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SID 5 Research Project Final Report

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Project identification

1. Defra Project code: WT0706
2. Project title: Benefits and Pollution Swapping: Cross-cutting issues for Catchment Sensitive Farming Policy.
3. Contractor organisation(s): Institute of Grassland and Environmental Research
   North Wyke
   Okehampton
   Devon
   EX20 2SB
4. Total Defra project costs: £104,605
5. Project: start date 01/01/06
   end date 27/10/06

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Executive Summary

7. The executive summary must not exceed 2 sides in total of A4 and should be understandable to the intelligent non-scientist. It should cover the main objectives, methods and findings of the research, together with any other significant events and options for new work.

Reducing diffuse water pollution from agriculture provides potential benefits to society, which include:

- Primary effects through improved terrestrial and aquatic habitats (biodiversity), amenity, recreation (including fishing and bathing), drinking water quality and health.
- Secondary effects on atmospheric emissions (with impacts on air quality and climate change).

This study was designed to estimate the public benefits from a set of policy options derived by Defra from 44 farm management methods (Cuttle, 2006). These methods were designed to reduce agricultural emissions of nitrate, phosphorus, faecal indicator organisms (FIOs) and sediment. Information on the physical effects of the methods and their costs was derived from project ES0203 (Anthony, 2006).

We reviewed the literature on the public benefits from improvements to water quality from DWPA measures (including secondary effects on air quality, and followed the structure of benefits used by the Environment Agency in EA (2005). The benefits from policy measures to address DWPA were estimated using the public's willingness to pay (WTP) (or equivalent welfare measure) for improvements in water and air quality and reductions in the cost of current measures that address water pollution and its secondary effects. The primary and secondary benefits and environmental costs were estimated for each of the mitigation methods used to reduce the agricultural losses of NO_3, P and sediment. Finally, we calculated the primary (NO_3, P and sediment) and secondary (ammonia emissions, greenhouse gas emissions and CO_2-energy loss) benefits delivered by each of the specified policy options (by Defra) and related these to the costs of delivering the options. In the absence of better information a pro rata approach was used in which a 1% reduction in the level of agricultural pollutant emission (above assimilative capacity) was assumed to deliver 1% of the total potential benefit.
Five options were evaluated in terms of the benefits delivered over and above Business as Usual (BAU). These were:

- Water Protection Zone (WPZ) designed to reduce overall P loads to watercourses by 48%
- WPZ + scheme (WPZ+scheme)
- WPZ+ advice +scheme (WPZ+scheme+advice)
- WPZ30 (designed to reduce overall P loads to watercourses by 30%)
- Financial instrument (FI)
- Financial instrument + advice (FI + advice)

Our analysis showed that BAU delivers significant benefits of between £230m and £300m (including benefits to bathing waters by reduction in FIO transfers). WPZ delivers benefits over and above those from BAU of £119-£175m per year (incl. benefits to bathing waters by reduction in FIO transfers). These derive mainly from P and sediment reduction, although there are significant benefits from reduced nitrate losses. Secondary effects make only a small contribution. WPZ + scheme delivers slightly higher benefits but advice contributes little additional benefit. WPZ30 has a relatively low cost but delivers substantial benefits. The cost is £16.4m per year and the benefit range is £69-£109m.

The FI had little impact, with benefits of only £1m-£2m, and advice given in addition to FI again produced little in the way of additional benefits. The upper bound estimates exceed the cost of the options in all cases expect that for FI + advice. In most cases the lower bound estimate of benefit was below the policy costs. Where advice was part of the policy package benefits did not cover costs. Overall, only the WPZ30 gives a high ratio of benefit to cost (range 4.2-6.6). The others give rather low and unsatisfactory benefit : cost ratios even when the upper bound benefit is used.

It should be noted that both the costs and benefits for WPZ were estimated using a general, and non-targeted approach. At Defra's request, the costs were estimated based on integrating a set of mitigation methods to reduce P loads to water by an average of 48% across all river courses, irrespective of water quality. The benefits analysis therefore took the same approach. In reality, should a more targeted approach be taken, i.e. policy instruments targeted at P impacted watercourses and not those with a high assimilative capacity, then the benefit : cost ratio would be improved in those catchments. A targeted approach would be the next step in the research.

In terms of timeframes, for nitrate, if N inputs were reduced significantly then it would be reflected in reduced N loads to water relatively quickly (probably within 1-2 years). Nitrate has little impact on the eutrophication of freshwater rivers and lakes, so the impact on benefits would be limited to nitrate removal costs from drinking water which would be seen within the same timeframe and would impact on operating and future capital investment costs. The same could also be said for FIOs, if the source was reduced. For P, however, if P inputs were reduced significantly, it could be decades before levels in rivers decreased. We believe that reduction in diffuse P losses from agricultural land will have little impact on the main river reaches due to the importance of point source contributions that already cause exceedance of concentration thresholds. However, it is possible that agricultural sources are significant in low order river systems and relatively unpopulated catchments.

Because of the uncertainty in knowing the rate of decline in P loads as a result of the decrease in agricultural P inputs, we did not feel it appropriate to 'guestimate' this in this study. So, although we recognize that most P impacts will not be immediate and could take many years to manifest themselves, due to a lack of information on the timeframe for benefits, in this study we have had to assume 'immediate' benefits and cost accordingly, i.e. assuming a zero discount rate.

8. As a guide this report should be no longer than 20 sides of A4. This report is to provide Defra with details of the outputs of the research project for internal purposes; to meet the terms of the contract; and to allow Defra to publish details of the outputs to meet Environmental Information Regulation or Freedom of Information obligations. This short report to Defra does not preclude contractors from also seeking to publish a full, formal scientific report/paper in an appropriate scientific or other journal/publication. Indeed, Defra actively encourages such publications as part of the contract terms. The report to Defra should include:

- the scientific objectives as set out in the contract;
- the extent to which the objectives set out in the contract have been met;
- details of methods used and the results obtained, including statistical analysis (if appropriate);
- a discussion of the results and their reliability;
- the main implications of the findings;
- possible future work; and
- any action resulting from the research (e.g. IP, Knowledge Transfer).

1. Introduction

Reducing diffuse water pollution from agriculture provides potential benefits to society, which include:
Primary effects through improved terrestrial and aquatic habitats (biodiversity), amenity, recreation (including fishing and bathing), drinking water quality and health.
Secondary effects on atmospheric emissions (with impacts on air quality and climate change).

This study was designed to estimate the public benefits from a set of policy options derived by Defra from the 44 farm management methods described in Cuttle *et al.* (2006)[1]. These methods were designed to reduce agricultural emissions of NO_3, P, faecal indicator organisms (FIOs) and sediment. Cuttle *et al*. did not quantify the effect of management methods on FIOs at the national scale, so the quantitative analysis of benefits was therefore restricted to NO_3, P and sediment. Information on the physical effects of the methods and their costs was derived from project ES0203 (Anthony, 2006).

1.1 Methods of benefit measurement

Pearce *et al*. (2006) reviewed the methods by which the costs and benefits of changes to the environment can be valued in monetary terms. Valuation methods have also been discussed by Turner *et al*. (2005) in the context of wetlands, and by Hartridge and Pearce (2001) in the context of agricultural pollution. The benefits from policy measures to address DWPA can be measured in two principal ways:

- The public's willingness to pay (WTP) (or equivalent welfare measure) for improvements in water and air quality. This measures the total benefit to the public from a policy intervention.
- Reductions in the cost of current measures that address water pollution and its secondary effects. Cost-based measures are typically used to provide a lower bound estimate of benefit when direct measures of benefit are not available. They are also appropriate when an existing policy framework (in the public or private sector) is in place from which the cost-saving from the introduction new measures can be assessed (e.g. water treatment costs; cost of achieving targets for greenhouse gas emission reductions).

1.2 Baseline losses

Losses of nitrate, P and sediment from agriculture in England were estimated for different sectors by ADAS and IGER (project ES0203, Table 1.1). This provided the baseline for the impacts of measures designed to reduce pollutant emissions. Data on the impact of policy options in reducing baseline emissions from the 2000 baseline were provided by Anthony (2006).

[1] Defra project ESO203

Table 1.1 Baseline losses of nitrate, P and sediment from agricultural sectors in England (2000) (source, project ES0203)

	Nitrate-N (kt)	Phosphorus (kt)	Sediment (kt)
Arable	157	5.65	891
Beef	21	1.14	298
Dairy	48	2.98	222
Pigs	8	0.28	*
Poultry	6	0.17	*
Rough grass	6	0.16	72
Sheep	2	0.07	**
Woodland	2	0.19	9
Total	**250**	**10.64**	**1,490**

Source Final report for Defra project ES0203. * = sediment loss from Pigs and Poultry is included in the value for Arable. ** sediment loss from Sheep included in Rough grass.

1.3 Method

We assume that policy options can be targeted to all catchments, and where water courses have the capacity to assimilate a certain level of pollution this has been taken into account when costing the benefits. We also assume that there are no changes in policy measures relating to non-agricultural emissions of NO_3, P or sediment that would have implications for the assessment of benefits from changes in agricultural management.

The general approach was as follows:

- Review the literature on the public benefits from improvements to water quality from DWPA measures (including secondary effects on air quality. In this we follow the structure of benefits used by the Environment Agency in EA (2005).
- Estimate the primary and secondary benefits and environmental costs in monetary terms that could be ascribed to the mitigation methods used to reduce the agricultural emissions of NO_3, P and sediment.
- Calculate the primary and secondary benefits delivered by each of the specified policy options and relate these to the costs of delivering the options. In the absence of better information a pro rata approach was used in which a 1% reduction in the level of agricultural pollutant emission (above assimilative capacity) was assumed to deliver 1% of the total potential benefit.

2. Estimates of potential primary benefits

2.1 Drinking water quality

Hanley (1989) surveyed households in East Anglia and used the contingent valuation (CV) method to estimate their WTP for water supplies containing a maximum of 50 mg/l nitrate. The mean WTP was £12.97 per household (aggregated to £10.8m per year over the East Anglia population). This exceeded the actual cost of water treatment to reduce nitrate levels to 50 mg/l. WTP estimates for P etc. were not available in the literature.

However, in this case a WTP measure of benefit is less relevant than a cost-based measure because water companies operate within an existing legislative framework that defines standards for drinking water quality. We take these standards as given. Any benefit from changes in the composition of water available to the industry will therefore be reflected in operating and/or capital costs. Any reduction in cost will benefit society either through reduced prices to consumers or increased profits to shareholders.

2.2.1 Treatment required

The agricultural pollutants in raw water for which treatment is required to achieve drinking water standards are pesticides, nitrates and Cryptosporidium. Pretty *et al*. (2000) estimated that 89% of pesticides, 80% of nitrate and 90% of Cryptosporidium in raw water were derived from agriculture (Table 2.1). Of these, this study was only concerned with measures that reduce nitrate emissions from agriculture. Pesticides are considered as a knock-on effect, but impacts are not quantified. Similarly, effects on Cryptosporidium oocyst levels were not part of the study, although there is clearly a link between FIO losses from agriculture and Cryptosporidium losses, with young cattle/sheep having particularly high shedding rates.

Table 2.1 Estimates of the financial costs of water pollution from agriculture (£m per year)

	Eftec (2004) (UK)		**Pretty *et al*. (2000) (UK)**	
	% of total cost attributable to agriculture	**Cost (£, m)**	**% of total cost attributable to agriculture**	**Cost (£, m)**
Pesticide removal by water companies	89	131.2	89	120
Nitrate removal by water companies	70	15.6	80	16
Phosphate and particle removal by water companies	43	34.5	43	55
Zoonoses removal by water companies	-	-	90	23
Pollution incidents -remedial treatment (restocking etc)	-	2.0 (E and W)	8.25 (2,600 incidents)	6
Monitoring and advice on pesticides and nutrients	-	Not estimated	100	11
Total		**183**		**231**

As regards P emissions, OFWAT (2006a) indicate that reductions in phosphate emissions have no effect on water company costs because drinking water is not treated to remove phosphates. The same is true for particulates (sediment). Both Pretty *et al*. (2000) and Entec (2004) give costs to remove phosphate and particulates from raw water by water companies (Table 2). Similarly, Pretty *et al*. (2000) ascribe all the costs of monitoring and advice on pesticides and nutrients to agriculture, when clearly there are other urban sources of pesticides and nutrients. The basis for their estimates is unclear.

2.1.2 Cost of water treatment

Eftec (2004) and Pretty *et al*. (2000) estimated the external costs of water pollution from agriculture in the UK principally in terms of the costs of treating drinking water (Table 2.1). Of the totals of £183m per year (Eftec) and £231m per year (Pretty), the largest component related to the cost of pesticide removal. The cost of nitrate removal from agricultural sources was estimated in both studies at around £16m per year for the UK.

These estimates are not very informative for the present study, because they do not focus on the cost reductions that would occur following policy measures that produced an improvement in raw water quality. The nitrate cost appears to be an upper bound on the total benefit to water companies from a cessation of raw water pollution flows from agriculture. However, this is not the case. Not only would surface and ground water supplies remain contaminated and require treatment, but costs would be likely to increase in the future due to increased drinking water demand from changes in population and demography. A different approach is needed that identifies the impact on water company costs of reductions in NO_3, P, sediment and FIO.

2.1.3 Reduction in costs due to agricultural measures

Water companies can reduce their treatment and blending costs through:

Reduced operating expenditure; and/or

Reduced capital expenditure (reduced physical depreciation or the avoided cost of additional future capital investment).

We approached OFWAT for information from the June returns made by Water Companies to OFWAT. Returns over the 1989-2005 period are analysed in OFWAT (2006b). Capital was depreciated over 28 years at 5% real interest. Table 2.2 summarises the costs for nitrate, pesticides and Cryptosporidium treatment. The historic costs of treatment to reduce nitrate levels was around £20m per year. The majority of the cost derived from capital investment in plant.

Table 2.2 Water companies costs to improve raw water

Pollutant	Total capital expenditure (1989-2005) (£m)	Additional operating expenditure (1989-2005) (£m per year)	Annual cost (£m per year)
Nitrates	250	4	20.8
Pesticides	965	34	98.8
Cryptosporidium	494	9	42.1

All figures adjusted to 2004/05 prices. Annual cost derived from OFWAT (2006b) data by depreciating capital over 28 years at 5% and adding operating expenditure.

Estimated costs of water treatment over the 2005-2010 period are given in OFWAT (2006c). Costs to reduce nitrate levels are £288m for new capital expenditure and £6.0m for annual operating expenditure. This level of annual capital expenditure (£57.6m per year) is higher than that in the past and is influenced by likely high requirements in the immediate future. If this trend was to continue the implied total annual cost would be £66.6m. However, this is not expected to be the case.

Cost savings from agricultural mitigation measures will be site and water company specific, depending on the location and nature of the measures, and their impacts on future capital investment and operating expenditures.

OFWAT (2006b) has undertaken a preliminary analysis of the cost saving from measures to meet the requirements of the Water Framework Directive (WFD), which were assumed to be put in place in 2009. They assumed that the impacts on surface water nitrate levels would occur after 18 months, and impacts on groundwater after 15 years. Four scenarios were examined. Of these, the extremes were (i) capital replacement delayed by 20% of the asset life and operating expenditure reduced by 30%; and (ii) closure of treatment/blending plants. Baseline costs were derived from the past three AMP[2] periods (1995-2006) and for the next AMP period (2010-11). Cost savings for the extreme scenarios varied from £81 to £411m in PV[3] terms.

However, taking a more forward looking approach it was concluded that nitrate concentrations in the raw water sources available to companies were likely to continue to rise driven by demographic and economic factors. This would require water companies to use more 'contaminated' sources. Based on alternative assumptions for investment required in new plant to treat ground and surface waters in the future, OFWAT (2006b) calculate a PV of the cost saving from agricultural measures of between £176 and £967m for a period lasting to 2070. OFWAT (2006b) stress the uncertainty and preliminary nature of the calculations. In terms of annual costs, OFWAT (2006b) convert the capital sums at 5% interest over the period to 2070 to give costs of between £9 and £50m per year. Since these are post tax estimates, the social benefits from WFD measures after adjusting for corporation tax receipts would be **£13m-£71m** per year.

We conclude that any cost saving to Water Companies from reductions in the levels of agriculturally-derived P, sediment, FIOs or Cryptosporidium would be minimal. Although, intuitively one could expect there to be a reduction in treatment costs for Cryptosporidium, unless livestock are completely removed from a catchment, it is unlikely that a Water Company would take the risk of not treating the water, as the risks to human health (and the company's public image) are too high.

2.2 River water quality (amenity)

2.2.1 WTP for quality improvements (amenity)

There are two main studies (Wills and Garrod, 1996; Georgiou *et al*., 2000b) that measure the public's use value of improvements to river water quality. In both cases, the benefits are a mixture of improvements to amenity and biodiversity, but it is not clear whether all benefits were captured in the studies. The studies have been reviewed in detail by Bann *et al*. (2003) in the context of benefits assessment guidance for PRO4[4]. In order to raise the benefits per household to a national scale, both EA (2005) and Eftec (2004) have linked the river water quality standards used in the studies to the River Ecosystem (RE) classification. The system of

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[2] Water companies' five year Asset Management Plans

[3] Present Value of the future stream of cost saving. This is the present sum obtained by discounting the future annual sums.

[4] In 2004 the Director General of Water Services set the price limits that water companies will be able to charge for the period 2005-2010. This price review process is referred to as the 2004 Periodic Review (PR04).

classification has six classes and is described in Defra, (2005) and NRA (1994). EA (2005) give the lengths of rivers in each class in England (Table 2.3).

Table 2.3 Quality of rivers in England and Wales by River Ecosystem (RE) class. Kilometres of river in each class (2002) (Grade 1 is the highest quality)

RE grade	km	% length of graded rivers
Ungraded	113	
1 (good quality)	12.856	31.9
2	14.188	35.2
3	6.886	17.1
4	3.087	7.7
5	3.125	7.8
Worse	139	0.3
TOTAL	**40.391**	**100.0**

Source: quoted in EA (2005).

The EA (2005) estimate of benefits is based on Willis and Garrod. EA (2005) estimate the benefits from improving the water quality of all rivers to R2 or R3 standards as £33.4-£47.1m per year (Table 2.4). In 2006/07 prices the aggregate WTP for improvements to river water quality was £34.9m[5] per year (to RE3) and £49.3m per year (to RE2).

Table 2.4 Benefits from improving river quality in England and Wales

	Improve all rivers to RE2	**Improve all rivers to RE3**
Length of river below RE3	6351 km	6351 km
Length of river at RE3	6886 km	0
Total length of river	13237 km	6351 km
Trips/km/year	7500	7500
£/person/trip for rivers changing from RE4 and below	£0.84 (1)	£0.701(1)
£/person/trip for rivers changing from RE3	£0.138(1)	0
Benefits from river improvement from RE4 and below ((£m 2004/5 prices)	£40m pa	£33.37m pa
Estimated benefits from river improvement from RE3 to RE2 (£m 2004/5 prices)	£7m pa	-
Total (£m pa 2004/5 prices)	£47.12m pa	£33.37m pa

Note: 2004/05 prices

Georgiou *et al*. (2000b) determined the WTP for specified levels of quality improvement in the river Tame. WTP estimates for improved quality were derived for three levels of improvement

[5] adjusted to 2006/07 prices using the GDP deflator.

(large, medium and small), which were described in terms of improvements to recreational amenity and biodiversity (roughly equivalent to RE1, RE2/3 and RE4). Raising the results from one river in one location to the national level is problematic. Georgiou found that with distance decay only the population within 36 miles of the river were willing to pay for a large improvement. It would need a GIS analysis to determine the relevant population for benefit transfer to each river in England and Wales. In addition, the socio-economic characteristics of households (and by implication their WTP for quality improvements) will vary in each river locality.

Eftec (2004) classified rivers as in 'fair' or 'poor' quality and aggregated the WTP calculated by Georgiou *et al*. on a per km basis using the figure of £5,560 per km per year. They multiplied this by the length of rivers in E and W (12,669 km) to give a total damage of £70m per year (£74.8m at 2006/07 prices). Whilst this is a rather crude method of estimating the aggregate benefit from quality, we use it together with the EA estimate to give a range of potential benefit, as it is the best estimate available.

The available evidence on the public's WTP for river quality improvements is quite limited and transfer of values to the national scale may have large errors. However, the available estimates indicate that benefits from improvement to good quality (RE2) range between £49m and 75m per year. But reductions in agricultural pollution will only account for part of this improvement. If we take the Environment Agency (EA, 2005) figure of 30-50% attributable to agriculture the potential amenity benefit from improved water quality is in the range **£15m-£37.5m** per year.

2.2.2 GQA river classification and assimilative capacity

The scenarios in the two WTP studies did not specifically relate quality to any of the NO_3, P, FIO or sediment parameters. Nor are these easily related to the RE scale, since this is not defined in terms of these characteristics. The RE classes are mainly determined by the levels of dissolved oxygen, biological oxygen demand and total ammonia. However the Environment Agency also uses a General Quality Assessment for Rivers (GQA) which takes into account P and NO_3 concentrations. These are graded on a scale of 1 (very low) to 6 (extremely high) (Table 2.5). In the case of P (measured as total reactive P on unfiltered samples) a mean level of 0.1 mg P/litre (class 4) is regarded as high and is considered indicative of potential existing or future problems of eutrophication, although clearly a number of factors influence whether a river will be eutrophic. In England in 2005, 57.1% of river lengths were assessed to have a high P status (i.e. Class 4 or above) (EA, 2006), and conversely 43% of river lengths were assessed as already below eutrophication limit values, and hence would not benefit from reductions in P loads. Anthony (2007) calculated that 5.2kt of total P flow into river classes 1, 2 & 3, i.e. these river classes could assimilate 5.2kt P (assimilative capacity for P). GQA river classes 5 and 6 (>0.2 mg P/litre) account for 41.3% of the England's river lengths, where concentrations (and hence loads) would have to be reduced by a minimum of 50% in order to attain a concentration of <0.1 P/mg. It seems unlikely that 50% reductions in P concentrations in class 5 and 6 river lengths are achievable. Therefore, when estimating benefits from policy options it is only expected that benefits will occur in 15.7% of catchments (class 4). A more targeted approach with measures directed at river catchments in classes 4, 5 and 6, would increase this percentage to 27.5[6]% of problem catchments.

In this analysis, we have assumed that N and P loads will need to be reduced in proportion with concentrations as we do not have models that explicitly represent the in-channel processes etc. NB, it may be that loads have to be reduced by a greater percent than the intended reduction in concentrations if there is a negative relationship between concentration and flow.

[6] class 4 length as % of length in classes 4,5, and 6.

Table 2.5. GQA River classifications, concentrations of P and NO_3 and proportion of river lengths in England (2005)

Class	1 (very low)	2	3	4	5	6 (extremely high)
P conc (mg P/l)	<0.02	>0.02 to 0.06	>0.06 to 0.1	>0.1 to 0.2	>0.2 to 1.0	>1.0
Proportion of river length (%)	11.6	19.9	11.4	15.7	32.6	8.7
NO_3 conc (mg NO_3/l)	<5	>5 to 10	>10 to 20	>20 to 30	>30 to 40	>40
Proportion of river length (%)	12.9	12.1	22.0	21.0	18.4	13.6

Source: General Quality Assessment for rivers, EA 2006

Similarly, for nitrate, in 2005 the EA's GQA scheme estimated that 32% of river lengths were considered to be high in nitrate (i.e. above a mean value of 30 mg/l NO_3 (Class 5), which roughly corresponds with the 95%ile of 50 mg/l used in the EC Drinking Water Directive and the EC Nitrate Directive, and conversely 68% were assessed to be below the 30 mg/l mean value (Table 1.5). With nitrates, it was not necessary to derive an assimilative capacity because benefits were based on water company treatment costs (see Section 2.1). With sediment, the assimilative capacity was taken as zero, with any reduction in sediment losses reducing damage.

Of the water quality attributes studied, P concentration and associated sediment are most likely to limit water quality for amenity purposes. The evidence that in only around 27.5% of catchments are P measures likely to be successful in reducing the current high river P concentrations is accounted for in the calculation of the benefit from these measures (see 2.9).

2.2.3 Impact of agricultural measures

A pro rata approach was used to estimate the benefit derived from reduced emissions (see 1.3). There is some evidence for diminishing marginal benefits to pollution reduction (EA, 2005), such that the greatest benefits will occur where agriculture is a significant polluter of rivers currently with a poor quality status and which are located in highly populated areas. However, there was no evidence of diminishing marginal benefits in the Georgiou *et al*. (2006b) study.

The agricultural measures deliver different mixes of pollutant reductions, but the benefit estimates only deal with rather general concepts of improved water quality. We used a weighting system to derive the impact of a mix of pollutant reductions. The overall balance of weighting for river quality (amenity) was 10% NO_3, 40% P, 10% FIO and 40% sediment, although clearly this varied depending on the water quality benefit. The weightings were based on expert views within the team of consultants and took account of both amenity and biodiversity effects. The weightings can be interpreted as the % of occurrences when each pollutant is the limiting factor and where an instrument that reduces that pollutant will be effective in delivering a public benefit.

2.3 Fishing

The studies discussed above seemed unlikely to fully capture the specific benefits of improved water quality to anglers. We have therefore estimated an additional benefit to the angling population. Similar problems with the definition of water quality are associated with studies that value the benefits to anglers from water quality improvement (e.g. Davis and O'Neill, 1992; Spurgeon *et al*., 2001).

Spurgeon *et al*. (2001) estimated the WTP to maintain current angling opportunities at £2.86-£3.67 per trip. EA (2005) used the Spurgeon data and estimated the increase in trip numbers generated by an improvement in water quality, They derived a figure of £71m per year (£74m in 2006/07 prices) for the benefit from improved angling water quality in E and W. This is not an especially reliable estimate due to the lack of clear evidence on anglers' WTP for improved water quality.

There is evidence of a much higher WTP from a study of recreational sea angling (£26 per day for shore anglers) (Drew Associates, 2004). Sea anglers were also willing to pay £0.81 per fish for increased catches and £0.27 for each % increase in the size of fish caught. The potential gains from improved water quality for angling appear to be substantial if the population or size of fish was to respond to improvements in water quality.

Without other evidence we have used the EA (2005) figure of £74m per year. The EA (2005) give the agricultural share of fishing pollutant effects as 20-50%, which suggests that the potential for benefit from agricultural measures is **£15-£37m** per year.

The weighting used was 10% NO_3, 10% P and 80% sediment.

2.4 Freshwater eutrophication

Eutrophication is mainly a problem of standing water but it can also occur periodically in some rivers. Pretty *et al*. (2001, 2003) analysed the costs of damage caused by freshwater eutrophication in England and Wales. This was estimated at £62-106[7]m. This included an estimate of the cost associated with secondary emissions of N_2O, CH_4 and NH_3. The EA(2005) estimated the damage from excess phosphates in lakes in England and Wales at £45-73m per year.

Bateman *et al*. (2006) used a dichotomous choice (DC) method to estimate the WTP for the application of measures to prevent eutrophication in East Anglian rivers and lakes. The WTP per household was £38.5-£75.4 per year depending on the assumptions made about non-respondents. Scaling this up to the 21.6m households in England gives a range of £832m-£1,628m. This assumes that the public's WTP in East Anglia is representative of that throughout England. This value is markedly higher than the largely cost-based estimate made by Pretty *et al*. (2003) and EA (2005) (see below). However, the number of algal blooms in East Anglia per decade (624) was slightly higher than the average for England (528) (Pretty *et al*., 2003). In addition, the EA (2005) estimate that only 45% of freshwater eutrophication is attributable to agricultural pollution. Moreover, Anthony and Lyons (2006), in a detailed study of P emissions for Defra, estimate a figure of 24% for the diffuse agricultural component of P emissions to freshwaters. Using this latter figure and adjusting for pollution incidents gives a range for the benefit of **£168m-£330m** per year.

Clearly it would be preferable to have WTP data from other regions of England to provide a better base for aggregation. In addition, integration of benefit measures with the critical load framework used to identify eutrophication risk would be desirable. However, this level of detail on benefits is not available. We therefore use the Bateman *et al*. (2006) results as the best guide

[7] Pretty *et al* .include drinking water treatment costs to remove N. This is excluded from the figures used here to avoid double counting.

to the public benefits from measures to prevent eutrophication. However, it could be argued that some measures are already in place that contribute to reducing eutrophication (e.g. Nitrates Directive and targets to reduce ammonia emissions), a cost based approach may be a more appropriate for valuing policy benefits. However, current policy is not sufficiently well developed in relation to eutrophication to provide a measure of marginal cost. We therefore prefer the direct measure of public benefit.

The consultant team estimated a weighting of factors responsible for eutrophication damage as NO_3 (10%), P (70%) and sediment (20%).

2.5 Marine eutrophication

Marine and estuarine eutrophication is mainly considered to be caused by elevated nitrate levels. Hartridge and Pearce (2001) quote a study by Turner *et al*. (1999) as the only one available that has addressed the welfare costs of coastal and marine eutrophication. However, this was a study of nutrient fluxes and pollution in the Baltic to which the UK is not a direct contributor of pollution. Hartridge and Pearce regarded the estimates of WTP by the Swedish and Polish populations for a 50% reduction in pollution as unrealistically high and unusable in a UK context. The EA (2005) were unable to find any estimates of the damage cost of marine eutrophication.

We are therefore unable to quantify the benefit from reduced marine eutrophication.

2.6 Bathing water quality

Contamination of marine bathing waters with microbial pathogens increases the risk of disease and hence reduces the quality of the bathing experience. Mourato *et al*. (2002) estimated that there were around 1.3m cases of stomach upset in England and Wales associated with faecal contamination of bathing waters.

A number of studies have elicited the WTP for varying degrees of bathing water quality improvement in Europe. The main studies were by Georgiou *et al*. (1998) who found a WTP of £13.90 per household per year to improve bathing water quality to mandatory standards at two beaches in East Anglia. Scaling to the number of households in England and Wales (22.5m) (DCLG, 2006) gives a WTP of £312m per year. A later study (Georgiou *et al*., 2000a) found a WTP to reduce the incidence of gastro-enteritis of £21-£22 per person per year. A revised WTP figure of £4.2 per household per year is given by Georgiou in EA (2003).

More recently, Mourato *et al*. (2002) used a choice experiment to elicit the WTP of households for a reduction in the risk of a stomach upset from sea bathing. Respondents were willing to pay between £1.10 and £2.00 per household per year for this 1% reduction. Scaling to the number of households in England (21.6m) (DCLG, 2006) gives a WTP of £23.8-£43.2m per year per 1% improvement. Based on the work of Eftec (2002), Eftec (2004) estimated a benefit of £61.8m per year from achievement of Bathing Water Directive Standards in England and Wales. An equivalent benefit for England re-based at 2006/07 prices would be £64m per year. The Georgiou *et al*. (1998, 2000a) estimates were much higher. It may be that there are embedding[8] effects in the Georgiou (1998) study that would preclude its use for aggregation. EA (2003) exclude it for this reason and preferred the Eftec study for aggregation.

EA (2005) estimate that 35-65% of faecal pathogens in bathing waters were attributable to agriculture. Using the Eftec (2002, 2004) estimate, this gives a maximum benefit from agricultural measures to reduce FIOs of between **£25 and £45m** per year in England. Given the

[8] Embedding (part-whole bias) is a problem in contingent valuation where the value of a good depends on whether it is valued as part of a wider, more inclusive group of goods, or on its own (Garrod and Willis, 1999). Surveys on small areas aggregated up are potentially subject to bias unless the issue of embedding is addressed in the design.

Georgiou results, this may appear to be an underestimate of the potential benefit. However, recent work by ADAS/CREH/IGER (Defra project ES0140) estimated that in 3 problem bathing water clusters in England, that on average agriculture was responsible for 30-40% of non-compliance events, so the EA figure of 35-65% may be a small overestimate.

2.7 Ecosystems, habitats and biodiversity – rivers

Willis and Garrod (1996) derived a WTP for improvements in river water quality of £0.00217 per household per km of river length per year for an improvement from medium (RE3) to Good (RE2/1) quality. The WTP for a change from RE4 (poor) to RE3 (medium) was £0.0065 per km. The EA (2005) derived their damage estimate from these water quality gains by aggregating gains over the length of rivers in England according to their current RE status. They produced an estimate of the total non-use value of improved environmental quality to RE3 status. For England and Wales, the estimate was £1,486m per year at 2006/07 prices. EA (2005) estimated that 25-35% of ecosystem damage was attributable to agriculture. The potential benefit from agricultural measures can thus be estimated as **£370-518m** per year.

In the weighting of benefit between changes in NO3, P and sediment levels, the consultancy team estimated that the split was 20%, 40% and 40%, respectively.

2.8 Ecosystems, habitats and biodiversity - wetlands

Turner *et al*. (2005) discussed the various methods that may be used to value wetlands. They list 25 valuation studies that relate to wetlands of which five referred to the UK. However, no synthesis of the empirical studies was made. The UK studies referred to wetland landscape and land use preservation, flood prevention and wetland purchase for conservation, none of which are of direct relevance to the benefits from pollution reduction on wetlands. They do, however, suggest that the public have a WTP for the conservation of wetlands and that this mainly consists of non-use value. Willis (1990), for example, found that the public were willing to pay over £500 per ha to preserve the current state of some wetlands. Ninety-five per cent of this was a non-use value. This is different, however, from the public's WTP to improve current wetland ecosystems using, agricultural and other measures.

EA (2005) estimate the benefits from PRO4 in reducing damage to wetlands from agricultural abstraction and pollution. With agriculture estimated to account for 25-35% of the damage, the potential benefit was **£13-42m** per year (2006/07 prices). A part of this relates to improvement in flow, but it was not possible to isolate flow effects from quality (NO_3, P, FIO, sediment) effects. The benefits may therefore be overestimated. In terms of the partition of effects the team assessed that 40% of the benefit was due to changes in P levels and 60% to changes in sediment.

2.9 Conclusions on potential primary benefits

Table 2.7 summarises the benefit estimates from agricultural measures to improve water quality. The main benefits are from reduced freshwater eutrophication and improved riverine ecosystems and habitats. Reductions in nitrate levels mainly impacted on drinking water. Changes to FIOs mainly impacted on bathing water quality. Benefits from P and sediment reduction were more widespread and potentially more valuable in terms of public benefit. However, it should be borne in mind that many of the aggregate estimates were based on the transfer of values from localised studies and therefore could be subject to substantial error.

Table 2.7: Estimates of benefits from improved water quality due to agricultural measures in England (£m per year at 2006/07values)

Water quality benefit category	**Benefit from agricultural mitigation (2006/07 values) (£m per year)**	**Weighting attributed to primary pollutants (% of total)**			
		NO3	P	FIO	sediment
Drinking water quality (surface and groundwater)	13-71	100	0	0	0
Improved river water quality (amenity)	15-37.5	10	40*	10	40
Improved fishing	15-37	10	10	0	80
Freshwater eutrophication	168-330	10	70*	0	20
Marine eutrophication	Not available				
Bathing water quality	25-45	0	0	100	0
Ecosystems, natural habitat impacts – rivers etc	370-518	20	40*	0	40
Ecosystems, natural habitat impacts – wetlands	13-42	20	40*	0	40

Note all adjusted to 2006/07 values using the GDP deflator.

* These weights were reduced to 15.7% of the asterisked values in the table to reflect the expectation that policy options that reduced P losses would only be effective in this proportion of cases (see 2.2.2).

Table 2.7 also lists the weightings given to individual pollutants with regard to their assessed effect in delivering benefits. As discussed previously, the weighting can be interpreted as the % of cases where the pollutant is the limiting factor in determining the environmental damage. It is the % of cases where a marginal reduction in the level of that pollutant will deliver a public benefit.

3. Estimates of potential secondary benefits

3.1 Introduction

The study also quantified secondary impacts of DWPA measures. Quantitative assessments were carried out of the impacts of DWPA measures on ammonia and greenhouse-gas emissions (CH_4 and direct and indirect emissions of N_2O). The greenhouse gas effects include an estimate of the greenhouse gas implications of changes in energy use brought about by the agricultural measures. These energy effects were added to the gaseous emissions and expressed in CO_2 equivalent terms. A detailed assessment of these secondary impacts can be found in Appendices 1-4.

Qualitative assessments were also made of the impacts of individual DWPA mitigation methods on changes to soil quality and biodiversity, and losses of BOD, NO_2, NH_4. These assessments can be found in Appendices 2-5.

3.2 Greenhouse gas emissions (climate change)

It is now well established that greenhouse gas emissions damage the global environment. Economics research has attempted to quantify the cost to global society of these emissions and hence the benefits from emission reduction. Damage costs have not been derived directly from the public's willingness to accept damage, or willingness to pay for mitigation. Rather, estimates have been made from indirect modelling of the cost of global impacts (Helm, 2005).

Under the Kyoto Protocol, and subsequent agreements amongst EU countries, the UK has a commitment to reduce greenhouse gas emissions by 12.5% below the base year[9] level, on average over the first commitment period, 2008-2012. There is also a manifesto commitment aiming to reduce CO_2 net emissions by 20% by 2010. In addition, following the publication of the Energy White Paper in February 2003, the UK has a longer-term goal to put the UK on a path to reduce carbon dioxide emissions by 60% by 2050, with real progress by 2020.

Carbon sequestration and release is accounted for in the UK Greenhouse Gas Inventory which reports greenhouse gas emissions by source and removals by sinks. The UK Climate Change Programme is currently being reviewed to determine the action needed to put the UK on track to meet its domestic 20% target.

3.2.1 Value of sequestered carbon: damage estimates

The social cost of carbon is the monetary value of worldwide damage from the anthropogenic emission of carbon dioxide into the atmosphere. The models used to estimate the social cost of carbon relate emissions to atmospheric changes; atmospheric change to temperature change; and temperature change to damage. This damage includes sea level rise, floods and storm events, the impact of climate change on agricultural production, and effects on population health and disease. Given the uncertainly involved in the impacts it is not surprising that estimates of the social cost of increased carbon emissions vary widely depending on the assumptions used, including the discount rate. Marginal damage costs depend strongly on the choice of discount rate, because they reflect the additional future damage from small changes in current emission (Tol, 2005). The models also vary in the weight given to extreme and catastrophic events.There is also disagreement on how to deal with spatial variation in the impacts of global warming, and especially how impacts on people with different incomes are treated. Results are sensitive to the way in which equity weighting (the weights given to people of different income levels) is applied. These issues are discussed at length by Pearce (2005).

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[9] 1990 is the base year for emissions of CO_2, CH_4, and N_20 and 1995 is the base year for emissions of HFCs, PFCs and SF_6.

A study by Clarkson and Deyes (2002) for H. M. Treasury estimated the social cost of carbon at £70/tC (with a range of £35 to £140/tC) and this is the value that Defra and other government departments currently use in appraising policies that lead to changes in carbon emissions. They were used, for example, in the compilation of environmental accounts for agriculture made by Entec (2004). It is suggested that these estimates should be increased by £1/tC per year in real terms to reflect the increasing costs of climate change over time.

This value is considerably higher than estimates made by most other economists. The meta analysis undertaken by Tol (2002a, b) of 103 estimates of marginal damage costs of carbon dioxide emissions; produced a mode of $2/tC, a median of $14/tC and a mean of $93/tC, because of the lognormal distribution of the cost estimates. Studies with lower discount rates and equity weighting had higher estimates and much greater uncertainties, whilst peer reviewed studies had lower estimates and smaller uncertainties. Tol concluded that climate change impacts were very uncertain, but that using standard assumptions about discounting and aggregation, a marginal damage cost of carbon dioxide emissions of $15/tC (i.e. £8/tC) seems justified and that marginal damage costs were unlikely to exceed $50/tC (i.e. £27/tC). Mendelsohn (2005) concluded that the social cost of carbon would rise over time, but there was every reason to expect that it would remain below $10/tC (£5/tC) for the next 30 years. Pearce questioned the Clarkson and Deyes (2002) estimate and concluded that it was too high, preferring a marginal social cost of carbon of £4-27/tC.

An alternative cost-based approach to estimating the marginal cost of carbon is to assess the marginal cost to government of current measures to control emissions. Pearce (2005) derived a range of values from £16-31/tC depending on the measures and quoted a Defra estimate of the cost of controlling carbon emission at £45/tC. According to Pearce, the evidence from policy intervention to control carbon emissions is that the £70/tC cost is not currently applied in a consistent way in the policy arena. Defra is currently undertaking a review of values of the social cost of carbon, but for this study we have used the government's preferred estimate of £70 per tonne C.

3.3 Ammonia emissions

3.3.1 Effects

Ammonia emissions have two main impacts on the environment - acidification and nutrient enrichment. As a source of nitrogen, ammonia can increase soil fertility and cause damage to ecosystems that exist because of naturally low levels of soil nitrogen. It can also contribute to the eutrophication of water bodies.

- Deposition of acidifying pollutants (including NO_x and NH_3) has a range of detrimental effects (Defra, 2001), These include:
- Increased acidity of freshwaters which has led to the loss of fish and other organisms from many rivers and lakes;
- Increased acidity of soils which alters soil chemistry and can leave forests and other terrestrial ecosystems vulnerable to drought, disease and insect attack;
- Acidic groundwaters which damage water supply infrastructure and increase the levels of harmful metals in drinking water;
- Eutrophication (excess nutrient enrichment of natural ecosystems) from deposition of the nitrogen-containing atmospheric pollutants NO_x and NH_3. Over half the ecosystems in the UK exceeded critical loads for eutrophication in 1990, although this value is expected to fall to 29% by 2010 (27,000 km^2).

Ammonia emissions are also associated with secondary particulate formation and have implications for human health. However, this aspect of impact was not included in this study because we were unable to locate information on the effect of changes in ammonia emissions on the relative risk of human health and morbidity as a result of exposure[10].

3.3.2 Cost and benefit estimates

Valuing a marginal change in NH_3 output is most readily done in terms of the cost (at the margin) of current or possible future measures to reduce NH_3 output. This is a more feasible measure of benefit than damage costs or WTP-based measures of benefit. A range of policy measures are in place under Defra's UK Air Quality Strategy (Whitfield and Bareham, 2005). Since agriculture is responsible for around 80% of ammonia emissions, we use the marginal cost of agricultural measures to reduce NH_3 as a measure of public benefit.

The Defra NARSES project (ADAS, 2004) examined the costs and effectiveness of a wide range of agricultural mitigation methods, and produced a cost curve for ammonia reduction. The most cost effective methods were covering of slurry stores, immediate incorporation of pig slurry/cattle slurry/poultry manure into arable land by disc cultivation and, allowing cattle slurry lagoons to crust over. The lowest cost measures had costs in the broad range of £0.2-1.0 per kg NH_3-N. Costs were based on the increased expenditure of farmers and therefore underestimate the actual exchequer cost since policy transaction costs are not included. However, they provide an indication of the cost saving from reductions in NH_3 output that DWPA measures may deliver.

Pretty *et al*. (2001) estimated a marginal cost for NH_3 of £171 per tonne (£0.159 per kg NH_3-N at 2006/07 values). Eftec (2004) used a damage cost range of £87-270 per t NH_3 (£0.08-0.25 per kg NH_3-N at 2006/07 prices). These are at the lower end of the ADAS estimates and appear to equate to the cost of inorganic nitrogen fertiliser of ca. £150/t.

We used a benefit range of £0.2-1.0 per kg NH_3-N to price marginal changes in NH_3 output. There may be some double counting of eutrophication effects since the public's WTP for reduced eutrophication of water bodies has already been accounted for (see above). However, any double counting is likely to be small, as freshwater eutrophication by NH_3 deposition is likely to be small and very localised.

[10] The lack of information on this aspect is likely to have on a minor impact on the total benefits of the selected policy options because they did not have major secondary impacts on ammonia emissions from agriculture.

4. Benefits from mitigation measures in agriculture

4.1 Calculation method

So as not to exaggerate the benefits of policy instruments, it was necessary to determine what degree of benefits is likely to occur as a result of existing policies (e.g. NVZs and Single farm Payments) and advice (e.g. via the ECSFDI), between the years 2000 and 2015, prior to any implementation of additional advice, schemes, grants or policies. Therefore, the primary and secondary benefits were estimated using information of potential uptake and efficacy of a specific combination of methods (from Anthony, 2006) and using information on potential changes in land use and animal numbers (Business As Usual – BAU Phase 2 report, 2006).

Briefly, three potential supportive approaches to method implementation were investigated. These were:

- Advisory
- Grant
- Scheme

The 'Advisory' approach was defined as the existing England Catchment Sensitive Farming Delivery Initiative (ECSFDI), but implemented across the whole country. It was assumed that low cost mitigation methods would be targeted by the advice provided by expert advisors, and would exclude high cost (with little return) methods. This approach therefore focused on improvements in agricultural practice, such as fertiliser application timing and fully accounting for manure nutrient value.

The 'Grant' approach was defined as a 40%, standard cost based, contribution to the implementation of high cost mitigation methods, such as increased manure (slurry) storage and fencing to exclude livestock from streams/river.

The 'Scheme' approach was defined as an extension of the existing ELS and HLS schemes, by provision of additional water protection options. Advice to support the schemes would be available as at present through the ADAS-led Conservation and Environment Sensitive Farming programmes, and via Natural England staff for HLS applications – this would be at a much lower level than the advice available under ECSFDI.

The first policy instrument defined by Defra was the Water Protection Zone (WPZ). This was a generic term used to indicate that the instrument would include all mitigation methods necessary to protect water quality, though it was accepted that the list of methods might vary between catchment areas, dependent on the level of pollution, environment conditions and type of agriculture. This instrument was defined by Defra as potentially including all available mitigation methods that had an impact on phosphorus loss. The instrument specifically excluded some mitigation methods (see final report for ES0203). The policy instrument was set up so that sufficient methods were implemented to achieve a 48% reduction in phosphorus losses from each of the agricultural sectors (WPZ). For comparison, a less stringent policy instrument was set up to achieve a 30% reduction in phosphorus loss (WPZ30).

The second policy instrument defined by Defra was a Financial Instrument (FI). This was based on financial incentives to reduce individual farm phosphorus budget surpluses (further details are provided in Shepherd *et al.*, 2006).

Five options were evaluated in terms of the benefits delivered over and above Business as Usual (**BAU**). These were:

Water Protection Zone (WPZ)

WPZ + scheme

WPZ+ advice +scheme

WPZ30

Financial instrument (FI)

Financial instrument + advice (FI + advice)

4.1.1 Primary pollutants

The primary benefits were estimated using the methods outlined earlier. The impacts of each option in reducing losses of NO_3, P and sediment were calculated by Anthony (2006) for each farming sector (arable, dairy etc.). This indicated the % reduction in loss due to each policy measure. This was applied after adjustment for the assimilative capacity for P (see Section 1.2). To estimate the impact on the potential benefit to the public a pro rata approach was used. In this a 1% reduction in the level of agricultural pollutant emission (above assimilative capacity) was assumed to deliver 1% of the total potential benefit given in Table 2.7.

Estimates of the benefits from reduced P emissions were made on the basis that only reductions affecting river GQA class 4 (27.5% of P damaged catchments) will be effective in providing public benefits (see Section 2.2.2 and Table 2.7). This was based on the rationale that GQA river classes 1, 2, and 3 are already of good P status, i.e. <0.1 mg/l P, and that river classes 5 and 6 have P concentrations that are so far in excess of the 'trigger' value of 0.1 mg/l P (Table 2.5), that even a 50% reduction in P load would not result in the required reduction in P concentration.

Where only one pollutant (e.g. NO_3) is relevant, as in the case of drinking water quality, the benefit from reduced NO_3 emissions was calculated by multiplying the proportionate reduction in loss from agriculture (from Anthony, 2006) by the public benefit (Table 2.7). Hence a 10% reduction in NO_3 emissions would deliver a lower bound estimate of benefit of £(0.1*13)m for drinking water quality. This is £1.3m per year, 10% of the lower bound potential benefit of £13m per year (Table 2.7). The upper bound estimate is £7.1m per year

The effect on river water quality is more complex because we assume NO_3 is only responsible for 10% of the benefit at national level (Table 2.7). (This could be interpreted as NO_3 being the limiting pollutant in 10% of locations). Hence the lower bound estimate of benefit from river water quality improvement attributable to a 10% reduction in NO_3 emissions is £(0.1*0.1*15). This is £0.15m per year. The upper bound estimate is £0.375m per year.

This process is applied to each quantified primary pollutant (NO_3, P, sediment) in each category of water quality improvement for which benefit estimates were available (Table 2.7).

4.1.2 Secondary effects

Secondary effects on greenhouse gases and ammonia emissions were estimated directly, using the marginal benefit data given in Sections 3.2 and 3.3. Hence a package that reduced greenhouse gas emissions by the equivalent of 1t of CO_2-C per year was valued in terms of secondary benefits at £70. A package that reduces NH_3-N output by 1kt per year delivered a secondary benefit valued at £0.2-£1.0m per year.

We anticipate that any impact of policy measures on commodity prices would be extremely small and can be ignored. Policy measures may, however, reduce farm output depending on the options considered. Care needs to be taken in interpreting any associated reductions in greenhouse gas emissions because farm production may be displaced elsewhere. There is thus a case for discounting reductions in greenhouse gas outputs where these are achieved by measures that primarily achieve their effect by reducing domestic farm output.

4.2 Benefit estimates

Results for BAU and the five options are given in Table 4.1. Lower and upper bound estimates relate to the benefit ranges given in Table 2.7. The costs listed in Table 4.1 were calculated using the cost-effectiveness calculations described in project ES0203 (Anthony, 2006). In that study, Anthony (2006) calculated the national cost of implementing the range of mitigation methods invoked by each of the policy options and policy support approaches. Assuming that benefits flow immediately from the application of the instruments, we can draw a number of conclusions from Table 4.1.

Table 4.1 gives the estimate benefits for each policy option in £m per year. The table also gives information on the benefits from reductions in each pollutant (NO_3, P and sediment). Table 4.2 presents the results by type of primary and secondary benefit.

BAU delivers significant benefits of between £224m and £288m (excluding benefits to bathing waters by reduction in FIO transfers, Table 4.1). **WPZ** delivers benefits over and above those from BAU of £115-£169m per year. These derive mainly from P and sediment reduction, although there are significant benefits from reduced nitrate losses. Secondary effects make only a small contribution.

WPZ + scheme delivers slightly higher benefits but **advice** contributes little additional benefit. **WPZ30** has a relatively low cost but delivers substantial benefits. The cost is £16.4m per year and the benefit range is £69-£109m. The **FI** had little impact, with benefits of only £1m-2m, and **advice** given in addition to FI again produced little in the way of additional benefits.

The upper bound estimates exceed the cost of the options in all cases expect that for **FI + advice**. In most cases the lower bound estimate of benefit was below the policy costs. Where **advice** was part of the policy package benefits did not cover costs.

Overall, only the WPZ30 gives a high ratio of benefit to cost (range 4.2-6.6). The others give rather low and unsatisfactory benefit : cost ratios even when the upper bound benefit is used. In interpreting the results it must be noted that benefits in Table 4.1 are likely to be slight underestimates in that benefits from reduced FIOs (which mainly impacts on bathing water quality) and from reduced marine eutrophication were not included in the table. A separate study WT0713 (Application of the FIO-SA model to poor bathing waters and shellfish waters) has, however, generated information that we discuss below.

We estimated at a national level that BAU would result in an approx 25% reduction in livestock numbers which would reduce the FIO loading to water by 25% at the national scale and hence would result in a 25% benefit in terms of bathing waters. Using the value in Table 2.7, the additional benefits would amount to between £6.3m and £11.3m. Project WT0713 specifically assessed the relative impacts of **BAU** and policy instruments on FIO transfers at the national scale. It concluded that the decrease in animal numbers and uptake of some management practices by 2015, i.e. the Business as Usual scenario, would result in a decrease in the average FIO loss by up to 26% (a similar value the one estimated using the simple approach described above). This would therefore accrue additional benefits of between £6.5m and £11.7m (using the values in Table 2.7)

Project WT0713 showed that the average FIO loss as a result of **BAU+WPZ** was decreased by up to 41% (resulting in additional benefits of between £10.3m and £18.5m). **BAU+WPZ+Scheme** had no additional effect on FIO losses. Whereas

BAU+WPZ+Scheme+Advice reduced average FIO losses by up to 43% (resulting in a additional benefits of between £10.8m and £19.4m).

Clearly, many of the reduction in FIO losses would also improve shell fish water quality, however, we have not taken this into account in this cost:benefit analysis.

Moreover, it should be noted that both the costs and benefits for the **WPZ** were estimated using a general, and non-targeted approach. At Defra's request, the costs were estimated based on integrating a set of mitigation methods to reduce P loads to water by an average of 48% across all river courses, irrespective of water quality. The benefits analysis therefore took the same approach. In reality, should a more targeted approach be taken, e.g. to the class 4 rivers (classes 1, 2 and 3 already have P concentrations below the critical threshold, classes 5 and 6 have concentrations far in excess of the critical P threshold concentration – Table 2.5), then the benefit : cost ratio would be improved in these catchments. A targeted approach would be the next step in the research.

All estimates are subject to uncertainty. In relation to the welfare benefits this is captured through the use of upper and lower bounds in the size of the potential benefit (Table 2.7). Uncertainty in the underlying physical data and is discussed in the Appendix 6.

4.2.1 Timeframe of benefits

For nitrate, if N inputs were reduced significantly then it would be reflected in reduced N loads to water relatively quickly (probably within 1-2 years). Nitrate has little impact on the eutrophication of freshwater rivers and lakes, so the impact on benefits would be limited to nitrate removal costs from drinking water which would be seen within the same timeframe and would impact on operating and future capital investment costs. The same could also be said for FIOs, if the source was reduced.

For P, if P inputs were reduced significantly, it could be decades before levels in rivers decreased. We believe that reduction in diffuse P losses from agricultural land will have little impact on the main river reaches due to the importance of point source contributions that already cause exceedance of concentration thresholds. Indeed, Jarvie *et al* (2006) have concluded that the over-riding evidence indicates that point rather than diffuse sources of phosphorus provide the most significant risk for river eutrophication at times of risk, even in rural areas with high agricultural total phosphorus losses. Discharges from even quite small settlements might overwhelm the diffuse source contribution. However, it is possible that agricultural sources are significant in low order river systems and relatively unpopulated catchments. Assessment of the potential for ecological improvement following reduction in agricultural inputs requires an improved assessment of relative point and diffuse source contributions within catchments and their bioavilability in aquatic systems.

Also, we do not know what the transient storage is in these systems, but can envisage that whilst there may be some immediate benefit, the total impact would take at least 10+ years to come through. In conclusion, we are uncertain as to the rate of decline in P loads as a result of the decrease in agricultural P inputs and do not feel it appropriate to 'guestimate' this in the current report. So, although we recognize that most P impacts will not be immediate and could take decades to manifest themselves, due to a lack of information on the timeframe for benefits, in this study we have had to assume 'immediate' benefits and cost accordingly, i.e. assuming a zero discount rate.

Table 4.1 Estimates of public benefits from the policy options (£m per year)

	Primary effects						**Secondary effects**			**Total benefits**		**Cost**
	Nitrate-N		**P**		**Sediment**		**NH3-N**		**CO2-C equivalent**			
	LB (£m)	**UB (£m)**	**LB (£m)**	**UB (£m)**	**LB (£m)**	**UB (£m)**	**LB (£m)**	**UB (£m)**	**(£m)**	**LB (£m)**	**UB (£m)**	**(£m)**
BAU	19.6	40.1	15.9	27.2	22.5	36.7	4.4	21.9	161.8	224.1	287.7	272.4
Policy option												
WPZ	15.8	32.4	23.6	40.4	56.3	75.0	0.5	2.7	19.0	115.3	169.5	136.3
WPZ +scheme	16.1	32.9	23.6	40.4	59.1	96.6	0.5	2.3	20.4	119.6	192.5	138.7
WPZ+ advice + scheme	16.7	34.1	23.5	40.3	59.6	97.4	0.5	2.4	22.2	122.5	196.3	193.9
WPZ30	9.9	20.3	15.5	26.5	34.8	54.2	0.0	0.0	8.4	68.6	109.4	16.4
FI	0.0	0.0	1.4	2.3	0.0	0.0	0.0	0.0	0.0	1.4	2.4	na
FI+advice	0.7	1.5	2.2	3.7	2.3	3.8	0.2	0.9	5.1	10.5	14.9	43.8

Note; LB is lower bound estimate, UB is upper bound estimates.

NB. Benefits from the policy options are over and above those derived from BAU. Costs of the policy options are over and above those for BAU. Source of costs Anthony (2006)

na = the cost of the FI was not assessed.

Benefits do not include FIO reductions and benefits in bathing water quality. These are described in the text.

Table 4.2 Estimates of public benefits from the policy options (£m per year) (by primary and secondary benefit)

	BAU	BAU	WPZ	WPZ	WPZ +Scheme	WPZ +Scheme	WPZ+ advice +scheme	WPZ+ advice +scheme	WPZ30	WPZ30	FI	FI	FI+advice	FI+advice
	LB	**UB**	**LB**	**UB**	**LB**	**UB**	**LB**	**UB**	**LB**	**UB**	**LB**	**UB**	**LB**	**UB**
Primary Benefits														
Drinking water quality (surface and groundwater)	2.3	12.7	1.9	10.3	1.9	10.4	2.0	10.8	1.2	6.4	0.0	0.0	0.1	0.5
Improved river water quality (amenity)	1.3	3.2	2.4	6.1	2.5	6.1	2.5	6.2	1.5	3.7	0.0	0.1	0.1	0.3
Improved fishing	1.9	4.7	4.1	10.1	4.1	10.2	4.2	10.3	2.5	6.1	0.0	0.1	0.2	0.5
Freshwater eutrophication	13.3	26.2	19.5	38.2	22.0	43.2	22.1	43.5	13.4	26.3	0.6	1.1	1.4	2.7
Marine eutrophication	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Bathing water quality	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ecosystems, natural habitat impacts – rivers etc	37.8	53.0	65.0	76.8	66.0	92.4	66.7	93.4	40.2	54.0	0.7	1.0	3.3	4.6
Ecosystems, natural habitat impacts – wetlands	1.3	4.3	2.3	6.2	2.3	7.5	2.3	7.6	1.4	4.4	0.0	0.1	0.1	0.4
Secondary Benefits														
NH_3	4.4	21.9	0.5	2.7	0.5	2.3	0.5	2.4	0.0	0.0	0.0	0.0	0.2	0.9
CO_2-C equivalent	162	162	19	19	20	20	22	22	8	8	0	0	5	5
Total	224.1	287.7	115.3	169.5	119.6	192.5	122.5	196.3	68.6	109.4	1.4	2.4	10.5	14.9

5. Future research recommendations

Targeting of mitigation measures

The recent suite of Defra projects has adopted a non-targeted approach to assessing the cost-effectiveness and cost-benefits of individual mitigation methods , policy instruments and supportive approaches to tackle DWPA. A targeted approach is now required, particularly to class 4 waters (GQA classification), since classes 1, 2 & 3 are below the critical P threshold for eutrophication, and classes 5 and 6 may require significantly greater than 50% reductions in P loads to achieve acceptable P concentrations.

The costs-effectiveness and cost-benefits of targeted approaches will need to be determined. For example, an assessment of the impacts of CSF delivery is required on individual or regional groups of catchments.

The impact of policy measures on bathing water quality has been conducted on ‘poor’ bathing waters, this now needs to be extended to those bathing waters classified as ‘sufficient’. Impacts of targeted mitigation measures on shell fisheries now need to be determined.

Impact of Mitigation Methods on Water Quality Status

There is a requirement to determine the impact of diffuse pollution mitigation methods on ecological status. The focus should be on an ecological life-cycle analysis of indicator species to identify sensitive periods. This could be achieved through workshops and a technical review, with equal contribution from freshwater ecologists and catchment modelers.

Source Apportionment of Nutrient Losses from Urban and Rural Sources

There is compelling evidence indicating that point rather than diffuse sources of phosphorus provide the most significant risk for river eutrophication at times of risk. Further research is required to enhance our ability to identify the importance of agricultural and non-agricultural source of nutrients (especially P), sediment and FIOs in targeted catchments, taking into account a better representation of the range of urban diffuse pollution source areas (parks, residential and industrial areas, road surfaces and pavement etc) and attributes (population density, traffic density, SUDS implementation).

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9. This section should be used to record links (hypertext links where possible) or references to other published material generated by, or relating to this project.

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