A Preliminary Assessment of the Environmental Costs of the Eutrophication of Fresh Waters in England and Wales

J.N. Pretty, C.F. Mason, D.B. Nedwell and R.E. Hine

Centre for Environment and Society
and
Department of Biological Sciences
University of Essex

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Contents

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The authors can be contacted at:

Centre for Environment and Society and the Department of Biological Sciences
University of Essex
Wivenhoe Park
Colchester CO4 3SQ, UK

Author E-mails:

jpretty@essex.ac.uk; masoc@essex.ac.uk; nedwd@essex.ac.uk, rehine@essex.ac.uk

Acronyms

BAP	Biodiversity Action Plan
BW	British Waterways
EA	Environment Agency
EEA	European Environment Agency
FWR	Foundation for Water Research
DEFRA	Department for Environment, Food and Rural Affairs
DETR	formerly the Department for Environment, Transport and the Regions (now DEFRA)
HAP	Habitat Action Plan
MAFF	The former Ministry of Agriculture, Fisheries and Food (now DEFRA)
NSA	Nitrate Sensitive Area
NVZ	Nitrate Vulnerable Zone
OECD	Organisation for Economic Cooperation and Development
SAP	Species Action Plan
STW	Sewage Treatment Works
UNEP	United Nations Environment Programme

Executive Summary

The enrichment of water bodies and water courses with inorganic plant nutrients is a process known as eutrophication. It can adversely affect the diversity of the biological system, the quality of the water and the uses to which water may be put. No national study of the costs arising from the problems caused by eutrophication has yet been undertaken in the UK. In order to inform management decisions and assist in the prioritisation of remedial actions, this research was commissioned to produce an indicative assessment of the environmental costs of cultural eutrophication.

This research was commissioned by the Environment Agency, and comprises a scoping study with three aims:

i. to produce a preliminary estimate of the cost of eutrophication in fresh waters of England and Wales;
ii. to begin to put the costs of eutrophication in perspective with other environmental problems;
iii. to identify areas where costs are currently unknown and further research is required.

We have developed a new framework of cost categories for assessing the costs of eutrophication, based on the pressure-state-response framework. The pressures driving nutrient enrichment and eutrophication arise from both point and non-point sources of nutrients. We distinguish between two major types of cost category (see Categories A and B in Figure 1). These are the damage costs (or value-loss costs) arising from the reduced value of clean or non nutrient-enriched water, and the policy response costs incurred in responding to eutrophication damage plus the costs of changing behaviour and practices to meet legal obligations. Damage costs cannot be added to policy response costs, as the later is a measure of how much is being spent to deal with eutrophication problems.

Damage costs, by definition, represent a loss of existing value, rather than an increase in costs, and are divided into two categories: use values and non-use values. Use values are associated with private benefits gained from actual use (or consumption) of ecosystem services, and can include private sector uses (eg agriculture, industry); recreation benefits (eg fishing, water sports); education benefits; general amenity benefits; and option values (the desire of an individual to maintain the choice to use an ecosystem's services in the future). Non-use values are of two types: bequest values and existence values.

Social damage costs (or use values) can usually be valued by market prices, whereas non-use values are more difficult to calculate. We identify ten types of use value:

- reduced value of waterside dwellings;
- reduced value of water bodies for commercial uses (abstraction, navigation, livestock watering, irrigation and industry);
- drinking water treatment costs (treatment and action to remove toxins and algal decomposition products);
- drinking water treatment costs (to remove nitrogen);
- clean-up costs of waterways (dredging, weed-cutting);
- reduced value of non-polluted atmosphere (via greenhouse and acidifying gases);
- reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling, and general amenity (picnics, walking, aesthetics);
- net economic losses for formal tourist industry;

- net economic losses for commercial aquaculture, fisheries, and shell-fisheries;
- health costs to humans, livestock and pets.

Ecological damage costs (non-use values) comprise the damage caused to biota and ecosystem structure by nutrient enrichment.

We also assess costs arising from policy responses to the problems of eutrophication, as it is currently difficult fully to assess all damage costs in category A. When a water body is recognised in some way as being adversely affected by nutrient enrichment, then costs may be incurred through a variety of social responses by both statutory and non-statutory agencies. We divide these direct costs of responding to eutrophication and costs of changing behaviour and practices into two types: compliance control costs, and direct costs incurred by agencies.

There are three types of compliance cost arising from the adverse effects of eutrophication, including sewage treatment costs (to remove phosphorus arising from large point sources); costs of treatment of algal blooms and in-water preventative measures (biomanipulation, destratification, straw bale deployment etc.); and costs of adopting new farm practices that emit fewer nutrients.

The final cost category comprises the direct costs incurred by statutory agencies for monitoring, investigating and enforcing solutions to eutrophication, and comprise monitoring costs for water and air; and the cost of developing and implementing eutrophication control policies and strategies.

Putting a cost on eutrophication as an environmental problem is a complex task for the simple reason that there is no absolute definition of when nutrient enrichment becomes a problem. The central issue is the nature of the relationship between nutrient enrichment, the biological responses and resultant effects on other sectors, the biota and water users. This can be difficult to define. For each category, we begin by reflecting on the causes of the problem and summarise existing scientific evidence. We then scope the cost relationship in the form of an equation. Where possible, we then quantify the extent of the problem in the UK (frequency and extent), and summarise the existing economic data on costs and value-losses.

Despite the many gaps in the datasets, we estimate the damage costs of freshwater eutrophication in England and Wales to be £75.0-114.3 million yr^{-1}. A breakdown of these costs for each category is shown in Table E1. The annual cost of cultural eutrophication in England and Wales is dominated by seven items with costs of £10 million or more each:

i. reduced value of waterfront dwellings;
ii. drinking water treatment costs for nitrogen removal;
iii. reduced recreational and amenity value of water bodies;
iv. drinking water treatment costs for removal of algal toxins and decomposition products;
v. reduced value of non-polluted atmosphere;
vi. negative ecological effects on biota; and
vii. net economic losses from the tourist industry.

These findings indicate the severe effects of nutrient enrichment and eutrophication on many sectors of the economy. The substantial damage costs (£75 to £114.3 million) are causing considerable loss of value to many stakeholders in the UK. The policy response costs are a measure of how much is being spent to address this damage, and these amount to £54.8 million yr^{-1}. In common with other environmental problems, it would represent net value (or cost reduction) if this damage was prevented at source. A variety of efficient economic, regulatory and administrative policy instruments are available to seek to internalise these costs, thus ensuring that both the `polluter pays' the cost, and the `provider (of clean or unpolluted water) gets' the benefits.

Table E1. Summary of the annual costs of freshwater eutrophication in the UK

Cost categories		**Range of annual costs (£ million)**
A. Damage costs – the reduced value of clean or non nutrient-enriched water		
A1. Social damage costs		
i.	reduced value of waterside dweliings;	£9.83
ii.	reduced value of water bodies for commercial uses (abstraction, navigation, livestock watering, irrigation and industry);	£0.50-1.00
iii.	drinking water treatment costs (treatment and action to remove algal toxins and algal decomposition products);	£19.00
iv.	drinking water treatment costs (to remove nitrogen);	£20.10
v.	clean-up costs of waterways (dredging, weed-cutting);	£0.50-1.00
vi.	reduced value of non-polluted atmosphere (via greenhouse and acidifying gas emissions);	£5.12-7.99
vii.	reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling, and general amenity (picnics, walking, aesthetics);	£9.65-33.54
viii.	revenue losses for formal tourist industry;	£2.94-11.66
ix.	revenue losses for commercial aquaculture, fisheries, and shell-fisheries;	£0.029-0.118
x.	health costs to humans, livestock and pets.	unknown
A2. Ecological damage costs		
i.	negative ecological effects on biota (arising from changed nutrients, pH, oxygen), resulting in changed species composition (biodiversity) and loss of key or sensitive species.	£7.34-10.12
	TOTAL	£75.0-114.3
B. Policy response costs – costs incurred in responding to eutrophication		
B1. Compliance control costs arising from adverse effects of nutrient enrichment		
i.	sewage treatment costs to remove phosphorus arising from large point sources;	£50.30
ii.	costs of treatment of algal blooms and in-water preventative measures (biomanipulation, stratification, straw bale deployment);	£0.50
iii.	costs of adopting new farm practices that emit fewer nutrients.	£3.39
B2. Direct costs incurred by statutory agencies for monitoring, investigating and enforcing solutions to eutrophication		
i.	monitoring costs for water and air;	£0.44
ii.	cost of developing eutrophication control policies and strategies.	£0.20
	TOTAL	£54.8

We identify five major policy and research priorities:

1. This study has mostly used aggregated national data to produce an estimate of the total cost of freshwater cultural eutrophication, as there is a lack of comprehensive and harmonised data on specific catchments and river basins. There is an urgent need for greater analysis of representative catchments in order to understand better the nutrient budgets and loads, costs being occurred, and the most beneficial and cost-effective actions, all of which are requirements under the Water Framework Directive.

   We recommend model/pilot studies are conducted on representative whole catchments or river basins to identify the sources of nutrients, produce detailed nutrient budgets, analyse eutrophication outcomes, and produce estimates for the costs and benefits of prevention and remediation.

   We further recommend that such a study focus on developing the appropriate methodology, and on implementation of large-scale comprehensive rehabilitation of one moderate to large catchment.

2. This study set out to examine the costs of cultural eutrophication only in fresh waters. This inevitably leaves open the question of the costs being incurred in marine and estuarine waters.

   We recommend further research on the degree of eutrophication in UK marine and estuarine waters, and the costs currently being incurred both in the UK and in other European countries. The Water Framework Directive requires river basin management, including effects on the coastal environment.

3. There remains uncertainty over the definition of the point at which nutrient enrichment becomes a eutrophication problem with adverse economic effect (both costs and value losses).

   We recommend further analysis of the nature of the nutrient-enrichment and eutrophication relationship, and more coordination of data on eutrophication between agencies to ensure joint and efficient responses.

4. There are many gaps in the datasets held by a wide variety of agencies and organisations with both statutory and non-statutory interests and responsibilities in eutrophication. There is a requirement for improved data on the extent of ecological and social damage, and on the costs of in-water preventative and remedial measures.

   We recommend more coordination between agencies of data on eutrophication and on its effects and costs to ensure improved joint responses.

5. There remains considerable uncertainty over the specific effects of eutrophication on recreation and tourism, and on the lives of those living and working by affected water courses.

   We recommend further research on the value of water-based tourism and sports (both freshwater and marine) and the specific value losses caused by eutrophication.

1. Scope of the Research Project

Nutrient enrichment of water bodies and water courses is a process known as eutrophication, defined by the Environment Agency (2000a) as: *"the enrichment of water by nutrients, stimulating an array of symptomatic changes including increased production of algae and/or higher plants, which can adversely affect the diversity of the biological system, the quality of the water and the uses to which the water may be put"*.

The Environment Agency has recently published a strategy for the management of aquatic eutrophication in England and Wales (EA, 1998). This outlines a framework for managing eutrophication using a combination of national activities to reduce nutrient inputs to water and local catchment-based action in partnership with other organisations and individuals.

The Agency recognises eutrophication as a priority environmental issue, as evidenced in a recent report on the state of the freshwater environment in England and Wales (EA, 2000b), its Environmental Vision (EA, 2001a), and report on phosphorus and river ecology (Mainstone *et al.*, 2000).

Eutrophication, though, has many consequences, and there exists little information on the environmental and health costs imposed on other sectors and interests in society and the economy at large (Mason, 1996; MAFF, 1998; Pretty, 1998; LWRRDC, 1999; Gaterell and Lester, 2000; EA, 1998, 2000). No national study of the costs arising from the problems caused by eutrophication has yet been undertaken in the UK. In order to inform management decisions and assist in the prioritisation of remedial actions, this research was commissioned to produce an indicative assessment of the environmental costs of cultural eutrophication.

This research was commissioned by the Environment Agency, and comprises a scoping study with three aims:

i. to produce a preliminary estimate of the cost of eutrophication in fresh waters of England and Wales;

ii. to begin to put the costs of eutrophication in perspective with other environmental problems;

iii. to identify areas where costs are currently unknown and further research is required.

A wide range of organisations was consulted and published and unpublished literature accessed. The overarching goal of the project was, therefore, to produce preliminary indicative estimates of the environmental damage costs incurred in England and Wales as a result of eutrophication in fresh waters. Included in the scope of the project are costs associated with and resulting from the adverse effects of eutrophication (see Annex A).

2. Cost Category Framework for Assessing the Costs of Eutrophication

We have developed a new framework of cost categories for assessing the costs of eutrophication, using the Pressure-State-Response framework developed by the OECD as a model (see Figure 1). The pressures driving eutrophication arise from both point and non-point sources of nutrients (Mason, 1996; EA, 2000a; Mainstone *et al.*, 2000). Point sources include:

i) sewage treatment plants (which release nitrogen and phosphorus in the effluent water);
ii) industrial plants (which release biological and chemical wastes containing nutrients);
iii) power plants (which emit atmospheric nitrogen products that are carried in wet and dry precipitation to water bodies).

Non-point or diffuse sources of nutrients include:

i) agriculture (nutrients in leachate, run-off from fields, or gaseous emissions derived from inorganic fertilizers, animal manures and sewage sludge);
ii) aquaculture and fish farming;
iii) forest management (nutrient leaching);
iv) transport sector (atmospheric nitrogen oxide products);
v) rural septic tanks;
vi) natural background sources (including guanotrophy from gull and water fowl roosts).

We distinguish between two major types of cost category (see A and B in Figure 1). These are the damage (or value-loss) costs arising from the reduced value of clean or non nutrient-enriched water, and the policy costs incurred in responding to eutrophication damage plus the costs of changing behaviour and practices to meet legal obligations. Damage costs cannot be added to policy response costs, as the later is a measure of how much is being spent to deal with eutrophication problems.

Damage costs (A), by definition, represent a loss of existing value, rather than an increase in costs, and are divided into two categories: use values and non-use values. Use values are associated with private benefits gained from actual use (or consumption) of ecosystem services, and can include private uses (eg agriculture, industry), recreation benefits (eg fishing, water sports, bird watching), education benefits, general amenity benefits; and option values (the desire of an individual to maintain the choice to use an ecosystem's services in the future)[1].

Use values (social damage costs) can usually be valued by market prices, whereas non-use values are more difficult to calculate. We identify ten types of use value (A1) for water bodies that can be adversely affected by eutrophication (see Figure 1):

i. reduced value of waterside dwellings;
ii. reduced value of water bodies for commercial uses (abstraction, navigation, livestock watering, irrigation and industry);

[1] There are also two types of non-use values: bequest values – those attached to the preservation of an asset (eg habitat or species) so that a future generation may also have an option to use it; and existence values, resulting from an individual's desire to preserve an environmental asset and ensure its continued existence, even though they do not envisage using it (eg the value arising from knowing that particular species or habitats continue to exist).

iii. drinking water treatment costs (treatment and action to remove toxins and algal decomposition products);
iv. drinking water treatment costs (to remove nitrogen for human health and ecological reasons);
v. clean-up costs of waterways (dredging, weed-cutting);
vi. reduced value of non-polluted atmosphere (via greenhouse and acidifying gases);
vii. reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling, and general amenity (picnics, walking, aesthetics);
viii. net economic losses for formal tourist industry;
ix. net economic losses for commercial aquaculture, fisheries, and shell-fisheries;
x. health costs to humans, livestock and pets.

Ecological damage costs (non-use values) (A2) comprise the damage caused to biota and ecosystem structure by nutrient enrichment. These values do not have market prices, and so are difficult to assess. Non-use values include the negative ecological effects on biota (arising from changed nutrient status, pH, and oxygen content of water), resulting in both changed species composition (biological diversity) and loss of key or sensitive species.

The second major category is direct costs arising from a policy response to the problems of eutrophication. When a water body is recognised in some way as being adversely affected by nutrient enrichment, then costs may be incurred through a variety of social responses by both statutory and non-statutory agencies. We divide these direct costs of responding to eutrophication and costs of changing behaviour and practices into two types: compliance control costs, and direct costs incurred by agencies (B1 and B2) (see Figure 1).

There are three types of compliance cost arising from the adverse effects of eutrophication:

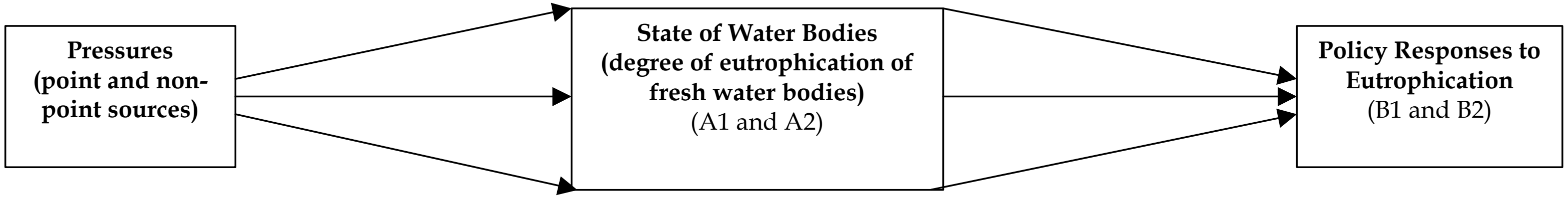
i. sewage treatment costs (to remove phosphorus arising from large point sources);
ii. costs of treatment of algal blooms and in-water preventative measures (biomanipulation, destratification, straw bale deployment etc.);
iii. costs of adopting new farm practices that emit fewer nutrients.

The final cost category comprises the direct costs incurred by statutory agencies for monitoring, investigating and enforcing solutions to eutrophication, and comprise:

i. monitoring costs for water and air;
ii. the cost of developing and implementing eutrophication control policies and strategies.

In addition to costs, there are in some circumstances beneficial effects arising from nutrient enrichment of water bodies. There are three recognised benefits of eutrophication: i) possible increased value of some freshwater and marine fisheries; ii) positive fertilisation effect on farmland through the use of irrigation water enriched with nutrients; and iii) improved sources of food for some wild birds in water bodies. In this study, we focus only on costs.

Figure 1. Cost-Category Framework for Assessing the Costs of Eutrophication



Point sources

i. sewage treatment plants (N and P into water);
ii. industrial plants (biological wastes, N and P);
iii. power plants (atmospheric N products, carried in rain to water bodies).

Non-point sources

i. agriculture (nutrients from inorganic fertilizers, animal wastes, soil erosion); gaseous emissions from soils
ii. aquaculture and fish farming;
iii. forest management (nutrient leaching);
iv. transport (atmospheric N products);
v. rural septic tanks;
vi. natural background sources.

A. Damage costs – the reduced value of clean or non-nutrient-enriched water

A1. Social damage costs

i. reduced value of waterside dweliings;
ii. reduced value of water bodies for commercial uses (abstraction, navigation, livestock watering, irrigation and industry);
iii. drinking water treatment costs (treatment and action to remove algal toxins and algal decomposition products);
iv. drinking water treatment costs (to remove nitrogen for human health and ecological reasons);
v. clean-up costs of waterways (dredging, weed-cutting);
vi. reduced value of non-polluted atmosphere (via greenhouse and acidifying gas emissions);
vii. reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling, and general amenity (picnics, walking, aesthetics);
viii. net economic losses for formal tourist industry;
ix. net economic losses for commercial aquaculture, fisheries, and shell-fisheries;
x. health costs to humans, livestock and pets.

A2. Ecological damage costs

i. negative ecological effects on biota (arising from changed nutrients, pH, oxygen), resulting in changed species composition (biodiversity) and loss of key or sensitive species.

B. Policy response costs – costs of addressing and responding to eutrophication

B1. Compliance control costs arising from adverse effects of nutrient enrichment

i. sewage treatment costs to remove phosphorus from point sources;
ii. costs of treatment of algal blooms and in-water preventative measures (biomanipulation, destratification, straw bale deployment etc.);
iii. costs of adopting new farm practices that leach fewer nutrients.

B2. Direct costs incurred by statutory agencies for monitoring, investigating and enforcing solutions to eutrophication

i. monitoring costs for water and air;
ii. cost of developing and implementing eutrophication control policies and strategies.

C. Benefits of nutrient-enriched (eutrophic) water

i. increased value of freshwater and marine fisheries;
ii. fertilisation effect on farmland;
iii. improved sources of food for wild birds.

3. Summary of Economic Valuation Methodologies

Most economic activities affect the environment, either through the use of natural resources as an input or by using the `clean' environment as a sink for pollution. The costs of using the environment in this way are called externalities, because they are side effects of the economic activity. As they are external to markets, their costs are not part of the prices paid by producers or consumers. When such externalities are not included in prices, they distort the market by encouraging activities that are costly to society even if the private benefits are substantial (Baumol and Oates, 1988; Pearce and Turner, 1990; EEA, 1998; Pretty, 1998; Brouwer, 1999; Pretty *et al.*, 2000, 2001).

An externality is any action that affects the welfare of or opportunities available to an individual or group without direct payment or compensation, and may be positive or negative. The types of externalities encountered in the water sector have four features: i) their costs are often neglected; ii) they often occur with a time lag; iii) they often damage groups whose interests are not well represented; and iv) the identity of the producer or source of the externality is not always known.

Such technological externalities constitute a form of `market failure'. The market `fails' because more pollution occurs than would be the case if the market or other institutions caused polluters to bear the full costs of their actions (Davis and Kamien, 1972; Pretty *et al.*, 2001). It is for these reasons that we need to know more about the cost consequences of certain economic actions. Industries and farmers, for example, allow nutrients to be emitted to water bodies because they do not have to pay the cost of cleaning up the environmental consequences. This implies that rational self-serving behaviour would lead to increased pollution.

In practice, there are very few data on the economic cost of eutrophication (NRA, 1995; LWRRDC, 1999). This is partly because the costs are highly dispersed and affect many sectors of economies. It is also necessary to know about the value of ecosystem services, and what happens when these largely unmarketed goods are damaged or lost. Many studies indicate that current systems of accounting grossly underestimate the current and future value of nature's goods and services (Abramovitz, 1997; Costanza *et al.*, 1997, 1999; Daily, 1997; Heimlich *et al.*, 1998; *Ecological Economics*, 1999; Hodge and McNally, 2000). But such valuation of ecosystem services is vital, though it remains controversial because of the importance these values have in influencing public opinions and policy decisions.

It is relatively easy to assess abatement and treatment costs following pollution, but much more difficult to calculate positive functions. Environmental economists have developed a variety of methods for assessing people's stated preferences for environmental goods through hypothetical markets, which permits an assessment of their willingness-to-pay for ecosystem services or their willingness to accept compensation for losses (Garrod and Willis, 1994, 1995; Bateman *et al.*, 1996; Boon and Howell, 1997; Stewart *et al.*, 1997; Hanley *et al.*, 1998; Langford *et al.*, 1998; Brouwer, 1999).

Economic valuation involves the assignment of monetary values to changes in environmental services and functions and to stocks of environmental assets (Farrow *et al.*, 2000; Pearce and Seccombe-Hett, 2000). These values are effectively prices. Yet most environmental services and assets have no obvious market price (obvious exceptions being coal or oil), and so tend to be undervalued in normal economic transactions.

The rational tendency is for them to be undervalued and overused, as those who cause damage do not pay the full price, but do obtain the benefit. Constanza *et al.* (1997) put it this way: *"if ecosystem services were actually paid for, in terms of their value contribution to the global economy, the global price system would be very different from what it is today. The price of commodities using ecosystem services directly or indirectly would be much greater."*

In this study, we draw upon a wide range of published valuation studies that, in turn, use a variety of methodologies, such as contingent valuation, travel cost methods and hedonic pricing (see Appendix B for summary). It is not the aim of this study to evaluate or compare these methods, nor to address the many methodological debates and controversies.

For a selection of material addressing the values of water ecosystem services and the damage costs, see Green *et al.* (1990), Crutchfield *et al.* (1995), Carpenter *et al.* (1996), Choe *et al*. (1996), Sandstrom (1996), Edwards-Jones *et al.* (1997), Gren *et al.* (1997), Hanley (1997), Bystrom (1998), Pearson (1998), Norberg (1999), Everard (2000), Gaterell and Lister (2000), Keeney and Muller (2000), Matthews and Lave (2000), Söderqvist *et al.* (2000), WRI (2000).

4. Problems in Developing Cost Estimates

Putting a cost on eutrophication as an environmental problem is a complex task for the simple reason that there is no absolute definition of when nutrient enrichment becomes a problem – that is, when it has adverse effects. Algal and higher plant growth is determined by a combination of interdependent hydrochemical, geographic and climatic factors, and so a given level of nutrients in one water body may give rise to adverse effects with associated costs, but in another water body, or the same one at a different time, there may be no effects and thus no costs. Moreover, the threshold at which nutrient enrichment becomes a problem varies. The central problem is the nature of the relationships between nutrient enrichment, the resultant effects, and the costs. These can be difficult to define.

Annex C illustrates six types of potential cost curve, in which axes vary on an arbitrary scale from 0 to 1. The simplest of these is cost curve 1, in which costs vary linearly with increasing nutrients. This is the least likely to be accurate. Cost curve 2 implies that increasing nutrient content incurs no costs until a particular threshold has been passed, following which costs increase linearly. Cost curve 3 implies costs increase more rapidly than nutrients at high levels of enrichment, whereas curves 4 and 5 describe relationships in which costs increase until an asymptote is reached, after which marginal increases in nutrients incur no marginal increase in costs.

Given current knowledge, it is impossible to say which of these relationships pertains in which situation. As indicated earlier, some of the costs of eutrophication arise from social responses to the problem (B1-B2 in Figure 1, Annex C). In other words, they occur because a social response is triggered by some given level of nutrients or their effects (eg algal blooms). Such social responses may differ from place to place and over time.

Thus, for a variety of reasons, the economic data on the costs of nutrient enrichment and eutrophication are limited:

a) there are many different economic valuation methodologies, and these are not necessarily comparable across cost-categories;

b) there are limited data relating to the circumstances in the UK, so some costs are drawn from elsewhere in the world, so can only be used as an illustration of costs (all have been converted into £ sterling);

c) some costs are for a wider problem (eg sewage treatment), of which only a proportion can be allocated to nutrients and eutrophication, and others are imputed from studies that assess the value of water quality improvements;

d) for some cost categories, we have an example of costs but do not know the incidence of the problem per year or geographic extent (eg cost to angling of a fish loss due to toxic algae);

e) for some cost categories, we have no economic data (the studies have not been completed), but do know the extent of the problem, and for others, the costs are known for the whole UK system (eg cost of water treatment).

In this study, we report in detail on the current state of knowledge for each of the fifteen cost categories. We briefly summarise the three potential benefits of eutrophication. For each category, we review the causes of the problem and summarise existing scientific evidence.

We then scope the cost relationship in the form of an equation. Where possible, we then quantify the extent of the problem in the UK (frequency and extent), and summarise the existing economic data on costs and value-losses. Despite the many gaps in the datasets, we are able to summarise the range of costs currently being incurred due to eutrophication in England and Wales.

5. The Environmental Costs of Eutrophication

A. Damage (or value-loss) costs – the reduced value of clean or non-nutrient-enriched water

The calculation of the value-loss costs of eutrophication is fundamentally dependent upon an estimate of the extent and frequency of the eutrophication problem. This could be expressed as the number of days of closure of a water body per season or per year, or the probability of any water body suffering a eutrophication problem leading to value-loss. Though there is no national dataset, we have used the Environment Agency's 1990-1999 national dataset on blue-green algal blooms to estimate frequencies. Blue-green algal blooms are reported centrally, and the dataset for the ten year period 1990-1999 records 3993 reported incidents in 2710 water bodies in the eight water regions of England and Wales (Table 1).

Table 1. Incidence of reported blue-green algal blooms over ten years in eight water regions of England and Wales, 1990-1999 (calculated from Environment Agency dataset)

Region	Number of incidents (1990-99)	Number of water bodies (1990-99)	Incidence of blooms per water body per decade
Anglian	624	440	1.42
Midlands	928	635	1.46
North East	365	241	1.52
North West	486	305	1.59
Southern	95	85	1.12
South West	649	452	1.44
Thames	551	368	1.50
Welsh	295	184	1.60
TOTAL	3993	2710	1.47

The average frequency of a blue-green algal bloom over the ten years is 1.47 per water body, a value that is remarkably consistent across the regions, varying from a low of 1.12 per water-body in Southern region to 1.59-1.60 in North West and Welsh regions[2]. In order to calculate a closure-rate (where the value of the water body is temporarily severely reduced) for water bodies arising from the extreme problem of blue-green algal blooms, we make the following assumptions:

1. All blue-green algal blooms have been recorded (even though we know that the 3993 reported incidents will be an underestimate of the total, as such reporting is reactive and voluntary[3]);

2. Some value-losses or costs will have accrued before the occurrence of a bloom, so the use of this dataset will represent a further underestimate of the problem;

[2] In the USA, surface-water quality failed to meet the goals for swimming quality in 1996 for 20% of rivers, 25% of lakes, and 16% of estuaries (Ribaudo *et al.*, 1999). According to the US National Research Council (1992), 50% of US perennial rivers and streams have fish populations that are adversely affected by turbidity, high temperatures, toxins or low levels of dissolved oxygen.

[3] The EA has a reactive sampling strategy whereby water-bodies are sampled in response to external enquiries and internal identification. Sites with a history of algal blooms are generally not revisited but instead caution is advised based on the probability of toxicity.

3. The frequency of occurrence is a record only for those water bodies with at least one recorded incident. The EA estimates that there are some 6000 lakes and reservoirs greater than one hectare in size in England and Wales, and so the frequency used here may overstate the problem as many water bodies have had no recorded blooms.

4. We assume that confirmed blue-green algal blooms lead to the closure of the water body (ie a restriction in use and some access) for between 5 and 30 days. Based on an understanding of these blooms, we judge that 25% of incidents cause closure for 30 days, 50% cause closure for 15 days, and 25% for just 5 days each. Thus on average, each incident causes a closure for 16.25 days. For severe toxic blooms, this could be an underestimate, as the effects can persist for many weeks. For other blooms, this could overestimate the losses in value.

5. We calculate the frequency of closure using two denominators - the summer seasons of six months, and the whole year. Most uses of waterbodies and watercourses occur during the summer months, though there are notable exceptions (eg angling, rambling, potable water supply), and most blooms occur in the summer months.

6. We do not differentiate the data according to geographic location in the UK. This may be important, as significant numbers of access points to water courses have precautionary notices concerning blue-green algae, which prevent activities from taking place.

We describe the equation for the relationship as:

$$f_c = (I_{bg} \times N)/(C \times (S_{½} \text{ or } S_1) \times Y) \qquad \text{(equation 1)}$$

where:

f_c = frequency of closure
I_{bg} = number of incidents of blue-green algal blooms
C = number of water bodies affected
N = number of days water body closed for each incident
$S_{½}$ = season length (days in half year)
S_1 = season length (days in full year)
Y = number of years of data

Thus for a half-year season ($S_{½}$), f_c = 0.0131 or 1.31%, and for a full-year season (S_1), f_c = 0.0066 or 0.66%. We use the range 0.66-1.31% as the closure rate for water bodies and courses due to blue-green algal blooms to calculate value losses[4]. Thus the probability of a water body having to be closed on any given day is between 1 in 76 and 1 in 151[5].

[4] These incidence rates are low compared with those in Australia (New South Wales and Victoria), where weekly algal alerts in water bodies occurred 46% of the time between 1995-98, rising in some water bodies to 89% (LWRRDC, 1999). See also DNRE, 2000. This high incidence is at least partly explained by higher light levels facilitating blooms.

[5] There is no evidence in the dataset for any significant changes year on year, and so it is impossible to say whether recent policies are having a desired effect on algal bloom incidence rates.

A1. Social damage costs

A1i. Reduced value of waterside dweliings

Water quality affects the value of property adjacent to or in the immediate vicinity of a water body or water course. Residential properties with a water front, for example, generally have a higher value than equivalent properties without a water front, and this added value is estimated to be 0-15% (mean nearer 0%) for offices, 0-25% (mean nearer 10%) for leisure developments, and 10-40% (mean nearer 20%) for residential properties in the UK (Wood and Handley, 1999)[6]. But waterfront properties can lose value if the quality of the water falls, particularly if this is manifested as an increase in turbidity, algal blooms and unpleasant odours.

In the UK, there have been no national studies of value-loss in waterfront properties affected by eutrophication[7]. One regional study in the Mersey Basin has, however, found that leisure and residential property can be devalued by as much as 20% as a result of consistently poor physical water quality (Wood and Handley, 1999). Such a reduction entirely offsets any added value that properties receive for being located at the waterside.

Studies in the USA, Brazil and Australia also indicate that the losses arising from periodic and/or continuing eutrophication can be significant. In Maine, lakeside property prices fell by £243-4430 (US$340-6200) per 10 metre of shoreline frontage for each metre loss in water clarity. On Chesapeake Bay, the benefits of improving water quality, through reductions in nutrients, coliform bacteria and toxic contaminants, have been found to be £17,000 ($24,000) for each of the 494 waterside properties studied. In Brazil, the economic impact of eutrophication was estimated to result in a 50% devaluation in the price of properties, with an additional loss of recreational value of the water from bad odour (Michael *et al.*, 1996, 2000; UNEP, 1999; Legget and Bockstael, 2000).

In Australia, property values for waterside properties around Lake Boga fell following major algal blooms in the summers of 1993-4 and 1994-5. When properties were re-valued in 1995, the valuer concluded that lakeside properties were down-valued by 20-25% on average - equivalent to an annual loss of approximately 2% of property values (i.e. the cost per year in each year subsequent to a major bloom calculated at a discount rate of 8% over a period of 30 years) (DNRE, 2000). At another location, lakeside properties at Lake Colac were reported by real estate agents to be nearly `impossible to sell' after the algal blooms of 1993-4. Although the situation has since recovered, it was concluded that in the year immediately after a bloom properties were difficult to sell. This has been calculated as equivalent to 1% of property value loss (based on the interest forgone by delaying the sale of a house by one year and assuming that, without a bloom, houses would be sold on average every 10 years) (DNRE, 2000).

In order to calculate the potential effects of eutrophication on property values, data in needed on the length of rivers and still freshwater frontage affected by eutrophication, and the number of properties affected. Under the EC Urban Waste Water Treatment Directive, 2540 km of rivers and canals have been designated as Sensitive Areas (Eutrophic). As the

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[6] Other estimates vary between 5% (RPA, 1997) to 18.6 (Royal Institute of Chartered Surveyors (Jonathan Fisher, pers. comm.). Further estimates provided by Grahame Newman, British Waterways, per. comm. 28th March 2001, Glen Miller, British Waterways, pers. comm. 5th April 2001

[7] A University of Newcastle Scoping Study on property prices and eutrophication, commissioned by British Waterways, is due to be completed in 2001-2002.

quality of 40,000 km of rivers is assessed by the EA for chemical and biological quality, this equates to 6.35% of assessed rivers being deemed eutrophic. Under the General Quality Assessment scheme, waters classified as grades 4 and above (>0.1 mg P/l) exceed the guideline value for eutrophic rivers in implementing the Directive. For 1993-95, 51.6% of rivers were in grades 4 and above. Thus the proportion of rivers deemed eutrophic ranges from 6.35% to 51.6%. There are also 6300 standing waters in England and Wales with a size greater than one hectare.

We use a value of 10% for the loss in value per property, an average waterside property value of £100,000, and assume there are 75,000 waterfront properties exposed (based on an average density of 121 dwellings km^{-1} on built-up roads, assuming a density of half on waterfronts, and half of the 2540 km of sensitive watercourses as built-up; and with a cross-check on the number of properties now located in flood-prone areas (150,000 in the 1990s) (DLTR, 2002).

The value-loss relationship is: $VL_{A1i} = P_n \times f_c \times Vl_p$ = £9.83 million yr^{-1} (equation 2), where:
VL_{A1i} = total value-loss for waterside properties in England and Wales;
P_n = number of waterside properties;
f_c = frequency of loss of value due to some eutrophication;
VL_p = value-loss (£) per average 10 metre frontage.

A1ii. Reduced value of water bodies for abstraction, livestock watering, navigation, irrigation and industrial uses

In addition to recreational uses, water bodies and wetlands have a wide variety of industrial uses. These include direct use of clean water as an input to manufacturing, electricity generation, and farming for livestock watering and irrigation; the use of waterways for navigation and transport; and the value of wetlands and water bodies for waste treatment and attenuation.

Costs are therefore increased when nutrient enrichment reduces the value of clean water, and when the biomass of aquatic algae and macrophytes increases so that waterways are impeded for navigation. Wetlands also provide a very important function in waste attenuation, providing this service for free and so saving on investment in industrial plant. The total value of wetlands for water regulation and supply, flood protection, waste treatments, wildlife habitats, recreation and raw materials has been put at £9850 ha^{-1} yr^{-1}, varying from £5660 ha^{-1} yr^{-1} for lakes and rivers, £6660 ha^{-1} yr^{-1} for tidal marshes and mangroves, and £13,050 ha^{-1} yr^{-1} for swamps and floodplains (see Costanza *et al.*, 1997: Heimlich *et al.*, 1998; *Ecological Economics*, 1999)[8]. Once water bodies are eutrophic, they may be less able to perform these functions effectively (though much will depend on the specific circumstances, as some wetlands are naturally eutrophic and others naturally oligotrophic).

The value-loss relationship for this category is:

$VL_{A1ii} = V_w \times f_c$ (equation 3)

where

[8] The Costanza *et al.* (1997) article on the value of nature's goods and services has created some controversy. For a full summary of the pros and cons, see the whole issue of *Ecological Economics* (1999), with 11 responses (Ayres; Daly; El Serafy; Herendeen; Hueting *et al.*; Norgaard and Bode; Opschoor; Pimentel; Rees; Templet; Toman; and Turner *et al.*), and a reply from Costanza *et al.*

VL_{A1ii} = reduced value of water bodies for abstraction, livestock watering, navigation, irrigation and industrial uses
V_w = value of water for industrial, farming and navigation uses
f_c = frequency of closure (prevention of use of water for demand use)

However, there are no national datasets to calculate V_w, though case studies do indicate the importance of the problem. One study of Loch Leven in the summer of 1992 assessed the costs arising from an algal bloom arising from the release of P from the sea loch's sediments (Forth River Purification Board 1993). It was estimated that the revenue loss sustained in the summer by the waterside industries was £160,000 (one of the three paper mills had to stop production for a day due to blocked intake filters etc) (D'Arcy *et al.*, 2000).

An example of the adverse effects of spray irrigation contaminated with blue-green algae occurred in July 1998 at Hall Lane Nurseries in Preston. *Microcystis* colonies were detected on the leaf surface of lettuces consistent with plants having been watered by spray irrigation from a water source containing *Microsystis* scum. On the advice of the Environment Agency and MAFF the lettuce crop was voluntarily withdrawn from sale. In Australia, the annual eutrophication costs associated with abstraction, livestock watering, navigation and industrial use amount to £0.88 million (Aus$2.5 m). The total annual cost to irrigators in the State of Victoria is some £54,000 (Aus$152,000).

British Waterways are responsible for 2,000 miles of navigable rivers and canals in the UK, this represents about half of all inland navigations in Britain (British Waterways, 2001). Also The Association of Inland Navigation Authorities is a membership body comprising 27 navigation authorities in the UK who own, operate and manage some 5,000 km of inland waterways[9]. There are, however, no data on income from navigation for these bodies from which estimates of losses arising from eutrophication could be derived.

A proxy for the value of water abstraction can be derived from the amount of actual abstraction and the charges made for licenses (even though such licenses are a weak indicator of the value of water, which is best measured by what people are willing to pay for it) (Table 2). The total income for abstraction licenses is £6.67 million yr^{-1}, and using equation 3, this suggests a loss of £0.094-0.190 million per year (though it is well recognised in the water industry that abstraction charges significantly underestimate the value of water abstraction). But as shown in the Loch Leven case study above, the loss to one enterprise from a single incident was £0.16 million. If there were only two major incidents of this type per year (according to the EA dataset, there have been an average 400 blue-green algal blooms yr^{-1} over the ten years 1990-99), then some £0.50 million costs yr^{-1} for this category are likely to have occurred. Given the paucity of data on losses to abstraction, livestock watering, navigation, irrigation and industry, this is clearly an area requiring further research. In this study, we adopt a low estimate of £0.50-1.00 million yr^{-1}.

Table 2. Annual water abstractions in the UK and charges for licenses

Activity	Abstractions (million Ml per year)	median charge (£/Ml)	Total charge per year (£ million)
Spray irrigation	0.024	16.00	0.39
Public water supply	0.843	5.80	4.89
Industry	0.277	5.00	1.39
TOTAL			£6.67 million

[9] Revenue derived from transportation by freshwater (all types of goods but excluding taxes) in the USA total £257 ($360) billion yr^{-1}. In Europe, these arte estimated to be £120 ($169) billion yr^{-1} (Postel and Carpenter, 1997).

Source: DETR, 1999a

A1iii. Drinking water treatment costs (treatments and actions to remove algal toxins and algal decomposition products)

Nutrient enrichment and algal blooms incur significant costs for water supply and sewerage treatment operators. Some of these costs are to meet compliances established in national and European regulations, especially for nutrient concentrations (see B2ii), while others relate to the adverse effects of algal blooms and their decomposition products. In reservoirs, the effects of eutrophication can be costly, particularly if they mean the closure of treatment plants. In the process of water purification, filtration and straining measures can cope with large numbers of small algae, but can become blocked when large algae are present, so reducing the effectiveness for water treatment. When purification has to be stopped for filter cleaning, supply problems can occur, with consequent higher costs for water companies and receiving households and/or shareholders of water companies.

If small algae pass through the filters, they can decompose, so causing bacteria to develop, which in turn encourages fungi and invertebrates to feed on them within the pipe network. As a result, tap water can contain not only bacteria but fauna too, which when combined with turbid appearance, is unsatisfactory for water customers (Mason, 1996). Changes in the oxygen content and pH due to eutrophication and the products from the decomposing algae can also affect the chemical processes and reactions of water treatment. This can result in potentially harmful chemical products being present in drinking water, together with deposits in the pipe networks, both leading to unpleasant tastes and odours. Examples of these compounds include aluminium, iron, manganese (which affects the colour) and chloroform (Harper, 1992). In addition, removal of hepatotoxins (produced by cyanobacteria) from reservoirs is difficult as some forms are stable and resistant to chemical hydrolysis or oxidation and may persist for months or years and remain potent even after boiling (UNEP, 1999).

The damage cost relationship for this category is:

$$DC_{A1iii} = (C_o \times A_p \times ASP_o) + (C_c \times A_p \times ASP_c) + C_r \qquad \text{(equation 4)}$$

where

DC_{A1iii} = drinking water treatment costs (treatments and actions to remove toxins and algal decomposition products)
C_o = annual operating expenditure by water companies
C_c = annual capital expenditure by water companies
A_p = proportion of production liable to suffer from algal proliferation
ASP_o = proportion of Algae Sensitive Production (ASP) operating costs for eutrophication
ASP_c = proportion of ASP capital costs for eutrophication
C_r = annual cost of reservoir management systems

We assume that 10% of the direct operating costs and 5% of the capital costs for ASP arise from eutrophication. The best information available on operating costs for water treatment is derived from the government's Office of the Director General of Water Services company returns, which give the direct operating costs for England and Wales against `water resources and treatment' as £284 million yr^{-1}. Thus, the additional treatment costs are £284 m x 0.33 x 0.1 = £9.5 m yr^{-1}. Capital expenditure (Capex) on water treatment has declined over the last two asset-management planning periods to £332.9 million. Assuming that this level of Capex still pertains, this yields additional expenditure as £332.9 m x 0.33 x 0.05 = £5.55 m yr^{-1}. The combined capital and operating costs of reservoir systems to prevent proliferation of algae and development of anaerobic conditions is £4 million yr^{-1} (RCEP, 1996; D'Arcy *et al.*, 2000). Thus DC_{A1iii} = £19 million yr^{-1}.

A1iv. Drinking water treatment costs (to remove nitrogen)

Costs are incurred by water supply companies to comply with drinking water standards set out in EU legislation for pesticides and nitrates (the maximum for nitrate is 50 mg nitrate l^{-1} or 11.3 mg nitrate-N l^{-1}), to remove pathogens, particularly *Cryptosporidium*, to pay for restoring water courses following pollution incidents and algal blooms, and to remove soil from water. Companies incur both capital and operating expenditure for water quality treatment. These costs are reported annually by each of the 28 water companies in England and Wales to the government's Office of the Director General of Water Services (Ofwat)[10]. We assume the cost of compliance reflects extent of nitrogen enrichment, as treatment costs are highest for water companies in regions with greatest nitrogen loads to water.

We describe the damage costs for this category as

$$CC_{A1iv} = NC_o + NC_c \qquad \text{(equation 5)}$$

where

CC_{A1iv} = drinking water treatment costs (to remove nitrate)
NC_o = annual operating costs of removal of nitrate by water companies
NC_c = annual capital costs of removal of nitrate by water companies

We calculate from Ofwat returns for 1992-97 that the 28 water companies of England and Wales (of which 10 are water and sewerage companies, and 18 are water-only companies) expended £20.1 million per year on nitrate removal to meet compliances (see Pretty *et al.*, 2000). The total UK cost of achieving the nitrate standard for potable water has been estimated at £199 million over the next 20 years. Anglian Water, the most affected water utility estimated their costs would be £70 million over the next 10 years (D'Arcy *et al.* 2000)[11]. It could be argued, however, that nitrate removal from drinking water supplies is solely for reasons of human health and not eutrophication control. But it is also true that if nitrate were not removed, then there would be higher levels of nutrient in waters, and this would have an effect on eutrophication.

Thus CC_{A1iv} = £20.1 million.

[10] The government's Office of the Director General of Water Services sets industry price levels each five years, which determine both the maximum levels of water bills and specifies investments in water quality treatment. During the 1990s, water industry undertook pesticide and nitrate removal schemes, resulting in the construction of 120 plants for pesticide removal and 30 for nitrate removal (Ofwat, 1998). Ofwat estimates that water companies will spend a further £600 million between 2000-2005 on capital expenditure alone due to continuing deterioration of `raw water' quality due to all factors. Although Ofwat has sought to standardise reporting, individual companies report water treatment costs in different ways. Most do distinguish treatment for pesticides, nitrate, *Cryptosporidium*, and several metals (iron, manganese and lead). The remaining treatment costs for soil removal, arsenic and other metals, appear under a category labelled `other'. Of the 28 water companies in England and Wales, 3 report no expenditure on treatment whatsoever; and a further 3 do not disaggregate treatment costs, with all appearing under `other'. 20 companies report expenditure on removal of pesticides, 11 on nitrates, and 10 on *Cryptosporidium*. It is impossible to tell from the records whether a stated zero expenditure is actually zero, or whether this has been placed in the `other' category. Using Ofwat and water companies' returns, we estimate that 50% of expenditure under the `other' category refers to removal of nutrients. See also Ofwat, 2000a, 2000b.

[11] Water derived from river intakes in lowland England frequently has nitrate levels exceeding the maximum recommended level, and so has to be mixed with water low in nitrate before in enters the public supply (Mason, 1996).

A1v. Clean-up costs of waterways (dredging, weed-cutting)

A further cost is incurred by organisations with responsibility for keeping waterways clear of weeds. Both the Environment Agency and British Waterways (the owner of most canals in the UK) incur costs through twice-yearly clearance of weeds in eutrophic waters. Some of this weed cutting is primarily for flood control. The Environment Agency's policy is to maintain flood defence and preserve channel capacity through routine maintenance schedules for weed cutting of all submerged and emergent vegetation on river beds and lower portions of banks, and by desilting in culverts, clearance of weed screens at sluices, and vegetation clearance on river banks. It is almost impossible to separate the cost of dredging and weed-cutting due to eutrophication effects from the overall annual costs.

Once again, there are no national datasets for these costs, and we must rely on case study data. A EA recent review of weed cutting costs in the Rivers Sewer, Torne, Lidsey Rife, Mar Dyke, Cam, Sence, Soar, Alt, Wreake, and Glen put the annual cost at £286,000 for a total river length of 285 km (approximately £1000 per km)[12]. In the River Adur catchment, the total annual costs for the EA amount to £103,300, of which 75% is for physical infrastructure and maintenance, the remaining £25,600 relating to nutrient enrichment.

Eutrophication has led to problems for British Waterways. The excessive duckweed (*Lemna sp*) growth in the Oxford canal has completely covered the surface and has been a problem since 1998. This has resulted in the need for mechanical removal and chemical treatment, costing £50,000 per year[13]. The high cost is due to the intensive labour required and the length of canal affected - some 26km[14]. A much greater cost has occurred at Barton Broad on the River Ant, where restoration has included a £2.4 million project to dredge the nutrient-rich mud from the whole of the broad. This mud is being disposed to nearby arable land.

In the Chesterfield Canal, where the weed problems caused by nutrients from STWs, the costs identified for British Waterways are approximately £20,000 of annual operating costs for the weed-cutting boat, plus the capital cost for the boat of £42,800[15]. Another case study of Bosherton Lakes SSSI in Wales showed that decline in aquatic plants and species reduction was due to nutrient enrichment from receiving sewage effluent (Moss *et al.,* 1996). There were difficulties with remediation due to the sensitive conservation nature of site. Nutrient rich water was redirected to the sea using a plastic water pipe sleeved with concrete weights (cost of pipe £102,000) and removal of pondweed and weed cutting happen yearly at a cost of £1875 per year.

Bourne Mill Pond in Colchester is a National Trust owned property, and is one of a series of historic ponds that served mills in the past. It was last dug out in 1956 and the sediments were spread on local arable land. In 2000 a bio-assessment on the pond was carried out after letters were received from neighbouring residents and the local fishing club about the quality of the water. Bourne Pond was found to be shallow and subject to algal blooms due to increased siltation after flash flooding events and increased run-off from roads. The cost to NT of suction dredging was £130,683 and professional fees of £6,692. A new beam for emergent vegetation and a silt trap have also been created to try to reduce the frequency of

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[12] Rachael Dils, Environment Agency, pers. comm.. 5th November 2001

[13] Andrew Grimmett British Waterways pers. comm. 21st March 2001

[14] Matthew Saxon, British Waterways pers. comm. 21st March 2001

[15] Grahame Newman, British Waterways pers. comm. 28th March 2001

de-silting on such a large scale in the future[16]. Not all these costs, though, can be attributed to eutrophication.

The damage cost relationship for this category is:

$$DC_{A1v} = (\Sigma Wc_{i\text{-}j}) \times P \qquad \text{(equation 6)}$$

where

DC_{A1v} = clean-up costs of waterways (dredging, weed-cutting)
ΣWc = sum of cost of weed cutting for organisations i to j due to eutrophication
P = proportion of weed cutting that can be attributed to eutrophication

As there are only limited data to complete each component of this equation, we estimate average annual costs in the range of £500,000 to £1,000,000.

A1vi. Reduced value of non-polluted atmosphere (via greenhouse and acidifying gas emissions)

An important value-loss cost of eutrophication arises from the emissions of two greenhouse gases, nitrous oxide (N_2O) and methane (CH_4), and the gas, ammonia (NH_3). Microflora in wetlands and water courses produce ammonia, nitrogen gas and nitrogen oxides. N_2O is a greenhouse gas, contributing to atmospheric warming, as well as having a damaging effect on stratospheric ozone. Methane is emitted from watercourses where severe macrophyte growth leads to build up of large quantities of organic detritus, causing anoxia and gaseous emissions. Methane and nitrous oxide impose costs on the environment by contributing to climate change and to localised acidification (Conway and Pretty, 1991; Pearce *et al.*, 1996; IPCC, 2001).

The value-loss relationship is:

$$VL_{A1vi} = (E_{CH4} \times P_w \times C_{CH4}) + (E_{N2O} \times P_w \times C_{N2O}) + (E_{NH3} \times P_w \times C_{NH3}) \qquad \text{(equation 7)}$$

where

VL_{A1vi} = reduced value of non-polluted atmosphere
E = annual emissions of N_2O, CH_4, and NH_3 in tonnes
P_w = proportion of emission arising from waterbodies and watercourses
C = environmental cost per tonne of each gas (N_2O, CH_4, andNH_3)

Gaseous emissions are recorded in the UK National Atmospheric Emissions Inventory, DEFRA database, and the European Environment Agency inventory (DETR, 1999b; EEA, 1999). The total emissions of methane are 3.7 million tonnes per year. The main sources are landfill sites (46%), farm livestock (29%), and venting from coal-mines and gas leakage (21%). In addition, methane is released during waste treatment and disposal, and directly from wetlands. Approximately 1-2% of national emissions arise from waste disposal and sewage treatment works, putting annual emissions at between 37,000 and 74,000 tonnes.

The total emissions of nitrous oxide in the UK are 189,000 tonnes per year, of which 52% is from fertilizers and animal wastes, 37% from industrial combustion, and 5% from road transport. A small amount, some 200 tonnes, is released from watercourses during waste

[16] Keith Turner, National Trust pers. comm. 25th April 2001

treatment and nitrogen conversion. Ammonia can also be released from waterbodies rich in nutrients. Annual emissions amount to 320,000 tonnes, of which 88% is from livestock and fertilizers. We assume that a quarter of the remainder arises from waterbodies – some 9600 tonnes.

A variety of economic studies have sought to put a per tonne external cost on these gases (Pearce *et al*., 1996; Eyre *et al*., 1997; Holland *et al*., 1999). These studies on the external effects of climate change gases include analysis of impacts on climate change, health, parasitic and vector borne diseases, sea level rise, water availability, biodiversity, and storm, flood and drought incidence (Eyre *et al*., 1997). The data in the Open Framework and FUND models take account of differences in discount rate, are weighted according to wealth differences in affected countries, and take account of `social contingency' (the capacity of regions/countries to adapt to change).

Though uncertainty is still large, we use the Hartridge and Pearce (2001) analysis of marginal costs: for CH_4 these are £77.9 tonne^{-1}, for N_2O £2961 t^{-1}, and for NH_3 £171 tonne^{-1} (by comparison, the damage cost of carbon dioxide (as C) is £29.8 t^{-1}). Thus, using equation 7, the value-loss costs for this category are £5.12-7.99 million yr^{-1}.

A1vii. Reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling and general amenity (picnics, walking, aesthetics)

Many standing and running fresh and marine water bodies are used extensively for recreational and amenity purposes on the water, such as bathing, boating, windsurfing and canoeing, and for amenity at the waterside, such as angling, dog-walking, rambling and picnics. Eutrophication results in a loss of recreational and amenity value, particularly if water becomes turbid, emits unpleasant odours, and is affected by algal blooms. Such blooms may be simply unpleasant, with green slimy margins to the water, or toxic if blue-green algae are present (Pearson, 1996). But such blue-green algal blooms do not affect all recreational users in the same way. At high risk of harm are those engaged in swimming, diving, wind-surfing and water-skiing. At medium risk are canoeists, sailors and walkers, and at low risk would be those engaged in boating and pleasure cruising (some of whom may not even notice the presence of a blue-green algal bloom).

Value-loss costs are, therefore, incurred when people are prevented by eutrophication and algal blooms from enjoying the quality of a water body. Those people whose livelihoods rely on visitors who would otherwise have used the `clean' water body, such as instructors, boat-owners, and hotel-owners, suffer additional costs through reduced visitor expenditure (see category A1vii). It should be noted that there is the possibility of some double-counting, as value-losses for recreational and amenity uses could also affect house prices (category A1i).

There is no national database recording how eutrophication affects the recreational and amenity value of water bodies. Only study using contingent valuation methods calculated that the annual costs of the loss of recreational facilities at Rutland Water owing to blue-green algal blooms was £0.5 million (Pearson, 1998). We thus rely on data from 37 studies (mainly in the UK, USA, Canada and Australia) to allocate value-loss costs per visitor-trip. A range of valuation studies using a variety of methods (contingent valuation, willingness to pay and willingness to accept, benefits-transfer, travel costs, and meta-analysis) have been conducted by economists to put a value on the recreational uses of water bodies (see Annex D for summary of 37 studies).

As can be seen from Annex D, data is represented either as values per person or household per visit or trip, or per household per year. In order to calculate the costs of eutrophication in this cost category, we use values per person per year to estimate the benefit derived from water courses by visitors in the UK. This is technically termed the consumer surplus, the individual willingness to pay, and is mostly in the range £8-20 per person per visit (most of the values greater than £20 per day are from North America).

The upper limit of £20 per person is consistent with a recent review drawing on water and wetland studies across Europe (UK, Norway, Austria and Sweden), which puts the average value for water courses at £20 per person per year, and for wetlands at £24 per person per year (ten Brink *et al.*, 2000). In this study, we adopt a conservative range of £8 to £14 per person per year, noting also that there may be displacement when a toxic bloom closes one area, with users simply moving to another site. This depends on availability of substitutes, and whether saturation of sites has occurred.

We describe the value-loss relationship for this category as:

$$VL_{A1vii} = N_v \times f_c \times C_s \qquad \text{(equation 8)}$$

where:

VL_{A1vii} = reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling and general amenity (picnics, walking, aesthetics);
N_v = number of day and tourist-day visits to water bodies made each year;
f_c = frequency of closure (% of days);
C_s = consumer surplus (£ per day) for use of water-based ecosystem services.

We use the Countryside Agency (2001) and English Tourism Council (ETC, 2000) data derived from the UK Leisure Day Visits Survey and UK tourism surveys to calculate the number of visits made to the countryside and seaside for water-based leisure and recreational activities. The expenditure made by these visitors and tourists is not used in this category, as it appears in category A1vii. In 1998, some 1.261 billion tourist-day visits were made, of which 72% were to towns, 6% to the seaside, and 22% to the countryside. In addition to day visitors, a further 172 million tourist trips are taken by UK and overseas residents, in which one or more nights are spent away (the majority for 1-3 nights each). The total number of days spent is 707 million yr^{-1}.

Assuming that the same proportion of tourist trips are to town, country and seaside, this suggests that of the total 1.968 billion day and tourist-days in the UK, some 551 million per year are for countryside (433 million) and seaside activities (118 million). Not all of these, though, are for water-based activities. We use the Countryside Agency (2001) surveys to allocate proportions of countryside days and tourist-days to activities (Table 3).

Table 3. Breakdown of leisure activities in UK countryside, and days spent on each activity, 1998 (from Countryside Agency, 2001; ETC, 2000)

Activity	Proportion of day and tourist-day visits on each activity (%)	Ratio of activity that is water-based	Number of days per year (million) on each activity (assuming a total of 433 million)
Hiking, walking and rambling	19%	0.33	27.2
Swimming (of which 31% is outdoors)	16%	0.31	21.5
Heritage attractions	13%	0.50	28.2
Cycling	6%	0.33	8.6
Sailing	5%	1.00	21.7
Fishing/angling	5%	1.00	21.7
Fields and nature	4%	0.50	8.7
Sport and leisure	20%	0.20	17.3
Pony trekking, riding, shooting, hunting, mountaineering, rock climbing, theme parks	14%	0.05	3.0
Additional angling adjustment (using rod-licenses and participation rate data)	-	-	25.0
TOTAL			182.9 million days

Note: ratio of activities that are water based are authors' own estimates

We make one adjustment to this data for angling participation rates, owing to the relatively good quality of data. According to Environment Agency records (EA, 2001b), there are 1.14 million angling licences issued each year for coarse, trout and salmon fishing. Using the estimate that each angler fishes for 43 days per year, together with the fact that many licenses are only for 1 or 8 days (810,000 licenses are full and concession, 36,000 are eight-day, and 264,000 are one-day), we estimate that there are some 35.4 million angling days per year in England and Wales.

This, though, will underestimate the total, as some anglers are neither members of clubs nor pay for licences. Nonetheless, this may still be an underestimate, as the proportion of households with one or more anglers varies from 9% in the Thames region, to 10-12% in the midlands and north, to 15-16% in East Anglia and the southern England (NRA, 1995). Across Britain, 12% of households have one or more anglers. Thus given total of 20 million households, there are 2.4 million anglers making 43 day visits per year, giving a total of 103.2 million (see Table 3). To calculate the losses, we use the frequency of closure derived from the blue-green algal blooms database[17]. There is thus a further research need to cross-check closure rates with water sports associations and water companies.

To summarise, the participation data indicates that 182.9 million days were spent in inland water-based leisure and recreational activities in 1998, with a further 118 million spent at the seaside (total 301 million days). As this study assesses only eutrophication in fresh waters, we use the figure of 182.9 million days.

Thus $VL_{A1vii} = N_v \times f_c \times C_s$

$= 182.9 \text{ million} \times (0.0066 \text{ to } 0.0131) \times (£8 \text{ to } 14)$

[17] This probably underestimates the costs to anglers, as excessive weed growth in nutrient-enriched waters makes angling difficult or impossible. These waters may not be fished at all, whether or not algal blooms are present. Moreover, the contribution to anglers to economic activity is significant. According to the Salmon and Freshwater Fisheries Review by MAFF (2000), there are 12,000 full time equivalent jobs dependent on fishing tackle sales, and turnover of companies involved in tackle, bait and magazines is £380 million yr^{-1}. The total spent by anglers on their sport is between £3.3 and £5 billion yr^{-1}.

= £9.65 to £33.54 million.

A1viii. Net economic losses for formal tourist industry (inland and coastal)

This cost category refers to the direct revenue losses in the tourist industry arising from closures and restrictions on water courses caused by eutrophication and algal blooms. When visitors access water courses for the recreational purposes listed in A1ii, they also spend money for accommodation, food and other goods and services. When eutrophication prevents access, then this revenue is lost. This may mean losses for potential sites or locations, though others may gain if recreationalists go elsewhere to spend their money.

Once again, there are no national studies of these costs, but case studies indicate that the cost can be substantial. For example, the algal blooms in Loch Leven over the summer season of 1992 were estimated to have caused losses in Kinross shops, hotels, bed and breakfasts of £673,000 (Forth River Purification Board, 1993; D'Arcy *et al.*, 2000). However, it is important to note that this problem has only occurred on one occasion in one year. In Australia, the tourist industry loses £1.56 (Aus$4.44) million per year in the Victorian catchments, and £3.3 (Aus$9.4) million in the Barwon-Darling catchment of New South Wales due to algal bloom closures (Walker and Greer, 1992)[18].

We once again use Countryside Agency and English Tourism Council data on day and tourist-day visits, combined with expenditure per visit data, to calculate the total value of water-based expenditures. Two different measures are important in this category - the net economic value of tourism, and the value of the total economic activity. The net economic value of tourism and day visit expenditure is important, as it refers to the profits created by these activities. An expenditure of £100 on a meal in a waterside restaurant that cost £80 to prepare results in a net economic value (economic rent) of £20, as it is assumed that customers would have substituted their expenditure for something else had the restaurant been closed.

Nonetheless, avoiding the loss of economic activity is important for tourism and recreational users of water bodies. People spend money on goods and services as they use a water body, and by spending this money, jobs and infrastructure may be created and further spending induced by those who sell the goods and services. Thus the £80 spent in the restaurant covering costs helps to support cooks and waiters employed by the business, as well as local food suppliers. Such activity is technically not the value of the water body, but is brought into existence by the value of the water body. Loss of expenditure and jobs due to closure of a water body because of an algal bloom represents a local cost, but at the national level may simply mean recreationalists go elsewhere to spend their money. Measuring the loss of economic activity thus represents the local losses of income, but may not aggregate well at national level.

Given the scope of this study, it is impossible to assess the distributional and/or displacement effects of eutrophication and its consequences on tourism spending. These could be significant if poorer areas of England and Wales heavily dependent on tourism were to suffer disproportionately more through expenditure losses arising from

[18] One study in Brazil of a lake that had been kept stable and non-eutrophic for 25 years indicated that it had stimulated economic growth in tourism. An area of 7 km^2 has stimulated an investment of £179 (US$ 250) million in 25 years, a value of prevention rather than remediation (UNEP, 1999). It is impossible to say, though, whether such investment would have occurred anyway had the lake been eutrophic.

eutrophication (eg in the Norfolk Broads or Cumbria). This is an area requiring further research.

We describe the value-loss relationship for this category as:

$$VL_{A1viii\ TOTAL} = N_v \times f_c \times E_{day}$$
$$VL_{A1viii\ NET} = N_v \times f_c \times E_{dayP} \qquad \text{(equation 9)}$$

where

VL_{A1viii} = revenue losses for formal tourist industry (inland and coastal)
N_v = number of day and tourist-day visits to water bodies made each year
f_c = frequency of closure (% of days)
E_{day} = total expenditure per day and tourist-day visit
E_{dayP} = local profit arising from total expenditure per day and tourist-day visit (net economic value)

We use the same data for day and tourist-day visits to water-based activities as in category A1vi. However, as daily expenditure varies considerably according to whether individuals are UK residents, overseas tourists, or UK day-visitors, we disaggregate the data in Table 4 to calculate the annual total spent on freshwater-based days to be £4.45 billion. The total value of economic activity lost to eutrophication, $VL_{A1viii\ TOTAL}$ = £4.45 billion x (0.0066-0.0131) = £29.4-58.3 million yr^{-1}. In the service sector, profits are in the range of 10-20%, and so the net economic value lost, $VL_{A1viii\ NET}$ = £4.45 billion x (0.0066-0.0131) x (0.1-0.2) = £2.94-11.66 million yr^{-1}. We use net value for the losses in this study, even though there may not be perfect displacement, with local losses compensated for by gains in spending elsewhere in the economy.

Table 4. Expenditure by visitors to water-based activities in UK (adapted from Countryside Agency, 2001; ETC, 2000)

	Over-night tourism visits		**UK day visitors**	**Totals**
	UK	Overseas		
Total visit-days (million)	495	212	1261	1968
of which to countryside	109	47	277	433
of which water-based	39.7	17.2	126*	182.9
of which to seaside	30	13	81	124
Total water-based visits (million)	69.7	30.2	207	307
Average spend per day/night	£33.00	£58.40	£16.90	-
Total spend on fresh and marine water-based days and tourist days	£2.3 billion	£1.76 billion	£3.50 billion	£7.56 bn
Total spend on freshwater-based days and tourist days only	£1.31 billion	£1.01 billion	£2.13 billion	£4.45 bn

* includes a +25 million day adjustment for anglers

A1ix. Net economic losses for commercial aquaculture, fisheries, and shell-fisheries

Although the eutrophication of lakes and rivers increases the biomass of fish present, the associated changes in species composition due to ecosystem changes frequently results in a reduction in the economic value of the fishery. Whitefish, which are a high quality food fish, tend to decline and are replaced with cyprinids, such as bream, roach and carp, which are of lower food quality (Meijer *et al.*, 1990; Harper, 1992; Mason, 1996). In addition, shell-fisheries can be adversely affected by toxins from algal blooms and extreme eutrophication can result in deoxygenation that kills all aquatic life.

Thus the livelihoods of those involved in commercial fishing can be adversely affected, even though revenues from some fishing (eg recreational coarse) may rise. Fisheries have declined in the Dneiper reservoirs in central Europe due to eutrophication, shell-fisheries have been damaged in Chesapeake Bay, and algal blooms have caused fish deaths in the USA and Canada (Postel and Carpenter, 1997; Ribaudo *et al.*, 1999; UNEP, 1999). In the US, data shows that some 44-85,000 annual fish deaths in the late 1980s could be attributed to eutrophication (Steiner *et al.*, 1995). The value of each fish was taken to be from £1.2-7.1 ($1.70-10.00) (according to American Fisheries Society), putting the total annual cost in the range £54,000-607,000 ($75,000–£850,000). In Australia, commercial fishing suffers costs of £0.45 (Aus$1.27) million yr^{-1} due to eutrophication (UNEP, 1999).

In Loch Leven, Scotland, algal blooms arising from the release of phosphorus from loch sediments in the summer of 1992 resulted in fishery losses of £110,000 (Forth River Purification Board, 1993; D'Arcy, 2000). More recently, a diatom algal bloom in the Kennet and Avon Canal at Hungerford resulted in massive fish kill in 1998. The impact on the local trout farm was the loss of a whole year's stock (approximately 150 tonnes) at a cost of £1 million[19]. At Bourne Mill in Colchester, a National Trust owned property, the annual income received for letting the fishing was £220 in 1995. For the last six years, the fishing club have let the water for only £50 per year as it has been in such a poor state. Part of the restoration was to restore some fish swims for the club, when a permanent island replaced two floating islands (these are also used by nesting waterfowl)[20]. Note, though, that these problems may have partly arisen due to siltation and run-off from roads.

Once again, there are no national datasets that record the extent of the problem of eutrophication in the commercial fishing sector. We describe the value-loss relationship as

$$VL_{A1ixNET} = V_f \times f_c \qquad \text{(equation 10)}$$

where VL_{A1ix} = revenue losses for commercial freshwater aquaculture and fisheries;
V_f = value of commercial inland and shell-fisheries in UK;
f_c = frequency of closure (damage to fishery).

According to MAFF (2001), there are some 1000 fish and shellfish aquaculture businesses in the UK, employing 3000 people and turning over £289 million per year. These annually raise 110,000 tonnes of salmon, 16,000 tonnes of rainbow trout, and 12,000 tonnes of molluscan shellfish. In addition, shellfish landings bring in 124,000 tonnes per year, worth a total of £160 million. Most of these businesses, though, are situated in parts of Britain where no eutrophication other than that caused by the industry itself occurs. In this study, we do not assess the costs of eutrophication in marine waters. We assume that the closure rate (damage) holds for commercial fisheries, and that freshwater fish account for only 10% of the total, and that the profit in this sector is 10-20%. Thu s the net economic loss is:

$$VL_{A1ixNET} = \text{£}449 \text{ million} \times (0.0066 \text{ to } 0.0131) \times (0.1\text{-}0.2) \times 0.1 = \text{£}29{,}000 \text{ to £}118{,}000.$$

[19] Grahame Newman, British Waterways, pers. comm. 28th March 2001

[20] Keith Turner, National Trust, pers. comm. 25th April 2001

A1x. Health costs to humans, livestock and pets

Eutrophication carries three potential health risks to humans, livestock and pets. These arise from high nitrate content of drinking water, toxic algal blooms, and enhanced presence of bacterial pathogens[21].

Algal blooms in eutrophic water bodies comprise a potential health hazard to humans and animals in contact with the water. There are 25 species of cyanobacteria that produce a variety of toxins including neurotoxins, hepatotoxins and lipopolysaccharides. The most common toxin-producing species are *Anabaena*, *Microcystis* and *Aphanizomenon,* although a bloom will not always produce toxins (Harper, 1992; Mason, 1996). Public concern about cyanobacteria toxins was raised in 1989 when a summer bloom of *Microcystis* caused the deaths of several dogs and sheep after drinking water from Rutland Water in Leicestershire (Ferguson *et al.*, 1996; Mason, 1996). Although no human deaths have been recorded in the UK, toxins from a cyanobacterial bloom acutely poisoned soldiers swimming in Rudyard Lake in Staffordshire in the 1990s (Ferguson *et al.*, 1996).

A further risk arises amongst people prone to allergenic reactions in contact with water containing cyanobacterial blooms. In addition, water high in dissolved organic carbon, a by-product of dense algal blooms, can produce potentially carcinogenic and mutagenic trihalomethanes when disinfected by chlorination. In the tropics, eutrophic waters can contribute to the spread of diseases such as cholera and typhoid, and produce an environment in which mosquito larvae flourish, so encouraging malarial infection (UNEP, 1999). As these events appear to be rare, we take these costs in this category to be close to zero.

A2. Ecological damage costs

A2i. Negative ecological effects on biota (arising from changed nutrients, pH, oxygen), resulting in changed species composition (biodiversity) and loss of key or sensitive species

Eutrophication of water courses has a direct effect on the primary production of plants and, through changes in pH, available light and oxygen concentration, may indirectly affect the abundance and nature of the organisms within it (Harper, 1992; Mason, 1996; Moss *et al.*, 1996). When the nutrient loading increases, the first response is an increase in microscopic suspended algae (phytoplankton). All water bodies, regardless of trophic state, have more algae present at certain times of the year, but for eutrophic bodies, this continues for longer periods. With changes in trophic state, the species of phytoplankton also change: small algae species (e.g. *Chrysophycaea*) gradually decline as larger green algae species (e.g. *Chlorophyta*) become more abundant, and finally blue-green algae or cyanobacteria species (e.g. *Anabaena*) dominate and flourish. Accompanying this change in species composition is a large increase in the standing biomass of the water body.

[21] Nitrate at very high concentrations in drinking water, combined with presence of nitrate-reducing bacteria, can provoke methaemoglobinaemia (blue-baby syndrome) in infants (Comly, 1945; Conway and Pretty, 1991; Wilson *et al.*, 1999). The last recorded cases in the UK were in the 1950s, though several cases occur each year in rural areas in the USA. Nitrate is no longer considered a significant risk factor in stomach and oesophageal cancers. We thus do not allocate any costs to this category.

In eutrophic waters, a variety of changes disturb the normal ecosystem. With large blooms of blue-green algae, the light and oxygen available to other organisms is reduced. This increased turbidity prevents submerged and rooted macrophytes from growing. As such submerged vegetation normally provides refuges for the large zooplankton species, their loss results in increased predation by fish of zooplankton. The increased production of plants in eutrophic water bodies also leads to increased detritus and sedimentation, which can produce anoxic conditions. Though increased organic material in water increases the biological oxygen demand, daytime photosynthesis can cause saturation of the water with oxygen, causing changes in pH. These changes affect fish and zooplankton species present, but are likely to have minimal consequences for the cyanobacteria as they appear to be more tolerant to environmental alterations than other flora and fauna. After an algal bloom has subsided, another hazard can occur in eutrophic lakes. In the sediments of shallow eutrophic lakes, the conditions favourable to the development of the bacteria *Clostridium botulinum* can arise and outbreaks of botulism can occur. Wildfowl are susceptible to this disease, and birds can die as a result (Harper, 1992; Mason, 1996).

Once again, there is no national database with records of ecosystem changes and losses arising from eutrophication. Some 80% of inland standing freshwater in England is eutrophic (540 km^2), 40% in Wales (50 km^2), and 15% in Scotland (241 km^2), giving a total of 35% of inland water area in the UK as being eutrophic. One study of 135 inland lake SSSIs in England recorded that 84% were adversely affected by eutrophication, with the special interest severely affected in 52% of lakes, and another indicated that 800 (20% of all) SSSIs are dependent on the proper functioning of wetland ecosystems (see Carvalho and Moss, 1995, 1998). However, only a small proportion of this total represents a problem that incurs value-loss costs.

The value-loss costs related to the intrinsic value (non-use) of species and ecosystems affected by eutrophication are difficult to measure. However, several important water species and habitats that are adversely affected by eutrophication are listed in the UK national Biodiversity and Habitat Action Plans (UK Biodiversity Steering Group, 1995, 1998, 1999). We use the cost of restoring these species and habitats as a proxy for the eutrophication costs, even though these would be less than the real cost of eutrophication if rational public policy were to institute programmes only where benefits exceed costs. These plans contain costed targets and action plans for 406 species and 38 key habitats in the UK.

For those species and habitats for which eutrophication is identified as one of the factors causing problems, we have used details in the BAPs, HAPs and SAPs to estimate the costs. The average cost of Species Action Plans is £19,200 per plan yr^{-1} (£7.5 million for 391 SAPs), and there are 13 BAP species affected by eutrophication. The costs for habitat plans for eutrophic lakes is £0.38 to £0.66 million yr^{-1} (for 2000-04), and for mesotrophic lakes £0.35 million $year^{-1}$. But these figures for eutrophic lakes exclude the costs of full restoration of lakes to a favourable condition, and the mesotrophic lakes HAP costs are based on only fifty sites. Individual costs for restoration can be very high – more than £8 million was spent in the Norfolk Broads between 1995-2000 in returning open waters to pre-eutrophication conditions (Madgwick and Phillips, 1996; Moss and Carvalho, 1998), and English Nature plans to spend £1.5 million on BAP lakes during 2001-3. Based on the range of costs incurred for lake restoration, we therefore increase these HAP costs by a factor of ten to give a fairer estimate of value-losses.

We describe the relationship for the value-loss as:

VL_{A2i} $= C_e + C_m + (S \times C_s \times P)$ (equation 11)

where

VL_{A2i} = negative ecological effects on biota resulting in changed species composition (biodiversity) and loss of key or sensitive species
C_e = average annual cost of HAP addressing eutrophic lakes
C_m = average annual cost of HAPs addressing mesotrophic lakes
S = number of Species Action Plans potentially affected by eutrophication
C_s = average annual cost of SAPs
P = proportion of SAP affected by eutrophication

Thus

VL_{A2i} = (0.38-0.66 x 10) + (0.35 x 10) + (13 x 0.019 x 0.1) = £7.34 to £10.12 million.

B. Policy response costs – costs of addressing and responding to eutrophication

B1. Compliance control costs arising from adverse effects of nutrient enrichment

B1i. Sewage treatment costs (to remove phosphorus from household and industry point sources)

Sewage treatment companies incur costs to comply with environmental legislation for removal of phosphorus before it enters water courses. The P and N removal at sewage treatment works which come under the terms of the EC Urban Wastewater Treatment Directive is predicted to cost water companies £580-840 million for Capex during the period 2000 to 2010, and £13-17 million for operating expenditure during the period 2000 to 2010[22]. Under AMP3 (the water companies' investment programme for the period 2000-05), capital expenditure on phosphorus removal at STWs will be in the region of £250 million. For 44 of the 65 STWs where P removal has been approved due to the potential impact of discharges on SSSIs, the capital cost has been projected at £49 million, with an average annual operating cost of £0.06 million.

However, the P removal at STWs that comes under the EC Urban Wastewater Treatment Directive is predicted to cost water companies £58-84 million yr^{-1} for capital expenditure, and £1.5 million yr^{-1} for operating expenditure during 2000-2010. We thus take annual capital expenditure to be £50 million yr^{-1} and operating costs to be £0.3 m yr^{-1}.

The compliance costs for this category, $CC_{B1i} = PC_o + PC_c$ (equation 12)

where CC_{B1i} = sewage treatment costs to remove phosphate;
PC_o = annual operating costs of removal of phosphate by water companies;
PC_c = annual capital costs of removal of phosphate by water companies.

Thus CC_{B1i} is £50.3 million yr^{-1}.

B1ii. Cost of treatment of algal blooms and in-water preventative measures (biomanipulation, stratification, straw bale deployment)

[22] Open letter from the Director General of Water Services to the Secretary of State for the Environment "Setting the quality framework – an analysis of the main quality costings submission 2000-05

Water delivery and management companies incur additional costs through a variety of preventative and restorative measures for treatment of algal blooms. The measures available depend in part on the uses for the water body. Algicides and sediment sealing, for example, can be used when fish production, irrigation and in-water recreation are not the aim (UNEP, 1999). Physical manipulation measures include enhanced flushing of the lake, artificial de-stratification of the water, water level manipulation and sediment dredging. Chemical manipulation of a water body can also be used to seal the phosphorus into the sediments. A variety of biomanipulation measures are also available, and could include i) addition of piscivores; ii) herding of piscivores into a lake by temporarily adjusting the path of flow (by water level manipulation); iii) removal of fish altogether; iv) provision of zooplankton refuges; v) macrophyte harvesting; vi) straw-bale deployment (Reeders *et al.*, 1989; Moss, 1990; Welch *et al.*, 1990; Ridge and Barrett; 1992; Kelly and Smith, 1996; Mason, 1996; Moss *et al.*, 1996; Everall and Lees, 1996, 1997; Klein, 1998; Reidel-Lehrke, 1998: Penrow and Tomlinson, in press).

A common treatment of eutrophication by British Waterways is the deployment of barley straw. At Earlswood Reservoir (a canal feeder reservoir near Solihull in the West Midlands), £5-6000 yr^{-1} was spent on deployment of barley straw to combat eutrophication between 1994-1997. As there were additional costs for monitoring the trophic state of the reservoir, the total operating costs were £5-10,000 yr^{-1}. The commercial value of the fishery at the reservoir has not been threatened[23]. Barley straw is also used within canals. An example is on the Pocklington Canal where straw is applied in October and April each year along a 7.2 km stretch at a cost (for the straw, equipment, boat and man-power) of £5000 yr^{-1} [24].

The damage cost relationship for this category is:

$$DC_{B1ii} = \Sigma Ct_{i-j} \qquad \text{(equation 12)}$$

where

DC_{B1ii} = cost of treatment of algal blooms and preventative measures;
Ct = sum of treatment costs by water companies i to j.

There is no national database for these costs, nor available data for each of the organisations concerned with treatment. Though restoration through sediment dredging can be expensive, the total in this category appears to be small. We thus estimate costs to be £0.5 million yr^{-1}.

B1iii. Costs to farmers of adopting new farm practices

Agriculture is a major source of nutrients in surface and ground water. These nutrient losses arise from leaching and run-off of inorganic fertilizers, sludges and animal manures, and the mineralisation of soil and plant organic matter after conversion of grasslands and forests to arable use. The increased costs of treating drinking water to remove both nitrogen and phosphorus nutrients have led to increased efforts to limit losses from farms.

As nitrates are highly soluble, up to 50% of the nitrogen applied to crops can be leached into drainage systems and end up in surface waters and groundwater (Conway and Pretty, 1991; Mason, 1996; Ferguson *et al.*, 1996)[25]. The phosphorus in applied fertilizers can also reach

[23] Matthew Saxon, pers. comm. 21st March 2001, Grahame Newman, pers. comm. 28th March 2001

[24] Matthew Saxon, pers. comm. 21st March 2001

[25] Some 15% of the N fertiliser and up to 3% of pesticides applied to cropland in the Mississippi River Basin make their way to

groundwaters, although as it is largely in the form of insoluble particulates it is generally not lost via leaching, but via run-off and attached to soil particles. In areas of high erosion rates, up to 60% of the phosphorus in applied fertilizers may end up in water (Mason, 1996). In the UK, of the estimated 120,000 tonnes of phosphorus applied to agricultural land (90,000 P as fertilizer and 30,000 tonnes in animal feeds), some 40,000 tonnes (49% of all sources to waters) are transferred to surface water (D'Arcy *et al.*, 2000). This means that on average 1.2 kg P ha^{-1} yr^{-1} is lost from fields to watercourses on average (Fraser and Harrod, 1998).

In the past, policy measures have focused only on voluntary Codes of Good Agricultural Practice to limit the leaching of nutrients. These suggest a range of techniques, including reduction of soil erosion by better land management, use of slow-release fertilizers, retention of nutrients by buffer zones and in ditches parallel to water courses, the use of artificial wetlands at the ends of feeder streams, and increasing the in-stream nutrient metabolism by stream rehabilitation (Ferguson *et al.*, 1996).

But these have failed to control losses, and so Nitrate Sensitive Areas and Nitrate Vulnerable Zones have recently been established in many areas overlying sensitive aquifers. These mandate farmers to adopt certain practices, in return for which they receive financial compensation (NSA scheme) or capital grants (NVZs). Although both of these schemes are aimed at drinking water quality rather than reductions in eutrophication, we use the costs of subsidising and enforcing NSAs and NVZs as a proxy for this cost category (Lord *et al.*, 1999; Hanley *et al.*, 1999). There are 32 designated NSAs covering 35,000 ha of eligible land, of which 70% has been entered in to the scheme by 297 farmers. All NSAs fall within the 68 NVZs covering 600,000 ha, as designated under the Nitrate Directive (91/676/EEC). Farmers in NVZs are required to comply with mandatory Action Programme measures designed to protect both groundwaters and surface water against pollution caused by nitrate[26]. The payments for NSAs vary from £340-625 per hectare for conversion of arable to extensive grassland, £250 for conversion of intensive to extensive grassland, and £65-105 for low-nitrogen arable cropping. This puts the total annual expenditure at £3.39 million.

One water company, Wessex Water, is investigating alternative action that could be taken to halt the increase in nitrate levels by developing land management agreements with landowners over vulnerable water sources. It is offering to pay farmers in certain areas of Wiltshire and Dorset £40 per ha per year to begin conversion to organic farming though a two-year pilot scheme. In the Frome Valley in Dorset, it is working with FWAG and the EA on nutrient budgeting for participating farmers. The key principle is that it is cheaper to pay to help create sustainable and non-leaky farm systems than it is to clean up the pollution after it has been caused by non-sustainable operations.

B2. Direct costs incurred by regulatory bodies for monitoring, investigating and enforcing solutions to eutrophication

B2i. Monitoring costs for water

Statutory agencies incur additional costs to monitor water bodies for the presence of both nutrients and algae and their decomposition products. We describe the monitoring costs as:

the Gulf of Mexico (Ribaudo *et al*, 1999).

[26] See details of schemes at DEFRA [At URL www.defra.gov.uk]

$$MC_{B2i} = \Sigma Mc_{i\text{-}j} \quad \text{(equation 14)}$$

where

MC_{B2i} = monitoring costs for water;
Mc = monitoring costs for organisations i to j.

The Environment Agency spends £27,000 per year on routine monitoring of nitrate and phosphate as part of its General Quality Assessment scheme, and 8000 sites are sampled monthly in England and Wales (Haygarth *et al*, in D'Arcy *et al.*, 2000). In addition, the costs of monitoring sensitive eutrophic areas for the Urban Wastewater Treatment Directive in preparation for the 2001 review of Sensitive Areas have been estimated to be £210,000 for inland waters in England and Wales.

In Australia, management costs for the Victorian catchments amount to £0.42 (Aus$1.2) million per annum. At Hungerford, where a diatom algal bloom in the Kennet and Avon Canal caused a massive fish kill in 1998, a £100,000 project to investigate nutrient sources has been set up. Additional costs have been by British Waterways for consultancy and management time - estimated to be a further £100,000[27]. Many other local investigations are carried out to determine nutrient sources, links to cause and effect, and to set objectives and targets. Thus MC_{B2i} is estimated to be £0.44 million yr^{-1}.

B2ii. ***Costs of developing eutrophication control policies and strategies***

The final cost category refers to the costs incurred by statutory agencies for the development of eutrophication control policies and strategies. These can be broken down into national and local level activities with a range of statutory and voluntary drivers. The development of the EA's aquatic eutrophication management strategy cost £567,000 over two years (for additional staff and commission research). Implementation of the strategy, including national level action and policy, and local level action plans, is estimated to costs £184,000 per year. At Bittel Reservoirs in Worcestershire (canal feeder reservoirs), a Catchment Management Group has been set up, which is estimated to cost British Waterways in staff time £2000 per year[28]. Other similar management projects have been established by the Environment Agency in the Hampshire Avon (landcare) and Llyn Tegid in Wales. For this category, we estimate annual costs of at least £200,000 yr^{-1}.

C. Benefits of nutrient-enriched water

Ci. Increased value of freshwater and marine fisheries

We also report briefly on evidence for the benefits of eutrophication. Owing to shortage of data, we do not provide a comprehensive benefits value, so no benefit:cost comparisons can yet be made. Eutrophic and hyper-eutrophic waters (assuming that measures are taken to minimise health hazards) can be an integral part of the landscape, providing an important

[27] Grahame Newman, pers. comm. 28th March 2001

[28] Grahame Newman, pers. comm. 28th March 2001

aquatic habitat and increased volume of fish. The increase in primary productivity as a result of nutrient enrichment can have a positive result if fish yields are of species that are edible and marketable. Increased yields from aquaculture or fish harvest may overcome perceptions that eutrophic water is not aesthetically pleasing (UNEP, 1999). In Canada, one indicated that aquaculture in Lake Kootenay had benefited by £0.13 (CAN$0.31) million through the supply of nutrients that would otherwise have to have been purchased (UNEP, 1999).

Cii. Fertilisation effect on farmland

A second benefit arises from the nutrient content of water that may be used for irrigation purposes. Farmers would, therefore, benefit by substituting nutrient-enriched water for purchased inputs of fertilizer. As indicated earlier, annual abstractions of water for irrigation amount to 24,288 Ml per year. If we assume that average nitrate content varies from 10-30 mg NO_3/l (or 2.26-6.79 mg NO_3-N/l), then this amounts to a free input of 55-165,000 tonnes of nitrogen. As one tonne of nitrogen fertilizer costs £120, this represents an annual benefit to farmers of £120 x 55,000-165,000 = £6.6 to 19.8 million.

Ciii. Improved sources of food for wild birds

The third potential benefit of eutrophic water is via the provision of improved sources of food for wild birds. In Eastern Africa, naturally hypereutrophic soda lakes contain dense suspensions of cyanobacteria, which support immense numbers of flamingos and other birds that have become a major tourist attraction (UNEP, 1999). Large numbers of birds may result in increased tourism or hunting revenue to local economies. In the USA, there are three million waterfowl hunters, each year averaging 7 days of hunting migratory duck and geese (US Dept of Interior, 1991). Annual expenditures for these activities totalled £478 ($670) million (Postel and Carpenter, 1997). There are no data for the UK, though recent studies on the values of bird reserves to local economies have indicated the considerable contribution bird watchers make through expenditure on local goods and services (Rayment and Dickie, 2001).

6. Summary of Costs of Freshwater Eutrophication and Key Research and Policy Implications

These findings indicate the severe effects of nutrient enrichment and eutrophication. The damage costs are substantial, causing considerable loss of value to many stakeholders in the UK. Table 5 contains a summary of the cost estimates for the 16 cost categories. The total damage costs of freshwater eutrophication are £75-114.3 million yr^{-1}. The policy response costs are a measure of how much is being spent to address this damage, and these amount to £54.8 million yr^{-1}. These costs are higher than those reported in a recent study of the external costs of UK agriculture (Pretty *et al*., 2000). There still remains uncertainty over these costs, as we have had to extrapolate from specific data and case studies, use proxies for costs, draw on research findings from outside the UK, and make assumptions about the gaps in knowledge over the extent of eutrophication and the direct relationship between nutrient enrichment and damage costs and value-losses.

The damage costs are dominated by seven items each with costs of some £10 million yr^{-1} or more:

i. value-loss in residential dwellings,
ii. drinking water treatment costs for nitrogen removal;
iii. reduced recreational and amenity value of water bodies;
iv. drinking water treatment costs for removal of algal toxins and decomposition products;
v. reduced value of non-polluted atmosphere;
vi. negative ecological effects on biota; and
vii. net economic losses from the tourist industry.

The policy response costs illustrate how much is already being spent to meet legislative obligations, and so cannot be added to damage costs. It would, therefore, be expected that as policy response costs increase, so the damage costs should fall. If damage costs (A) continue to exceed policy response costs (B), then it is worthwhile reducing the damage.

In common with other environmental problems, it would represent net value (or cost reduction) if these losses were prevented at source. A variety of economic, regulatory and administrative policy instruments are available to seek to internalise these costs, thus ensuring that both the `polluter pays' the cost, and the `provider (of clean or unpolluted water) gets' the benefits (Gren *et al*., 1997; DETR, 1999; DNRE, 2000; Hodge and McNally, 2000; Pearce and Seccombe-Hett, 2000; Pretty *et al*., 2001). As a result of the identification of these costs of eutrophication, we identify five policy and research priorities.

Table 5. Summary of the annual costs of freshwater eutrophication in the UK

Cost categories		Range of annual costs (£ million)
C. Damage costs – the reduced value of clean or non nutrient-enriched water		
A1. Social damage costs		
i.	reduced value of waterside dweliings;	£9.83
ii.	reduced value of water bodies for commercial uses (abstraction, navigation, livestock watering, irrigation and industry);	£0.50-1.00
iii.	drinking water treatment costs (treatment and action to remove algal toxins and algal decomposition products);	£19.00
iv.	drinking water treatment costs (to remove nitrogen);	£20.10
v.	clean-up costs of waterways (dredging, weed-cutting);	£0.50-1.00
vi.	reduced value of non-polluted atmosphere (via greenhouse and acidifying gas emissions);	£5.12-7.99
vii.	reduced recreational and amenity value of water bodies for water sports (bathing, boating, windsurfing, canoeing), angling, and general amenity (picnics, walking, aesthetics);	£9.65-33.54
viii.	revenue losses for formal tourist industry;	£2.94-11.66
ix.	revenue losses for commercial aquaculture, fisheries, and shell-fisheries;	£0.029-0.118
x.	health costs to humans, livestock and pets.	unknown
A2. Ecological damage costs		
i.	negative ecological effects on biota (arising from changed nutrients, pH, oxygen), resulting in changed species composition (biodiversity) and loss of key or sensitive species.	£7.34-10.12
	TOTAL	£75.0-114.3
D. Policy response costs – costs incurred in responding to eutrophication		
B1. Compliance control costs arising from adverse effects of nutrient enrichment		
i.	sewage treatment costs to remove phosphorus arising from large point sources;	£50.30
ii.	costs of treatment of algal blooms and in-water preventative measures (biomanipulation, stratification, straw bale deployment);	£0.50
iii.	costs of adopting new farm practices that emit fewer nutrients.	£3.39
B2. Direct costs incurred by statutory agencies for monitoring, investigating and enforcing solutions to eutrophication		
i.	monitoring costs for water and air;	£0.44
ii.	cost of developing eutrophication control policies and strategies.	£0.20
	TOTAL	£54.8

As a result of the identification of these costs of eutrophication, we identify five major policy and research priorities:

1. This study has mostly used aggregated national data to produce an estimate of the total cost of freshwater cultural eutrophication, as there is a lack of comprehensive and harmonised data on specific catchments and river basins. There is an urgent need for greater analysis of representative catchments in order to understand better the nutrient budgets and loads, costs being occurred, and the most beneficial and cost-effective actions, all of which are requirements under the Water Framework Directive.

We recommend model/pilot studies are conducted on representative whole catchments or river basins to identify the sources of nutrients, produce detailed nutrient budgets, analyse eutrophication outcomes, and produce estimates for the costs and benefits of prevention and remediation.

We further recommend that such a study focus on developing the appropriate methodology, and on implementation of large-scale comprehensive rehabilitation of one moderate to large catchment.

2. This study set out to examine the costs of cultural eutrophication only in fresh waters. This inevitably leaves open the question of the costs being incurred in marine and estuarine waters.

We recommend further research on the degree of eutrophication in UK marine and estuarine waters, and the costs currently being incurred both in the UK and in other European countries. The Water Framework Directive requires river basin management, including effects on the coastal environment.

3. There remains uncertainty over the definition of the point at which nutrient enrichment becomes a eutrophication problem with adverse economic effect (both costs and value losses).

We recommend further analysis of the nature of the nutrient-enrichment and eutrophication relationship, and more coordination of data on eutrophication between agencies to ensure joint and efficient responses.

4. There are many gaps in the datasets held by a wide variety of agencies and organisations with both statutory and non-statutory interests and responsibilities in eutrophication. There is a requirement for improved data on the extent of ecological and social damage, and on the costs of in-water preventative and remedial measures.

We recommend more coordination between agencies of data on eutrophication and on its effects and costs to ensure improved joint responses.

5. There remains considerable uncertainty over the specific effects of eutrophication on recreation and tourism, and on the lives of those living and working by affected water courses.

We recommend further research on the value of water-based tourism and sports (both freshwater and marine) and the specific value losses caused by eutrophication.

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Annex A: Terms of Reference for Research

The terms of reference for this research were:

i) Classify the effects of eutrophication as monetisable or non-monetisable;

ii) Consider and develop a methodology for assessing effects;

iii) Collate available information on environmental damage cost estimates from relevant organisations and other sources, and show how these were derived;

iv) Refine estimates in light of a workshop or extended consultation with Environment Agency staff and other stakeholders, together with any new data/information obtained;

v) Produce report and recommendations.

Annex B. Summary of three main methods for valuation of ecosystem services

Contingent Valuation Method (CVM)

CVM involves asking people through a survey what they would like from the environment and its services and how much they are 'willing to pay' (WTP) to achieve that goal or how much compensation they are 'willing to accept' (WTA) to forgo it (Harris, 1984; Bockstael *et al.*, 1988; White *et al.*, 1997; Langfrod *et al.*, 1998; Carson, 2000; Loomis, 2000; Loomis *et al.*, 2000; Stevens *et al.*, 2000). Of the valuation methods currently used this is the most widespread. However, there is considerable scope for different valuations to arise from the method, as much depends on how questions are framed, and the specific wealth circumstance of the respondents. In addition, WTP values are often surprisingly different from WTA values for the same environmental goal. A methodological variation is conjoint analysis (Desvousges and Smith, 1986; Stevens *et al.*, 2000; Farber and Griner, 2000) in which the purpose is to analyse choice in a multi-attribute context. Individuals are presented with choice alternatives with varying values, and asked to select the best or rank them. This hypothetical choice setting seeks to mimic real choices by requiring an individual simultaneously to consider many dimensions of value.

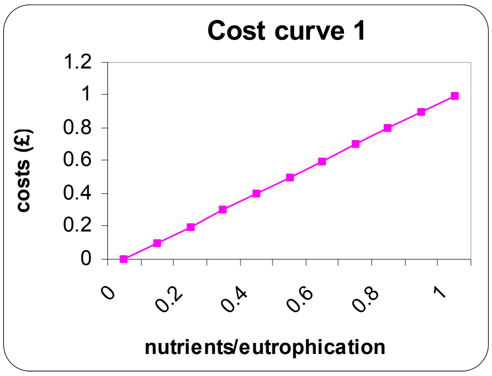
The Travel Cost Method (TCM)

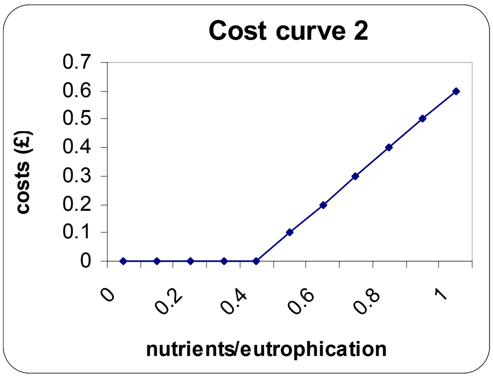
The TCM is commonly used to evaluate consumer demand for access to amenities and recreational sites. This is achieved by calculating the opportunity cost of the time taken to reach a site from a consumer's home combined with the actual expense incurred for petrol and fares (Bhat *et al.*, 1998; Liston-Heyes and Heyes, 1999). The environmental valuation element arises from a consumer's preference for one site that is clean but far away as opposed to a closer `polluted' site. The extra effort involved in reaching the clean site is viewed as the environmental value or premium of the site.

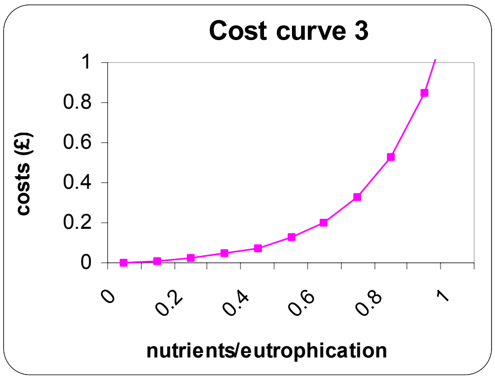
Hedonic Pricing

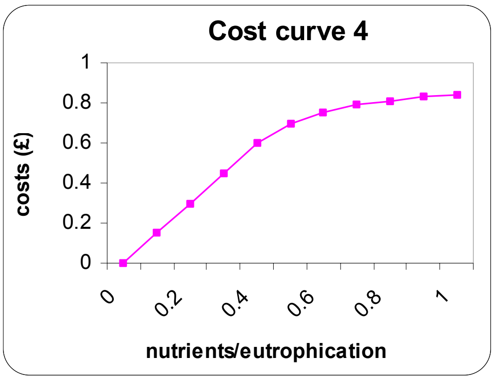
This technique can be used to attribute a proportion of the value of a resource, location or commodity to the environmental factors that influence the total value of those items (cf Doss and Taff, 1996; Pearce and Markandya, 1989; Luttik, 2000;). The most commonly used example is that of house prices. Take identical properties in different locations and with different monetary values (or alternatively the same property valued under varying circumstances over different time) as the total economic value. By controlling for all other variables through multiple regression analysis, such as inflation, seasonal variation or fashion, then a value can be established for the environment component of that property's value. This was the method used by Michael *et al.* (1996, 2000) for investigating the value of lakefront property prices in Maine, USA. The value of the property was paralleled with the clarity of the water as a guide to the effects of water quality on property prices.

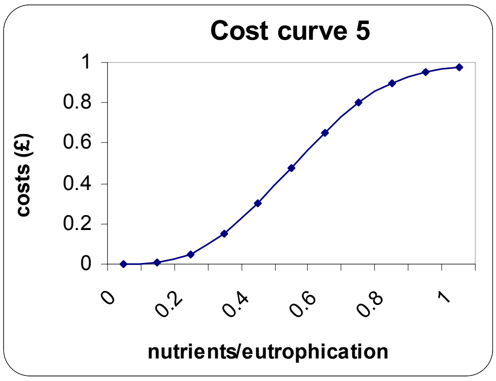
Annex C: Cost curves to describe potential relationship between nutrient enrichment and costs or value-losses (axes vary on a scale from 0 to 1)











Annex D. Summary of 37 economic valuation studies of water bodies giving consumer surplus and willingness to pay for water-based recreation activities

Type of recreation and location	Value per household or person per year (£ 1999 prices)	Value per household or person per visit (£ 1999 prices)	Authors
Angling visit values, UK		£4.50-8.00 (coarse) £2.05-6.60 (trout)	Smith and Kavanagh, 1969; Stabler and Ash, 1977; Willis and Garrod, 1990; NRA, 1995
Recreation value on Norfolk Broads	£128-268		Bateman *et al.*, 1993
Recreation value on Norfolk Broads	£96-281		Bateman *et al.*, 1992
Nature researves	£10.05		Garrod and Willis, 1994
Recreational use on and near canals, UK		£0.53	IWA, 1994
Riverside recreation, Middlesex	£22-30		Coker *et al*, 1989
Informal visits to canals, UK		£0.3-0.4	Willis and Garrod, 1990
Boatable and game fishing values, UK		£52-92	Hanley, 1988
Riverside recreation, UK	£22 - £30		Coker *et al.* 1989
Boatable and fishable water value, UK	£13-28		Forster, 1989
Informal water recreation, UK		£1.34	FWR, 1996
Freshwater angling		£4.3-28.7	
Freshwater canoeing		£1.34	
Marine bathing		£0.9-1.2	
Marine water sports and boating		£1.7-10.0	
Marine angling		£3.80	
River quality improvements, UK	£20-24		Gren and Söderqvist, 1994
River quality improvements, UK	£16		Green and Tunstall, 1991
River quality improvements, River Tame, Birmingham UK	£7-18		Georgiou *et al.*, 2000
Value of swimmable and boatable water, USA	£83-99		Carson and Mitchell, 1993
River quality improvements, USA	£39-49		Gren and Söderqvist, 1994
Recreational use values of forests and watercourses in Canada			Deloitte Touche, 1996
Fishing		£28	
Camping, picnicking, sightseeing		£30	
Fish and wildlife viewing		£35	
Canoeing		£64	
Recreational value of wetlands, Louisiana	£426 (£25 per		Gren and Söderqvist, 1994

	ha/year)		
Formal and informal recreational value of water bodies, USA		£1.6-4.6	Hof and Rosenthal, 1987
Formal and informal recreational value at lakeside, USA		£1.8	Parsons and Kealy, 1994
Recreational benefits of eliminating eutrophication, New Hampshire		£1.00	Needelman and Kealy, 1995
Value of water quality benefits in Wisconsin lakes, USA			Parsons and Kealy, 1992
Swimming		£1.1-5.4	
Fishing		£0.6-1.2	
Boating		£0.2-10.4	
Viewing		£0.2-8.5	
Improved water quality for angling		£0.2	Tay and McCarthy, 1994
Recreation, USA – all uses		£43.3	Walsh *et al.*, 1992
Swimming		£29.3	
Boating		£40-60	
Hiking		£37	
Fishing		£69-92	
Game hunting		£39-45	
Angling recreational use, USA		£5.5	Whitehead, 1992
Bathing water quality at Great Yarmouth and Lowestoft beaches, UK		£14.8 (holiday visit) £13.4 (day trip) £12.3 (locals)	Georgiou *et al.*, 1998
Beach recreation, UK		£1.1	Bateman *et al.*, 1993
Beach users in national parks, UK		£24-28	Bateman *et al.*, 1993
Beach recreation, UK	£7.4		Gren *et al.*, 1990
Beach visitors, USA		£54	Bell and Leeworthy, 1990 (in Freeman, 1995)
Recreational beach visitors, New Jersey		£24-86	Freeman, 1995
Benefits through water quality improvements, Chesapeake Bay, USA	£12-87 (non-users) £32-132 (users)		Bockstael *et al.*, 1987
Access to state park and coral reef, Florida		£218-3375	Freeman, 1995
Coastal and beach recreation, USA		£12.2	Penning-Rowsell *et al.*, 1992
Recreational benefits of improved water quality, Brittany, France		£16-21	Le Goffe, 1995
Amenity losses to algal blooms in Murray Darling Basin, Australia	Total £0.68 million per year		DNRE, 2000

Note: some of these studies do not distinguish between use and non-use values

Annex E. Summary of actions taken by water companies to remove nutrients

Water companies also incur a second tranche of costs during treatment of sewage, storm water drainage and industrial effluents. The principle of sewage treatment is to reduce the polluting capacity of effluents emitted to watercourses. However, sewage treated to secondary level is still responsible for large concentrations of nitrogen and phosphorus reaching waterbodies. Sewage effluents, for example, typically contain 10-30 mg P $litre^{-1}$ (Moss *et al.*, 1996). In the 1940s, detergents rich in phosphorus were introduced, and as their use rose rapidly, so did the amount of phosphorus present in domestic sewage. Tripolyphosphate was found to soften and buffer the water, emulsify grease and keep dirt particles in suspension, and so was widely used in laundry detergents. Although the amount of phosphorus in detergents is now decreasing as manufacturers become aware of the pollution potential, detergents are still often responsible for over 50% of phosphorus in sewage. Domestic sewage effluents may also contain industrial effluents, some of which contain large amounts of nutrients, especially discharges from breweries, dairies, food processing, sugar refineries and abattoirs (Harper, 1992; Mason, 1996).

The EC Directive on Urban Waste Water Treatment (91/271/EEC) guides the standards for nutrient removal at sewage treatment works (STWs), and it states that where eutrophication is occurring or where there is a risk of eutrophication, specific standards must be complied with. At a STW where phosphate is removed, it is precipitated from the final effluent by the addition of iron salts, sodium hydroxide or aluminium sulphate. The precipitated material rich in phosphorus then settles out, is removed as sludge and disposed of by landfill, land spreading, or recycled to industry. This stripping process removes up to 95% of phosphorus (Moss *et al.*, 1996). The phosphorus may also be removed biologically by activated sludge, which in optimum conditions can remove up to 90% of P content (Ferguson *et al.*, 1996). Tertiary treatments of these kinds are very efficient at removing phosphorus, but can double the cost of sewage treatment.

Sewage effluent is also high in nitrogen and the amount discharged can be markedly reduced by trickling nitrate-rich effluent through an artificial reed bed, which will filter out nitrogen. However, this method involves the availability of large areas of land. Nitrogen can also be removed from sewage effluent by ion exchange, but this is expensive. Depending on the individual situation for each water body, the reduction of phosphorus in the sewage effluent is generally cheaper and easier to achieve than reductions in nitrogen. As it is phosphorus that is the limiting factor in algal growth in most fresh waters, in most cases it is therefore advisable to concentrate on the phosphorus first.