table 7.72. The catchment, covering some 560 hectares, is composed primarily of rough grazing (63%), although permanent grassland (22%) and temporary pasture (12%) are also significant. Arable land and urban areas are negligible in comparison. Information in the parish returns indicates that about one quarter of farms operate on a part-time basis, and that the remainder are concerned with rearing and fattening either sheep or cattle and sheep. Small areas devoted to crops are used solely to produce feed for stock.

Table 7.72: Land Use in the Catchment of Malham Tarn

Land Use	Area (ha)	Percentage
Temporary Grassland	66.59	11.90
Permanent Grassland	120.89	21.61
Arable	9.22	1.65
Rough Grazing	351.40	62.82
Settlements and Roads	11.27	2.01
Woodland	0.00	0.00
Rivers	0.00	0.00
Total	559.37	100.00

In moorland areas, such that surrounding Malham Tarn, parishes tend to be extensive, compared to the lowlands. Consequently, the catchment of Malham Tarn covers only 13% of a single parish. The results for livestock numbers are shown in Table 7.73.

Table 7.73: Livestock Levels in the Catchment of Malham Tarn

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	68	2214	3	3	2

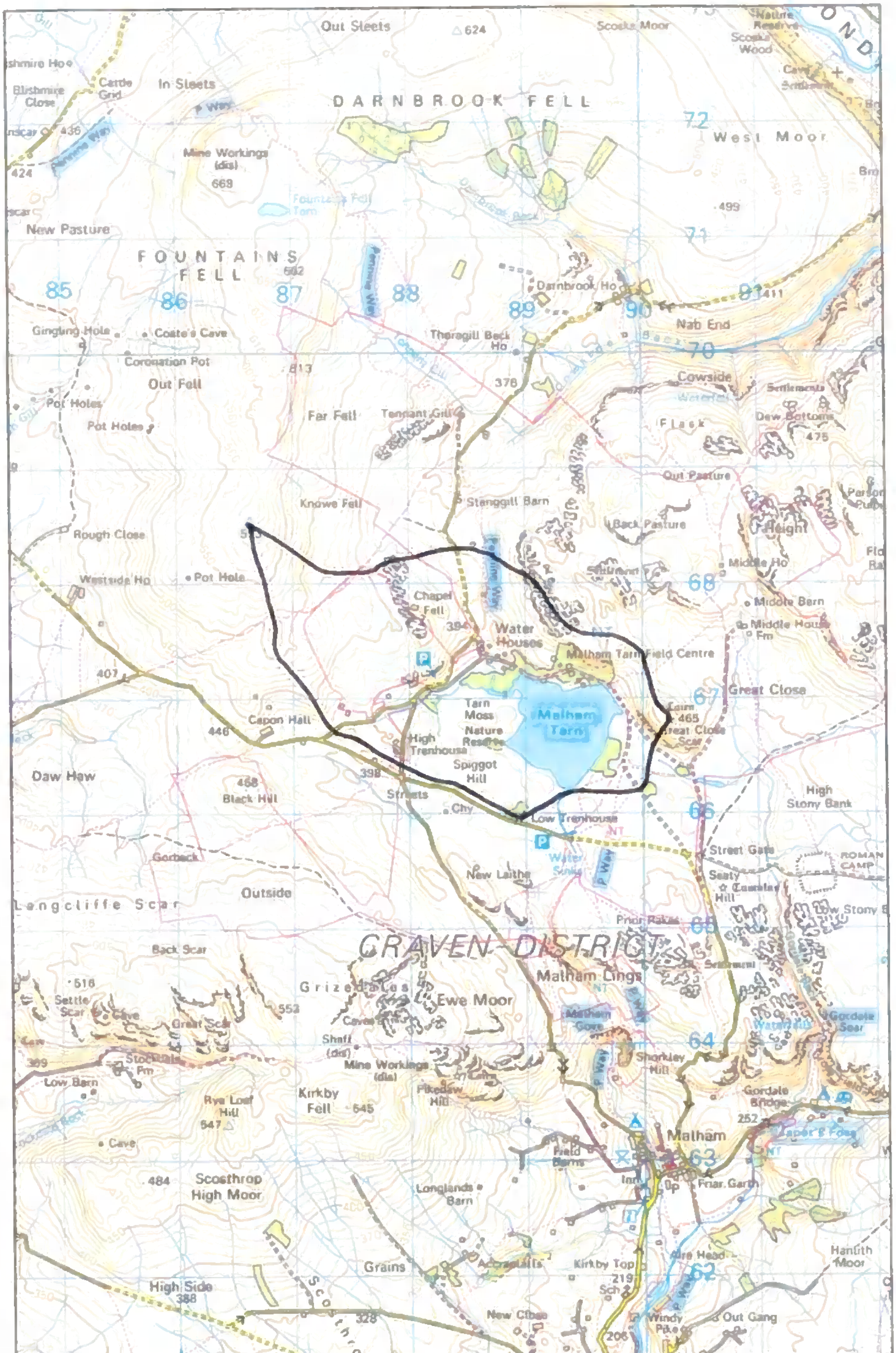

Fig.7.34: The Catchment of Malham Tarn

Phosphorus loading results for Malham Tarn are detailed in Table 7.74. Total crude load is predicted to be between 0.06 and 0.16t/a, which, on an areal basis, converts to 0.10 and 0.26g/m^2/a. These results produce a mean annual TP concentration from external sources of between 6 and 16μg/l, and an inflow TP concentration within the range of 8 to 25μg/l, which places Malham Tarn in the oligotrophic to mesotrophic category (Fig.7.35). There are no measured inflow data with which to compare these results, but for the Tarn itself a range of 3 to 17μg/l is recorded.

Table 7.74: Predicted External Phosphorus Loadings on Malham Tarn

Parameters	Low Export	High Export
Total Load (t/a)	0.06	0.18
Areal Load (g/m^2/a)	0.10	0.28
Lake Concentration (μg/l)	6	16
Inflow Concentration (μg/l)	8	25

Table 7.75 provides a breakdown of external phosphorus loading according to source. Animal waste contributes between 36% and 40%, while inorganic fertilisers accounts for about 34% under both low and high export levels. The amount provided by direct precipitation appears to be significant, but this is more a reflection of the relatively low contribution from the land.

Table 7.75: Export of Phosphorus to Malham Tarn According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.02	33.65	0.06	34.41
Organic Fertiliser	0.02	35.94	0.06	40.11
Sewage Effluent	0.00	0.00	0.00	0.00
Settlements and Roads	0.01	9.40	0.01	5.94
Woodland	0.00	0.00	0.00	0.00
Direct Precipitation	0.01	21.01	0.03	19.54

7.1.18.2 Recommendations

The nutrient status of Malham Tarn appears to be unchanged, as a result of low input farming and lack of development in its catchment. At one point, there was concern that septic tank effluent from Malham Tarn Field Centre was entering the lake, but this is now thought to occur only during heavy rainfall. New septic facilities are to be installed which will incorporate a reed bed system for tertiary treatment. However, the performance level of such schemes in terms of nutrient removal is low, especially over time. The use of reed bed systems may seem attractive, in the sense that they provide a quasi-natural means of treating wastewater. However, scientific realities should not be discarded in the pursuit of alternatives to technical and engineering solutions.

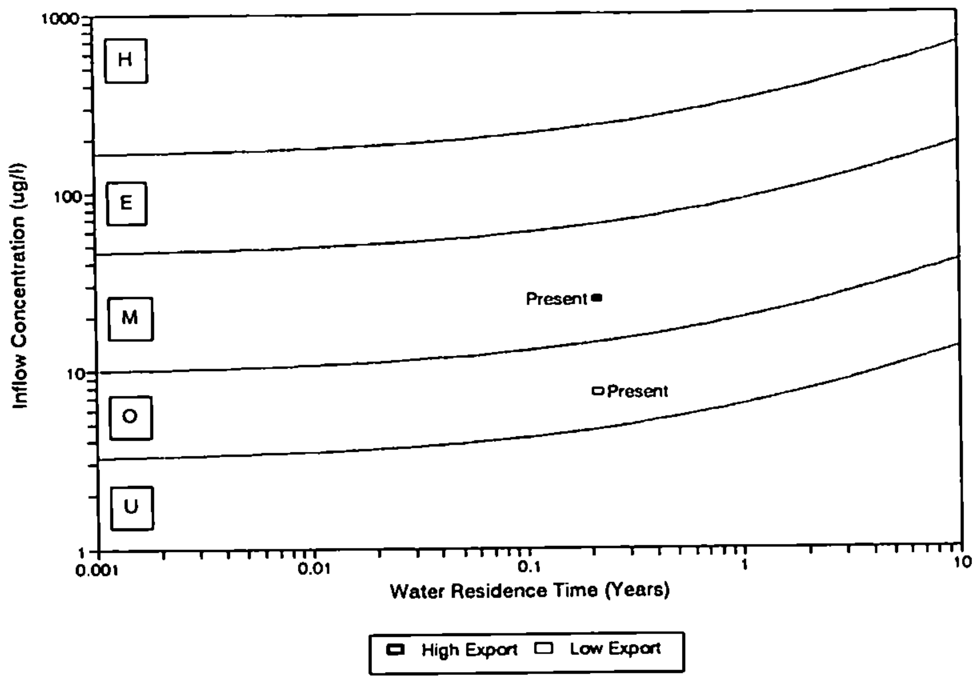

Fig.7.35: External Phosphorus Loading on Malham Tarn

In view of the maintenance of water quality in Malham Tarn, no phosphorus reduction strategies are necessary as yet. However, monitoring data should be regularly analysed in order to ensure that any increases in phosphorus concentrations are detected, and the source(s) identified.

7.1.19 Semerwater

7.1.19.1 Results

A description of this site may be found in Section 6.7.2. Land use results for the catchment of Semerwater (Fig.7.36), calculated from ITE data, are shown in Table 7.76. The catchment, which extends over some 4,400 hectares, is composed primarily of rough grazing (76%), although some permanent grassland (11%) and temporary pasture (5%) also exist. Information in the parish returns indicates that about one third of farms operate on a part-time basis. Of the remainder, half are specialist or mainly dairy farms, and over a third rear sheep or cattle and sheep. Several mixed farms also exist.

Table 7.76: Land Use in the Catchment of Semerwater

Land Use	Area (ha)	Percentage
Temporary Grassland	208.17	4.77
Permanent Grassland	490.84	11.25
Arable	73.47	1.68
Rough Grazing	3313.43	75.97
Settlements and Roads	70.41	1.61
Woodland	144.91	3.32
Rivers	60.21	1.38
Total	4361.44	100.00

The catchment of Semerwater covers almost all of the parish of Bainbridge, so that data on livestock numbers required reduction by only 12%. Results for livestock levels are given in Table 7.77. They indicate the predominance of sheep, but also reveal that cattle are much more important than in the catchment of Malham Tarn.

Table 7.77: Livestock Levels in the Catchment of Semerwater

Livestock	Cattle	Sheep	Pigs	Poultry	Other Fowl
Adjusted Total	1668	20606	189	142	7

Phosphorus loading results for Semerwater are detailed in Table 7.78. Total crude load is predicted to be between 0.41 and 1.11t/a, which, on an areal basis, converts to 1.55 and 4.18g/m^2/a. These results produce a mean annual TP concentration from external sources of between 28 and 77μg/l, and an inflow TP concentration within the range of 42 to 141μg/l, placing Semerwater in the mesotrophic to eutrophic category (Fig.7.37). There are no measured inflow data with which to compare these results, but a mean annual SRP concentration of 10μg/l, and a summer mean of 64μg/l [Barlow, 1994] have been recorded for the lake itself.

Table 7.78: Predicted External Phosphorus Loadings on Semerwater

Parameters	Low Export	High Export
Total Load (t/a)	0.41	1.11
Areal Load (g/m^2/a)	1.55	4.18
Lake Concentration (μg/l)	28	77
Inflow Concentration (μg/l)	42	141

Table 7.79 provides a division of the phosphorus loading according to source. Animal waste contributes the greater part (64% to 71%), and inorganic fertiliser accounts for 20% to 22%. Septic tanks provide about 2% to 5%, less than the proportion for settlements and roads.

Table 7.79: Export of Phosphorus to Semerwater According to Source

Source	Low Export		High Export	
	t/a	%	t/a	%
Inorganic Fertiliser	0.09	21.32	0.23	20.30
Organic Fertiliser	0.26	63.62	0.79	70.74
Sewage Effluent	0.02	4.69	0.02	1.74
Settlements and Roads	0.04	8.53	0.06	3.37
Woodland	0.00	0.35	0.01	0.65
Direct Precipitation	0.01	1.29	0.01	1.20

7.1.19.2 Recommendations

Semerwater, according to the model, lies in the mesotrophic to eutrophic category. It has been variously described as mesotrophic and moderately eutrophic, but the lack of water quality data renders assessment difficult. High turbidity is thought to be caused by suspended sediment, rather than phytoplankton productivity. Nevertheless, the more intensive agriculture in this catchment, compared to that of Malham Tarn, may cause Semerwater to be eutrophic. The occurrence of algal blooms in 1989 and 1990 also indicates that this lake may be overloaded with respect to nutrients.

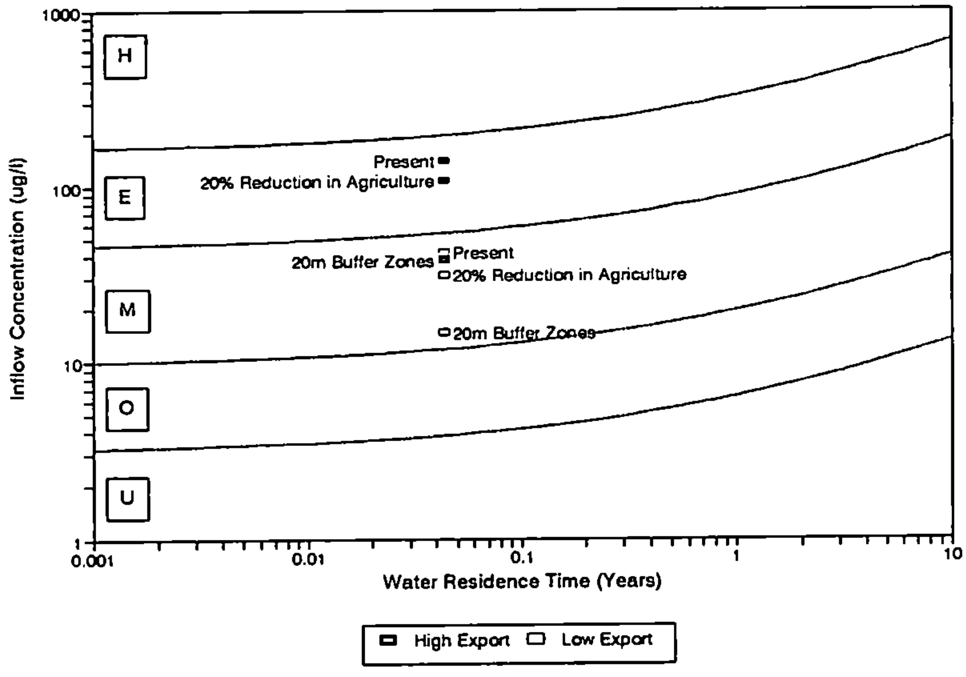

Fig.7.37: Effect of Reductions in External Phosphorus Loading on Semerwater

The prime source of nutrients is diffuse agricultural losses. A lessening in the intensity of agriculture throughout the catchment may therefore be necessary. This could be achieved by a 20% reduction both in stocking density and in inorganic fertiliser application, and the establishment of 20m riparian buffer zones.These measures would, in theory, produce a mean annual TP concentration of between 12 and 26μg/l, and an inflow TP concentration within the range of 15 to 38μg/l, placing Semerwater in the mesotrophic category.

7.2 General Discussion of Results

As emphasised in Sections 3.2 and 5.3.1, for the purposes of calibration and verification results generated by the use of export coefficients require measured data on phosphorus concentrations in inflows. All available phosphorus concentration data for the sites for which loading predictions were produced are shown in Table 7.80. These data do not include the reservoirs of Eastern England, since the majority of their loadings consist of phosphorus concentrations in pumped inputs. The table serves to illustrate the paucity of relevant data for the majority of sites, and the difficulties involved in verifying the results obtained in this study in a quantitative manner. Nevertheless, measured inflow values are available for a limited number of sites.

A comparison between measured and predicted inflow concentrations, shown in Fig.7.38, indicates that for two sites, Slapton Ley and Chapel Mere, there is a fair degree of concurrence. However, the model appears to underpredict for Comber Mere and Petty Pool, and to overpredict for Quoisley and Tabley Meres, using both low and high export coefficients.

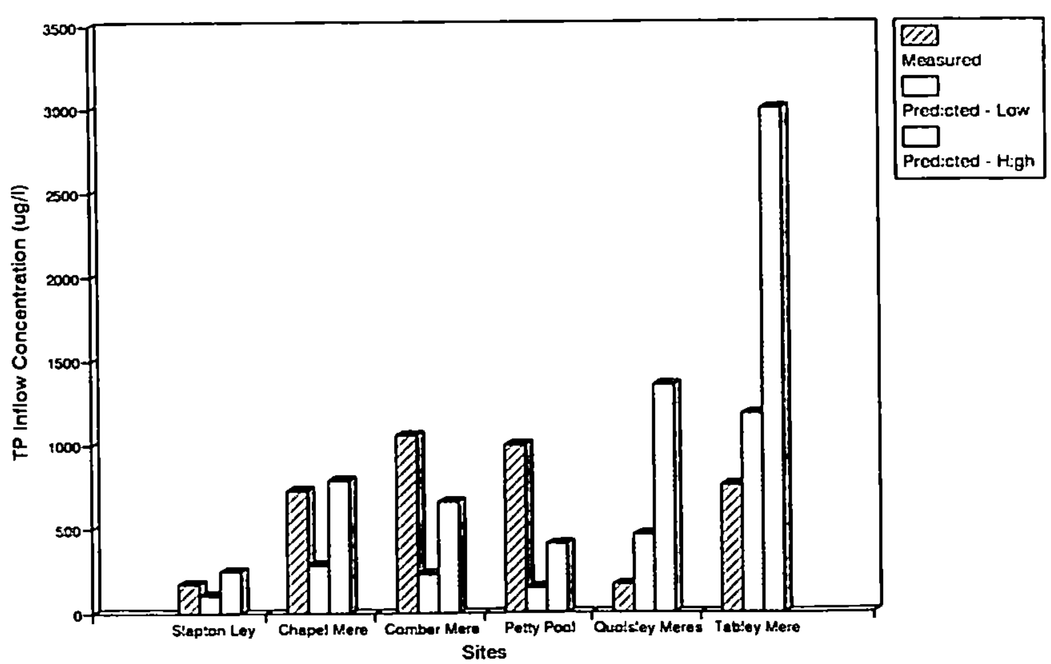

Fig.7.38: Comparison of Measured and Predicted Inflow Concentrations

Table 7.80: Measured and Predicted Phosphorus Concentrations

TP (μg/l) unless otherwise stated

Site	This Study	Moss *et al.* [1992]		Johnes, Moss and Phillips [1994]		Carvalho and Moss [1995]	Other Studies
	Inflow	Inflow	In-Lake	Inflow	In-Lake	In-Lake	In-Lake
Loe Pool	1092-1283	-	-	-	270	145 (SRP)	70 (PO_4)[1]
Slapton Ley	104-250	-	-	170	150	510 (SRP)	111 (SRP)[2]
Bay Pond	17-46	-	-	-	-	255	25-4520 (PO_4)[3]
Hedgecourt Lake	758-1619	-	-	-	-	60	-
Barton Broad	104-212	-	-	-	90	180	150[4]
Aqualate Mere	160-388	-	-	-	-	385 (SRP)	-
Chapel Mere	286-789	727	1267	-	-	-	-
Comber Mere	233-659	1060	362	-	-	360	-
Petty Pool	145-407	999	261	-	-	260	-
Quoisley Meres	460-1356	167	-	-	-	405	-
Tabley Mere	1182-3001	759	323	-	-	325	-
Tatton Mere	295-704	-	233	-	-	235	-
Malham Tarn	8-25	-	-	-	50	20	3-17[5]
Semerwater	42-141	-	-	-	-	125	10[6]

(1) Flory and Hawley [1994]
(2) Neal [1994]
(3) Data supplied by Surrey Wildlife Trust
(4) Data supplied by NRA Haddiscoe
(5) Cottrill *et al.* [1991], cited by Hinton [1991a]
(6) Yorkshire Water Authority, cited in Squance [1980]

Correlation between measured data and the most appropriate values either at the low or the high end of the predicted range produces a coefficient of 0.47. This figure indicates that there is a weak association between measured and predicted values. Regression analysis produces an equation for the 'best-fit' line plotted in Fig.7.39. The graph confirms that the results for Slapton Ley and Chapel Mere are reasonable, but that those for the other four sites do not compare well.

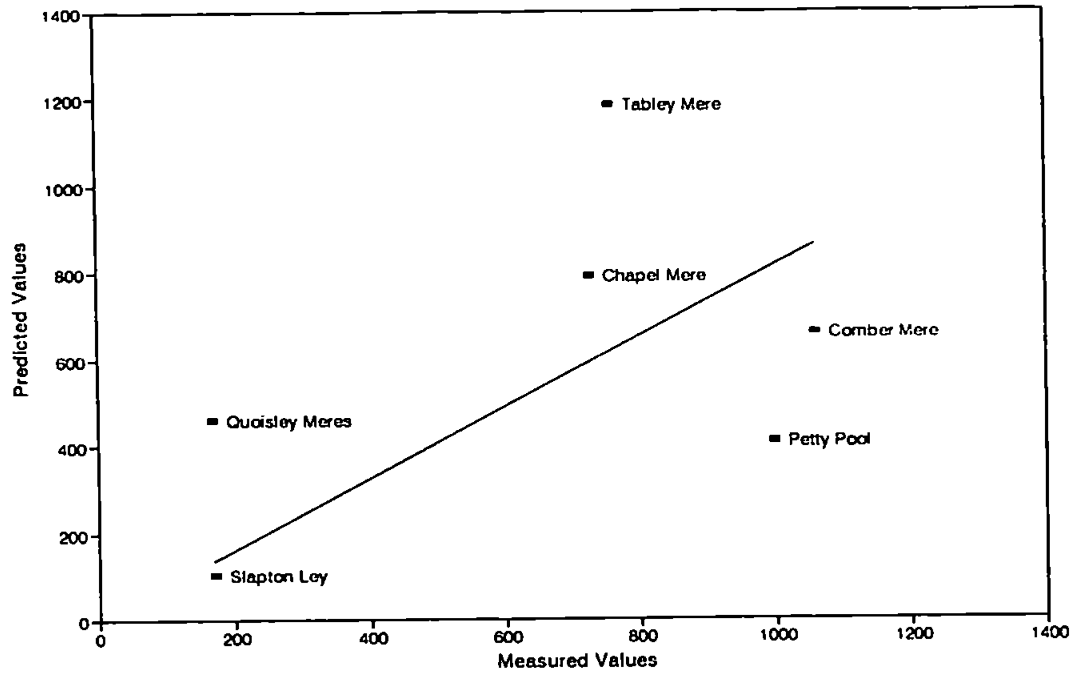

Fig.7.39: Regression Analysis of Measured and Predicted Inflow Concentrations

Prior to an analysis of possible reasons for differences between measured and predicted inflow concentrations, two points need to be stressed. First, although regression analysis appears to suggest that there is a weak association between measured and predicted values, the degree of discrepancy is not evident in terms of orders of magnitude. Even for Tabley Mere, where the greatest difference occurs, the value predicted by the use of low export coefficients is only one and a half times the measured data. As illustrated in Fig.7.40, one or other of the extremes of the predicted range lies within the same trophic category as the measured value, with the exception of Quoisley Meres. In other words, in terms of the ability of the model to predict trophic status it appears to work reasonably well, especially considering the degree of error inherent in the data used in the export coefficient approach.

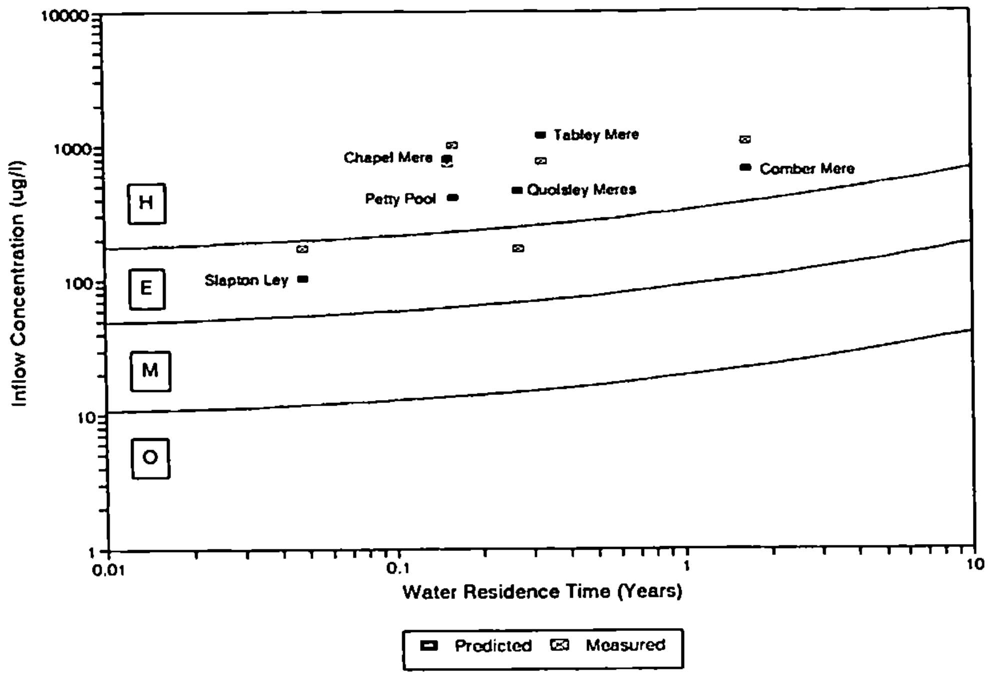

Fig.7.40: Trophic Status According to Measured and Predicted Inflow Concentrations

Second, the measured data with which the predicted values are compared may not necessarily be representative of contemporary inflow concentrations. Values for the Shropshire-Cheshire Meres are based on analyses of samples collected during one hydrological year (1991/92), while those for Slapton Ley are derived from studies undertaken during the 1970s and early 1980s. Taking these points into consideration, there appears to be a fair degree of agreement between predicted values and measured inflow data. Nevertheless, further insights may be gained through an analysis of circumstances under which discrepancies have been generated.

Lack of consistency between measured and predicted inflow concentrations may be attributable to a number of factors:

(a) the use of inappropriate export coefficients
(b) the use of inaccurate land use and livestock data
(c) the presence of sources of phosphorus which are not accounted for in the model
(d) lack of information on drainage patterns created by urbanisation and construction of major roads

(e) the operation of processes associated with phosphorus transport and cycling which are not accounted for in the model, such as deposition and biological uptake

(f) the use of unrepresentative measured data

In the case of Tabley Mere, the measured inflow concentration is considerably higher than the value predicted under low export coefficients. If the model were able to accommodate contributions from the two CSOs in the catchment the difference might be greater. However, two factors may have exerted an influence on the predicted value. First, about 15% of the catchment consists of urban land, which contributes roughly 23% of the phosphorus loading under low export coefficients. The drainage pattern of this area is unknown, and it is possible that surface runoff is directed into another catchment.

Second, data on total livestock numbers required reduction by 65%, thus rendering the results for this component of the loading somewhat unreliable. This same reservation may be applied to the data employed for Petty Pool, Chapel Mere, Quoisley Meres, and Comber Mere, in that livestock numbers required reduction by 66%, 87%, 88%, and 91% respectively. In contrast, numbers used for Slapton Ley represent data from one parish which lies wholly within the catchment and 56% from the remaining parishes.

The problem of ascertaining drainage patterns in urban areas may also have a bearing on the prediction for Petty Pool, which even under high export coefficients is considerably lower than the measured value. About 10% of the catchment consists of urban land, which contributes roughly 19% of the loading. It is possible, however, that the catchment receives drainage from a more extensive area of urban land.

As outlined above, it is only feasible to apply a quantitative analysis of results to those sites for which measured inflow data are obtainable. For most of the remaining sites, in-lake concentrations may be used to provide a qualitative assessment of the validity of the results. There are, in fact, only three sites where differences of any note occur, namely Loe Pool, Bay Pond and Hedgecourt Lake. For Aqualate and Tatton Meres, Malham Tarn and Semerwater predicted values are consistent with measured in-lake data (Table 7.80).

In the case of Loe Pool, where 62% to 81% of the loading is attributable to phosphorus levels in sewage effluent, and the predicted inflow concentration is considerably higher than measured in-lake values (Table 7.80), it seems likely that the contribution from sewage is overestimated. This hypothesis is strengthened by the discrepancy noted between predicted and measured figures for mean annual TP loads from STWs above Barton Broad. It is also possible, however, that rates of deposition and biological uptake of phosphorus in the 1km of river between Helston STW and Loe Pool, and in the lake itself, may be sufficient to reduce the concentration considerably. Data obtained through monitoring of this site by the NRA will be extremely useful in ascertaining the contribution of sewage effluent to loadings on Loe Pool.

A possible explanation for the low predicted inflow concentration for Bay Pond, compared to measured phosphate values was discussed in Section 7.1.3.2. It seems likely that the source of high in-lake phosphate levels is a sediment trap situated upstream of Bay Pond, since nitrate concentrations are reasonably low. In the case of Hedgecourt Lake the reverse situation exists, in that predicted inflow concentrations of between 758 and 1619μg/l are not consistent with a measured in-lake value of only 60μg/l. However, the measured value is based on a single sample and, as such, does not merit full consideration. It may be that the problems associated with calculating contributions from urban runoff affect this catchment, since 23% of the area is devoted to settlements and roads, but this question remains open without more adequate monitoring data.

In summary, the effectiveness of the export coefficient approach in predicting trophic status appears to be reasonably acceptable, taking into account the degree of error in the data required to produce predictions, and the lack of measured phosphorus inflow values with which to calibrate and verify the model. It may be, however, that a more pragmatic approach to the application of export coefficients lies in the production of values for the relative contributions of external sources of phosphorus, thereby enabling the construction of catchment management strategies. In addition, conclusions drawn from the site specific modelling exercise can be used to draw conclusions relevant to the problem of eutrophication of freshwater SSSIs in general. In the following Section the modelling results are synthesised to produce recommendations on a national level with respect to those sources which require most attention.

7.3 General Recommendations

As demonstrated in Table 7.81, diffuse losses of phosphorus from agriculture are the prime source of enrichment for the majority of sites. Other significant losses occur in urban and/or road runoff, in consented discharges, and from agricultural point sources, which affect individual lakes to a varying degree.

Table 7.81: Summary of Site Specific Phosphorus Reduction Targets

Site	Urban/Road Runoff	Consented Discharges	Agricultural Point Sources	Agricultural Nonpoint Sources
Aqualate Mere				**□**
Barton Broad				**□**
Bay Pond				
Chapel Mere			**□**	**□**
Comber Mere		**□**	**□**	**□**
Grafham Water		**□**		□
Hedgecourt Lake	**□**			**□**
Loe Pool		**□**		**□**
Malham Tarn				
Norbury Meres			**□**	**□**
Petty Pool	**□**	□	□	**□**
Pitsford Reservoir		**□**		□
Quoisley Meres			□	**□**
Rutland Water		**□**		□
Semerwater				**□**
Slapton Ley				**□**
Sutton Park	**□**	**□**		
Tabley Mere	**□**	**□**		**□**
Tatton Mere	**□**	**□**		**□**

□ Significant Sources of Phosphorus □ Possible Sources of Phosphorus

A diverse range of measures are necessary for the reduction of loadings from these sources, and the extent to which current legislation may be applied differs considerably. This section seeks to make general recommendations which may be applied within the constraints of the legislative structure. Further suggestions will also be made with respect to changes in the law necessary for the alleviation of eutrophication.

7.3.1 Urban and Road Runoff

Several sites are situated in catchments in which urban land and/or major roads cover a significant proportion of the area. As outlined in Section 2.1.5, urban and road drainage systems are designed to remove surface runoff as quickly and efficiently as possible, and therefore transport large volumes of water to rivers. Discharges represent intermittent influxes of high concentrations of nutrients, causing short- and long-term effects. The phosphorus content of stormwater outflows is not measured on a routine basis, yet a mean TP concentration of 340μg/l is reported for the UK. The problem may be addressed in a number of ways, the most fundamental of which is to allow rainwater to infiltrate into natural subsurface drainage systems. While it is feasible to incorporate infiltration facilities into new developments, and to replace impermeable surfaces in older conurbations gradually, there is the need for a more immediate solution.

The construction of detention ponds to accommodate road and urban runoff is a strategy worthy of consideration for those sites most affected by this source of phosphorus. Such a measure does not necessarily require the application of new legislation. Under current planning legislation the NRA may make strong representation for the installation of detention ponds in new developments. Within the consultation process, there is also the opportunity for English Nature (EN) to present comment on the benefits of detention ponds in cases where runoff would enter a freshwater SSSI. In the case of new major roads, however, changes in policy are necessary within the Department of Transport and the DoE so that detention ponds for runoff are incorporated as a matter of course.

7.3.2 Consented Discharges

Several sites affected by phosphorus levels in consented discharges are those which receive effluent from STWs or septic tanks. Since parameters selected for consent limits do not normally include TP, there is no scientific basis on which objections to discharges into SSSIs may be made. A prime example is seen in the case of Loe Pool. A request for data on nutrient levels in effluent from Helston STW met with the response "consent limits are not set for nitrogen and phosphorus as the levels of these nutrients in the final effluent are negligible" [letter from H. Richards, South West Water Services Ltd., St. Austell, 6th January 1993]. The argument therefore becomes circular, in that without data on phosphorus levels no problem is perceived.

Implementation of the UWWT Directive will reduce phosphorus loadings on a number of important sites, and provide the opportunity for monitoring of nutrient levels in effluent from larger STWs, including Helston. However, this legislation fails to address the cumulative effect of numerous smaller STWs on freshwater SSSIs. Phosphorus concentrations in effluent from *all* STWs discharging into rivers feeding SSSIs need to be monitored, so that data is at least available to assess the relative contribution of such discharges to eutrophication. Responsibility for the collection of such data lies with the NRA, but, under current economic restraints, its budget for monitoring work allows only a minimum amount to be undertaken.

Three sites in particular, Sutton Park, Tabley Mere and Tatton Mere, appear to be affected significantly by periodic discharges from CSOs. Mean concentrations of 10mg/l TP and 90mg/l BOD are reported for CSOs in the UK. It seems contradictory that water companies are legally able to discharge raw sewage into rivers, while farmers may be prosecuted for similar pollution events. Since it is stormwater surges which trigger the operation of CSOs, the problem needs to be addressed at that level, with the installation of facilities to detain surface runoff. However, changes in policy on urban drainage may be some time away, and a more immediate solution to the problem is necessary.

The upgrading of CSO facilities to provide additional stormwater capacity and enhanced screening of solids is being undertaken on a gradual basis by the water companies. It appears, however, that priority is given to those CSOs which cause greatest public annoyance; there does not seem to be any ecological criteria as a basis for urgent upgrading work. It is evident in the case of Tabley Mere, where two CSOs merge with polluted runoff from the M6 motorway, that such discharges cause severe disruption to the ecology of freshwater SSSIs. In order to address the problem of CSOs a change of policy is required within the DoE, to enable the NRA to apply regulatory pressure on the water companies.

7.3.3 Agricultural Point Sources

Several sites appear to be suffering from long-term chronic organic pollution, as a result of farm wastes entering watercourses. The sites in question all lie within areas of intensive dairy farming, and possess relatively small catchments, so that pollution of this nature is likely to produce a greater effect. It is probable that similar cases occur at other sites, but are not so readily detectable. As stated several times in the previous section, the solution to such problems lies with the NRA, and its ability to advise or prosecute offenders. There is also a role to be played by such organisations as the Farming and Wildlife Advisory Group, in persuading farmers to adopt Whole Farm Conservation Plans, which incorporate protection of water quality.

7.3.4 Agricultural Nonpoint Sources

It is clear that diffuse agricultural loss of phosphorus is the main contributory factor in the eutrophication of many freshwater SSSIs. The reduction of such losses is not readily managed within the current legislative framework. Indeed, this aspect of water quality appears to have received least attention from policy makers and legislators in the UK and the EC. Yet the present trend towards agricultural extensification presents a prime opportunity for the formulation of targeted programmes aimed at environmental problems such as eutrophication.

The legal instrument most suited to the reduction of diffuse agricultural phosphorus losses is the power to designate catchments as Water Protection Zones (WPZs) under the Water Resources Act 1991. As outlined in Section 4.4.2.2, there are several drawbacks to this measure. First, the NRA seems to be aware that the control of diffuse agricultural sources of phosphorus presents regulatory problems to which it is unaccustomed and for which it is ill-equipped. Second, the resistance of the farming lobby to further restrictions on their activities is inevitable, but this may be reduced if MAFF, rather than the NRA, is the main administrative body. Third, compensation is not automatically attached to designation. Clearly, the WPZ concept requires adaptation.

Certain criteria appear to be necessary for a comprehensive scheme aimed at the reduction of diffuse agricultural losses of phosphorus to eutrophicated lakes and reservoirs. A scheme specifically designed for this purpose would be:

(a) based on catchment-wide designation

(b) targeted at catchments of lakes and reservoirs affected by eutrophication, with priority given to those of freshwater SSSIs

(c) administered by MAFF, in cooperation with the NRA and EN

(d) monitored by the NRA

(e) with adequate compensation attached to designation

Bearing in mind the future requirements of the proposed Directive on the Ecological Quality of Surface Waters, it seems likely that such a scheme will be necessary. The NRA is already pursuing the possibility of collaborative research on eutrophication with the Environment Protection Agency in the Irish Republic [*Water Guardian*, 1995b], where a comparable scheme aimed at reducing phosphorus inputs to lakes is in existence [pers. comm., M. Wiseman, NRA, Sale].

Chapter 8

Conclusions and Recommendations for Future Work

The primary aim of this study, that is, to evaluate management strategies for shallow, eutrophicated lakes and reservoirs, with a view to identifying those best suited to the restoration of such ecosystems to ecologically desirable conditions, has been achieved. For each site the potential total phosphorus loading was quantified using an export coefficient approach, and the effect of possible reduction strategies on trophic status evaluated. A critical examination of these methodologies was undertaken. A review of legislation which may be related to the alleviation of eutrophication of freshwater SSSIs was carried out, and recommendations for future legislative and policy changes proposed.

8.1 The Eutrophication Debate

An understanding of the process of eutrophication necessitates an appreciation of the close relationship between the biogeochemical character of a lake and the biogeochemical nature of its catchment. The nutrient status of a lake is almost entirely dependent on the geological, biological, topographical and climatic characteristics of its drainage basin, and on the level and types of human activity within it. A catchment-wide approach is therefore necessary in programmes which seek to address eutrophication, in order both to implement strategies designed to restrict activities conducive to phosphorus losses, and to integrate long-term precautionary policies into the planning process.

The detrimental effects of eutrophication are often perceived as those which interfere with human uses of lakes, such as the production of toxic algal blooms. From the point of view of nature conservation, however, relatively small increases in the supply of nutrients to a lake may cause loss of species. By the time more obvious symptoms are apparent community structure may be severely disrupted. Eutrophication therefore needs to be addressed at an early stage, in order both to protect nature conservation interests, and to avoid the necessity for costly restoration measures.

A related issue in the debate surrounding eutrophication is the extent of the problem in the UK. If the definition of eutrophication is restricted to the more extreme cases in which noticeable deleterious symptoms are evident, then it is indeed less widespread. However, if a wider, and, from a scientific viewpoint, more accurate meaning is attached to the term, then nutrient enrichment is a much more widespread and common environmental problem, the impact of which requires recognition.

8.2 Prediction and Evaluation of Phosphorus Loadings

The export coefficient approach to the prediction of phosphorus loadings on eutrophicated lakes, as used in this study, carries a number of limitations and inherent errors, as discussed in Chapter 3. There are, however, two main problems with this method. First, the unit used for obtaining data on livestock is the parish, which seldom bears any spatial coincidence with catchments. The degree of error generated by the necessary adjustments may render the data unreliable in many cases. Second, although the use of separate export coefficients for organic and inorganic sources may be useful for identifying appropriate management strategies, the scientific basis for such an approach is dubious.

For these reasons, the results produced by this method cannot be taken as absolute. Nevertheless, the phosphorus loading and consequent trophic status predicted for each lake were not unreasonable. On the whole, they were as one might expect, given the catchment characteristics and nutrient enrichment history of particular lakes. However, it is unlikely that the accuracy of the results would be reduced by the utilisation of land use export coefficients only. Significantly fewer data would be required, and the need to employ parish returns obviated. Remote sensing techniques for obtaining land use data are continually being improved, and investment in such technology may need to be a priority for agencies with an interest in the effects of land use on the environment.

With respect to the Vollenweider-OECD model, the trophic status predictions for individual lakes were reasonable, as noted above. The primary limitation in its application is that, without scientific knowledge of levels of phosphorus inputs prior to eutrophication, loading reduction objectives cannot be evaluated. There is therefore a need to couple the model with a method of obtaining information on past productivity (see below).

8.3 Phosphorus Reduction Strategies

Those aspects of human activity which are mainly responsible for eutrophication may be categorised as the disposal of wastewater, and intensive agricultural production. It is evident that significant reductions in the contribution of phosphorus from these sources is required. The most effective means of achieving such reductions appear to be:

(a) tertiary treatment of domestic wastewater for the removal of nutrients
(b) diversion of sewage effluent to sea where feasible
(c) the detention of urban runoff to prevent stormwater surges, thus preventing the operation of combined sewer overflows (CSOs)
(d) the establishment of riparian nutrient buffer zones along watercourses
(e) a reduction in the intensity of agriculture throughout catchments

The phosphorus reduction strategy necessary for the majority of sites under assessment was a lessening in the intensity of agriculture within their catchments. However, a significant number required reduction of phosphorus in consented discharges, in the form of effluent from sewage treatment works (STWs) and septic tanks, or from CSOs. For some, agricultural point sources needed attention. In several less rural catchments, reductions in the contribution of phosphorus from urban and road runoff were necessary.

8.4 The Legislative Framework

As the discussion in Chapter 4 illustrated, there is no single UK law which may be used to implement reductions in phosphorus loadings on eutrophicated freshwater SSSIs. The synthesis of legislation relating to nature conservation, to water quality, and to the extensification of agriculture demonstrated that existing laws may be applied in a piecemeal fashion, but that there is a need for a new, integrated approach. The SSSI system is not designed to protect aquatic sites from the cumulative effects of activities carried out in their catchments. The regulatory system for water quality is not constructed to deal effectively with diffuse agricultural sources of nutrients. Legislation relating to the extensification of agriculture may provide opportunities for achieving reductions in nutrient losses, but as a side effect, rather than as a direct consequence.

In applying current policy and legislation to the phosphorus reduction strategies required for the sites under assessment, more specific deficiencies were apparent. The present planning system includes the necessary mechanisms to ensure the installation of detention ponds for surface runoff in new and substantially altered developments, but policy is not strong on this point. Implementation of the Urban Waste Water Treatment Directive will affect levels of phosphorus in effluent from large STWs, but does not address the chronic, cumulative effect of contributions from smaller works, especially in ecologically fragile contexts. CSO facilities are in the process of being upgrading, but progress is slow as a result of government policy in relation to the privatised water companies. With respect to agricultural point sources, the regulatory system operated by the NRA should ensure that action is taken against offenders. There is also legislation which may be applied to the reduction of diffuse sources of phosphorus, through the designation of Water Protection Zones, but such a scheme requires certain adaptations before it is entirely appropriate. The Habitat Scheme recently introduced at a pilot level by MAFF attempts to establish buffer zones along watercourses. However, the removal of field drains, which is essential for nutrient reduction, is presented as an additional option, rather than as a prerequisite.

Future policy and legislation in relation to the control of eutrophication therefore needs to address two main issues. First is the problem of phosphorus levels in effluent from STWs serving fewer than 10,000 people which discharge into watercourses feeding freshwater SSSIs. Tertiary treatment of such effluents should be seen as a cost-effective investment in water quality and in nature conservation. Second, a significant reduction in the intensity of agriculture within the catchments of freshwater SSSIs is required. This may be achieved through the designation of such catchments under a scheme specifically designed for the purpose. Prescriptions for the scheme might include the establishment of riparian buffer zones, reductions both in stocking levels and in fertiliser application rates, and measures designed to lessen soil erosion, such as contour cultivation and the creation of lateral hedgerows. Specific measures for each catchment would need to be established, based on knowledge of farming practices, soil type, climate and topography.

8.5 Recommendations for Future Work

There are two main areas in which the work of this study may be carried forward. First, the recommendation outlined above for a scheme designed to address the problem of diffuse agricultural losses of phosphorus requires an assessment of the economic implications. The expenditure involved in reduction strategies for other sources of phosphorus have been investigated at various levels, but as yet, the costs and benefits of dealing with diffuse agricultural sources are largely unknown. The success of a scheme such as the one proposed, in terms of public acceptance and support, depends on all aspects of the attendant economics being explored, including the benefits to society of maintaining a rich diversity of aquatic habitats.

Second, the eutrophication history of lakes requires quantification by a more direct means than the utilisation of 'hindcasted' values generated by export coefficients. Recent advances in the use of palaeolimnology to predict past nutrient levels provide considerable scope for the generation of reasonably reliable results. In particular, work on diatom-inferred phosphorus profiles developed by Anderson, Rippey and Gibson [1993] and Bennion [1994] has shown that it is possible to deduce pre-enrichment lake phosphorus concentrations from the sedimentary record. Although there are certain margins of error involved in this technique, they appear to be smaller than those generated by the use of export coefficients.

Finally, while our knowledge of pre-eutrophication phosphorus levels in English lakes is understandably slight, it is apparent from this study that there is also a paucity of data on their current nutrient status. The modelling of phosphorus transport and the effects of loadings on trophic status may be developed to an infinite degree, but it cannot act as a replacement for measured data. The necessity to produce a lake classification system in order to comply with EC legislation has elevated the priority given to monitoring by the NRA. It is regrettable, however, that although monitoring data is available for many major rivers for the past several decades, comparable information is lacking for lakes. The importance of lake monitoring data needs to be realised at government level, so that restoration measures may be based on the best available knowledge.

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