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Technical Report on Water Quantity and Quality in Europe: an integrated economic and environmental assessment

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Abstract

The economic assessment of priorities for a European environmental policy plan focuses on twelve identified Prominent European Environmental Problems such as climate change, chemical risks and biodiversity. The study, commissioned by the European Commission (DG Environment) to a European consortium led by RIVM, provides a basis for priority setting for European environmental policy planning in support of the sixth Environmental Action Programme as follow-up of the current fifth Environmental Action Plan called 'Towards Sustainability'. The analysis is based on an examination of the cost of avoided damage, environmental expenditures, risk assessment, public opinion, social incidence and sustainability. The study incorporates information on targets, scenario results, and policy options and measures including their costs and benefits.

Main findings of the study are the following. Current trends show that if all existing policies are fully implemented and enforced, the European Union will be successful in reducing pressures on the environment. However, damage to human health and ecosystems can be substantially reduced with accelerated policies. The implementation costs of these additional policies will not exceed the environmental benefits and the impact on the economy is manageable. This requires future policies to focus on least-cost solutions and follow an integrated approach. Nevertheless, these policies will not be adequate for achieving all policy objectives. Remaining major problems are the excess load of nitrogen in the ecosystem, exceedance of air quality guidelines (especially particulate matter), noise nuisance and biodiversity loss.

This report is one of a series supporting the main report: *European Environmental Priorities: an Integrated Economic and Environmental Assessment*. The areas discussed in the main report are fully documented in the various *Technical reports*. A background report is presented for each environmental issue giving an outline of the problem and its relationship to economic sectors and other issues; the benefits and the cost-benefit analysis; and the policy responses. Additional reports outline the benefits methodology, the EU enlargement issue and the macro-economic consequences of the scenarios.

This report is one of a series supporting the main report *European Environmental Priorities: an Integrated Economic and Environmental Assessment*. Reports in this series have been subject to limited peer review.

The report consists of three parts:

Section 1:
Environmental assessment
Prepared by B.J. de Haan, A. Beusen (RIVM), C. Sedee (TME)

Section 2:
Benefit assessment
Prepared by D.W. Pearce, A. Howarth (EFTEC)

Section 3:
Policy assessment
Prepared by D.W. Pearce, A. Howarth (EFTEC)

There are supporting Annexes that present additional information of the methodology and data.
A. Water supply and sectoral demand
B. Waste water treatment
C. Measure to restrict animal husbandry to carrying capacity
D. Methane emission reductions in agriculture.
Annex E demonstrates the impact of the policies contained in the baseline on the potential nitrate concentration in Europe's coastal zones.
Prepared by B.J. de Haan, A. Beusen (RIVM), C. Sedee (TME)

References made in the sections on benefit and policy packages have been brought together in the *Technical Report on Methodology: Cost Benefit Analysis and Policy Responses*. The references made in the section on environmental assessment follows at the end of section 1.

The findings, conclusions, recommendations and views expressed in this report represent those of the authors and do not necessarily coincide with those of the European Commission services.

Contents

1. Environmental assessment

1.1 Introduction

Human activities are placing severe pressures on Europe's water resources. Increasing demand, adverse climate conditions and increasing problems of pollution have focused governments' attention on how water resources are managed. Europe needs to ensure that a stable supply of clean water meets both the needs of society and the natural environment. There are three main threats to the quality of the supplied water:

- Water abstraction may cause problems with respect to low flows in rivers, lowering ground water tables in nature areas, and salt-water intrusion in coastal areas. The subsequent long-time loss of ground water as a natural resource affects drinking water supply, soil quality and biodiversity.
- Ground water is polluted by nitrogen and pesticides leaking through the root zone of the soil. Long residence times make nitrogen pollution now a risk for drinking water of future generations.
- Also, The condition of surface waters deteriorated as manifested by algae growth, periods of oxygen deficits, fish kills, etc. In many areas this poor environmental condition is attributable to enhanced nitrogen and phosphorus loading. Generally, phosphate excess distorts the ecosystems in inland waters, while nitrogen excess damages ecosystems in marine waters. Eutrophication of inland and marine waters was identified in the Dobris Assessment (EEA, 1995) as a major European environmental issue.

This Technical Report analyses the potential of new policy measures for water management at EU-level. Here, the assessment of water management consists of analysing the potential for reducing the pressures. At first, the existing policy measures will be described in the baseline (BL) assessment. The EEA has published this assessment in the State of the Environment Report (EEA, 1999). Next, it is argued that no generic emission reduction targets can be set as the ecological constraints for water management vary in Europe. Comparable data are lacking. Hence, a Technology Driven scenario (TD) and Accelerated Policies scenario (AP) assessment have not been carried out. The 'conclusions' section finalises the section.
Coastal zone water quality is only reflected through the analysis of nutrient overload by diffuse and point sources to rivers and atmospheric deposition. Annex E demonstrates the impact of the policies contained in the baseline on the potential nitrate concentration in Europe's coastal zones.

1.2 Environmental trends and abatement cost

Problem sketch (DPSIR) and related economic sectors

The underlying factor in water stress problems is the absence of owner liability for surface and ground water. Agriculture is responsible for diffuse pollution through run-off water carrying manure and artificial fertilisers (NO_3 and PO_4), that have not been consumed by the crops, entering the main streams and groundwater. Also, ammonia (NH_3) evaporated from animal manure is deposited downwind, upsetting fragile - mostly forest - ecosystems. Urban households and industries feed wastewater treatment plants, which - despite the efforts to remove pollutants - emit BOD/COD, NO_3 and PO_4 into surface water. The impact of the transport sector via NO_x emission and deposition is of minor importance.

Eurobarometer ranks water management as one of the three most serious environmental problems. Experts cite water management also as a serious problem. South European countries rank 'water scarcity and pollution' first. Most probably this reflects water availability rather than pollution. It is ranked fifth by North European countries.

1.2.1 Baseline assessment.

At EU level, the Urban WasteWater Directive and the Nitrates Directive create strong incentives for the Member States to prevent excessive nutrient loading of the aquatic environment. However, the implementation of these directives is behind schedule and should be accelerated. The nature of the directives is discussed below. The proposed Framework water Directive has been excluded from the Baseline assessment.

Full compliance with the Urban WasteWater Treatment Directive - as assumed in the BaseLine - by 2010 will remove most of the problems of surface water pollution by wastewater. The policy target is to be met in the BaseLine; no additional targets have been set. As no sufficient information was available on the area designated as sensitive by the Member States, the assessment assumes that the whole EU-15 territory is sensitive to eutrophication and the severest requirements of the Urban WasteWater Treatment Directive apply.

The EU Nitrates Directive limits the disposal of manure on agricultural land to 170 kgN/ha to protect ground water and surface water quality. Despite this Directive, the principal driving force of nutrients pollution remains intensive animal husbandry and particularly the pigs and poultry sector, which is concentrated at specialised farms in the regions of Niedersachsen, Nordrhein-Westfalen, the Netherlands, Flanders, Bretagne, Cataluna, and Lombardy. The share of pigs and poultry in these 'hot spots' of intensive livestock production exceeds 50% of total animals. Table C.1 in the Annex on animal husbandry displays the animal density in the 'hot spot' regions. Apart from nutrients from manure disposal, farmers apply artificial nutrients to a level ranging from 100 to 200 kg N/ha. At farm level pigs and poultry holdings may range up to a density of 20 LU/ha[1], thus exceeding the manure disposal limit by far. Compared to cattle, pig and poultry farms face more problems in reducing excess amounts of nitrogen from manure.

Current status of BL policies: policy gap or not

The Nitrate Directive and the Urban WasteWater Directive address the problem of ground water and surface water eutrophication. These directives have been initiated in the 90's; year 2003 awaits their full implementation. Excessive nitrate pollution remains a problem distorting marine ecosystems. These two directives make part of the baseline assessment (EEA, 1999).

Remarkably, there seems to be no directive to limit the phosphate emission of the agricultural sector. Regionally, phosphate saturation of the topsoil has become a major problem. However, the phosphate concentration in EU's main rivers has a decreasing trend. This is probably due to the introduction of phosphate-free detergents. A further decrease is not expected as agricultural pressures remain unchanged and amelioration has long delay times.

Good management of freshwater resources in the semi-arid regions is necessary for the maintenance of the required standards of water quality and quantity, and therefore for reaching the goals of EU policies. The targets for the 5EAP and for the EU Action Programme for Integrated Groundwater Protection (GAP) are to be implemented by the year 2000. One of GAP's main purposes is to integrate the groundwater protection requirements into the Common Agricultural Policy (DGVI) and into Regional Policy (DGXVI). This proved not to be a very effective strategy as GAP currently lacks legal status.

Both water scarcity and quality deterioration affect particularly the southern areas of Europe, where a high percentage of land is used for agricultural purposes and is supplied by groundwater for irrigation with its associated problems of nitrates and pesticides saturation. The 5EAP aims to maintain the water resources, so that the regional balances between demand and supply are guaranteed. Risk management is another aspect, which is targeted by the 5EAP objectives for the EU.

The main sources of risk to human health and the environment related to the semi-arid regions are floods, and forest fires. These are grouped as 'natural hazards', but there are no EU policy targets to reduce these events (ETC-IW, 1996).

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[1] LU is livestock unit: an indicator in which species and classes of livestock are converted to comparable size. For example, a dairy cow counts one LU, while sheep and goats count 0.1 LU, pigs count 0.25 LU, and poultry 0.0125 LU. 1 LU/ha can roughly be considered as equivalent to a manure disposal of 80 kgN/ha.

There are no BL policies directed to prevent ground water overexploitation, although sometimes, national laws set limits in relation to the natural recharge by precipitation. For example, Denmark limits ground water abstraction to 25% of the natural recharge.

Eurostat reviewed European regions where water shortages are known to occur and the estimated water demands based on an inventory among national governments
(Eurostat, 1998). Irrigation makes the largest part of Europe's water demand. A small decrease in the demand has been projected (EEA, 1999)

With respect to coastal zone management, the assessed indicator is nitrate load, though no absolute target can be set. The relation between nitrate load and good bathing water quality could not be defined. The North Sea Action Programme defined the policy target 'halving the load to coastal water'. Other catchment areas are yet to follow. As agricultural emissions constitute the largest burden for coastal sea loads, an agricultural nitrate load indicator seems to be most appropriate of all indicators.

Several other pressures threaten the quality of Europe's coastal zones, particularly, urbanisation and the development of sports and leisure facilities. Though several EU-directives call for good quality water quality for fish and shellfish, overfishing remains the largest threat to the natural balance of marine biota (EEA, 1999). Neither land use planning or overfishing has been part of this assessment.

Spillover
There is some spillover from and to the problem of acidification and eutrophication marine biodiversity would greatly benefit from actions against the excessive nitrogen inputs to surface water. Soil degradation is often associated with overexploitation of natural ground water resources.

Subsidiarity
Two reasons favour the subsidiarity of the problem solution. Firstly, the main ongoing pressure is a) water-thirsty crop production and b) intensive animal husbandry, both of which are concentrated in only a few European regions. Secondly, the varying soil properties make specific regional policy targets more effective than generic ones. Hence, regional authorities can best handle the acceleration of the current policies.

Sustainability
It is difficult to set generic policy targets based on ecological constraints. The proposed EU Framework Water Directive (COM(97) 49 final) (FWD) will provide legal status to water management policies by defining constraints based on local conditions. This requires an EU-wide inventory of local water conditions. Although many monitoring systems are in operation nationally, these need to be harmonised. On behalf of DGXI, EEA is establishing its EUROWATERNET. This monitoring and information network for inland water resources will allow the FWD to become effective (EEA, 1998). Several existing directives will be enforced under the framework. GAP's main requirements can now be found in the FWD will providing legal status. The FWD will also encompass the nitrates and urban wastewater treatment directive. The FWD does not make part of the baseline assessment as it was agreed that the final cut-off date for existing and proposed EU policies would be August 1997.

Recommendation
Water management is still a prominent European problem. The current policies are being implemented and the results are underway. Particularly, the eutrophication of marine waters and water abstraction for irrigation needs further attention. While pursuing the compliance of the current nitrate and urban waste water treatment directives, the Commission might want to accelerate its policies by introducing – in the relevant regions - a compulsory bookkeeping of irrigation water abstraction, nutrients and pesticides at farm level to stimulate good agricultural practises and enable control.
The introduction of the EUROWATERNET monitoring system should alleviate data problems in the near future. It is recommended to ensure that an associated assessment model tailored to the data and policy objectives should be developed.

1.2.2 Assessment methodology

1.2.2.1 Water demand

The water resources available to satisfy the European water demands are restricted by social-economic and environmental factors. The preferred source for water supply is ground water as it has good and stable quality. Most water abstracted, however, is surface water. Where it is abundant, it suitable for irrigation purposes. Ground water is the main source for drinking water.
Ground water demands are increasing and so are the environmental concerns as to water withdrawal. As a consequence, some regions in Europe have water problems. The preferred source of water for public water supply, small industrial water needs and the irrigation of private plots, is groundwater. Groundwater recharge and also the total of locally available water resources are governed by net precipitation, being the difference of precipitation and the actual evapotranspiration. Simple mathematical relations can be used to quantify potential and actual evapotranspiration. The difference between the two represents the existing agricultural water needs. We applied a GIS approach to quantify water needs for public and industrial water supplies and water needs for irrigation in the various European regions. Comparing calculated water needs and the availability of groundwater and local surface water, based on net precipitation, indicates areas where water shortages may be expected and where a minimum amount of water will be needed. Those areas generally correspond relatively well to areas with water problems as indicated by an inventory carried out among European national governments.

Ground water overexploitation for irrigation is probably a regional scale problem. However, the ETC-IW database supplies only national trends of water abstraction. Hence, the real problems will often remain hidden in the statistics. A secondary problem is that there seems to be no well-defined policy target as yet. The GIS assessment method analyses precipitation, evaporation, ground water recharge and surface flow on a very fine grid of 10’ by 10’ or 10 km x 20 km. The results show the clear North South divide in water availability. However, there is a large variability within the Mediterranean countries as well.

It is important to note that water demand should not be met regardless the true cost of supply. Probably, a great deal of the irrigation uses of water in the EU are uneconomic, i.e. they would fail a test that consumer’s ‘willingness to pay’ for water supply exceeds its true cost (Pearce, 1999). Correct water pricing is the preferred mechanism for bringing supply and demand into balance.

1.2.2.2 Waste water

The assessment methodology analyses the Member States 1995 waste water treatment and consequently updates this current treatment state (EEA, 1999) to the required state according to the Urban Waste Water Treatment Directive by 2010. We assume that the whole EU territory will be regarded as sensitive to eutrophication and hence the severest requirements are imposed. Not all Member States did assign sensitive areas as yet. Quite a few already assigned their whole territory though.
The urban wastewater treatment directive differentiates between rural areas and conglomerations. In rural areas, agglomeration smaller than 2000 inhabitants, connection to waste water treatment plants will not be obligatory, while larger conglomerations have to ensure sewerage connection and treatment of waste water by either biological or chemical nutrient removal. The following table reviews the gradual implementation of these requirements:

Table 1.1 Requirements urban waste water treatment directive

	Non-sensitive areas		Sensitive areas			
Agglomeration (x 1000 inh.)	Sewerage connection before	Secondary treatment before	Sewerage connection before	Secondary treatment before	Allowed maximal concentration in drain water (mg/l)	
					N	P
< 2	Not req.	Not req.	Not req.	Not req.	Not sp.	Not sp.
2 –10	2006	Not req.	2006	2006	Not sp.	Not sp.
10 – 15	2006	2006	1999	2006	15	2
15 – 100	1999	2001	1999	1999	15	2
> 100	1999	2001	1999	1999	10	1

Source: CEC, 1991.

The regulation on allowed maximal concentration in treatment plant drain water effectively demands agglomerations situated in sensitive areas and larger than 100,000 inhabitants to invest in tertiary – i.e. chemical - treatment.
Depending on the Member State population statistics, the Moses model assesses the required sewerage connection and mix of waste water treatment technology and the associated costs of investment and operation (TME, 1998). See Annex B on waste water treatment for details.

1.2.2.3Agricultural emissions

Based on livestock units (LU), animal density in the EU is almost 1 LU per ha. Given the requirement of the Nitrate Directive – not more than 170 kg N/ha manure should be spread – the EU as a whole seems not to have a problem. However, considering livestock density at farms, about half of the livestock is kept in densities exceeding 1.5 LU per ha. This implies that many farm holdings need to dispose of the excess manure by transporting the slurry to neighbouring farmers or to other regions.
Cattle are the dominant animal category with a share of about half of total livestock population. Pigs form 25% of the total population. The share of pigs in national livestock population is highest in Denmark (62%), Belgium and the Netherlands (both 42%). These estimates are based on well over 7 million farm holdings in the Farm Accountancy Data Network (FADN).
Several economic instruments (van Zeijts et al. 1999) have recently been discussed. Van Zeijts et al. recommend the establishment of a mineral accounts combined with a nutrient surplus levy in relevant regions. However, recent observations in the Netherlands have shown that a considerable number of farmers choose to pay the levy rather than the costs of reliable manure transport. Hence, the targets set in the Nitrate Directive are not met and the Dutch governments faces action in the Court of Justice. This suggest the levy is too low and must be based on the costs of transport (the marginal abatement costs).
Manure releases ammonia and methane, which are agents for acidification, eutrophication, and climate change. The *Technical Report on Acidification, Eutrophication and Tropospheric Ozone* discusses several technological solutions that minimise air emissions from stables and fields after manure spreading. We sought alternatives to these by setting limits to animal densities on farms and in regions.

1.2.2.4Outline of the models used

Several models have been applied to assess water management problems in the EU. The majority of the models have been used in previous policy assessments. Attention has been paid to ensure data homogeneity. Considering the subnational character of the problem, we inserted regional detail in the assessment. However, due to the variety of sources, some inconsistencies could not be avoided.

The Annexes A, B, C, D, and E specify the models and data used. The short account below lists the elements used.

Water abstraction

ETC-IW data set containing estimates of country specific data on water abstraction per economic sector (EEA, 1999).

CARMEN model accounts for the net precipitation to assess water supply and water demand (FSU, 1996).

Nutrients

LEI model calculates manure arising, application and the consequences of the introduction of livestock limits (LEI, 1999)

ETC-IW data set containing estimates of country specific data on waste water composition, sewage connection rates, sewage treatment (EEA, 1999)

RAINS model to calculate dispersion of ammonia (Alcamo et al. 1990)

CARMEN model accounts for all diffuse and point sources of nutrients to ground and marine water (RIVM, 1998).

Costs

RAINS model evaluates the costs of emission abatement, and

MOSES model to evaluate the costs of implementation of wastewater treatment.

TME model to estimate of manure transports in regions where exceedances of the norms (acidification strategy, nitrate directive) are expected.

1.2.2.5 Benefits from TD and AP

The water stress problem requires that local conditions be taken into account. The proposed Water Framework Directive, which might have come near to an AP scenario, takes this into account as it refers to local ecological targets and abatement action plans. However, models are not yet instrumented to assess the impact of the proposed framework due to lacking consistent regional data. Hence, the benefits of a TD or AP scenario have not been evaluated.

In the UK, the data availability situation did warrant a cost benefit analysis of the proposed water framework directive (WRc, 1999). Potentially, costs range from £3 to £11 billion. These costs are dominated by those of controlling municipal point sources and agricultural diffuse sources. Benefits are expected to range from £2 to £6 billion. The largest benefits arise from improved amenity for owners of riparian homes, and from improving low flow regimes in rivers.

1.2.2.6 Identification of major uncertainties

Contrary to problems like climate change, regional physical and ecological conditions define the water stress problem. However, this precise environmental data is scarce and often not comparable. Particularly, on irrigation water abstraction, wastewater composition and manure application the new EUROWATERNET must provide tailored information to assess the Action Programmes the Member States must submit to comply to the Nitrates Directive, Urban Waste Water Treatment Directive, and the proposed Water Framework Directive. Furthermore, the impact of such programmes is often ill defined as vulnerable areas requiring strongest action have not yet been formally designated.

1.2.3 Results

1.2.3.1 Measures related to water demand.

The water demand model projects national trends for each economic sector based on observed demand data series. However, the requirements of the EU Action Programme for Integrated Groundwater Protection (GAP) have not been made explicit in these data series. Where implemented, removal of subsidies on water supply to internalise water supply costs in the water price might already have contributed to meet these requirements. The water demand projections should reflect the impacts implicitly.

Considering the present and future water demand per sector, the total water demand in the EU will remain relatively stable. The rate of growth of the main driving forces is expected to slow and the efficiency of water use is expected to improve, as national water conservation policies and actions have an increasingly positive impact.

However, the semi-arid areas of the EU - Portugal, Spain, Italy and Greece – remain very susceptible to the effects of desertification due to the imbalance between natural resources and abstractions for irrigation. The trend in irrigated area and the allocated amount of water per hectare will slightly decrease. Hence, the demand for irrigation water is decreasing.

Table 1.2 Water demand for irrigation relative to the amount of available net precipitation and to the amount of annual resources (net precipitation and import).

	precipitation	evaporation	Net precip	Allocated to irrigation	irrigation to net precip	Irrigation to Resources
	mm	mm	mm	mm	%	%
Greece	849	492	357	312	20	16
Italy	983	428	555	1188	28	27
Portugal	886	474	412	607	18	9
Spain	662	432	230	546	25	25

Source: EEA, 1999.

1.2.3.2Measures related to waste water treatment.

Full compliance to the urban waste water treatment directive - as assumed in the BaseLine - by 2010 will remove most of the problems to surface water pollution by waste water. The policy target is to be met in the BaseLine, and no additional targets have been set in the other scenarios. As no sufficient information was available on the area designated as vulnerable by the Member States, the assessment assumes that the whole EU-15 territory is vulnerable to eutrophication and the severest requirements of the Urban WasteWater Treatment Directive apply.

In Table 1.3 the COD, Ntot and Ptot emissions before and after treatment are presented for the EU-15 in the 1995 and 2010.

Table 1.3 Implications of the implementation of the Urban Waste water Directive (common in teh BL, TD and AP scenario) and removal efficiency rates (%) in the EU-15.

	1995			2010		
	Before treatment	After treatment	Efficiency %	Before treatment	After treatment	Efficiency %
COD	23.22	10.45	54	24.17	6.31	73
Ntot	2.02	1.25	37	2.10	0.92	59
Ptot	0.36	0.22	35	0.37	0.14	65
Annual costs		41			48	

Based on Table 1.3, the following can be stated about the COD-, Ntot- and Ptot-emissions before and after treatment in the EU-15 in 1995 and 2010:

- Untreated emissions will increase in the period 1995-2010. This increase is linearly related to the growth of the population in the EU-15;
- Treated emissions in 2010 are much lower than those in 1995. The increase of sewerage connection (prerequisite for treatment/emission reduction) and stronger reduction requirements (see Table 1.3) will cause this improvement.

Despite the increase of untreated emissions with 4%, implementation of the UWWTD will reduce the actual emissions in 2010 with 40%, 26% and 37% for COD, Ntot and Ptot respectively compared to the actual emissions in 1995.

Figure 1.1 presents the change in nitrate concentration at the river mouth. Agriculture is the main emitter of nitrates. Except for the hot spots of animal husbandry, nutrient emissions hardly change. *Figure 1* also presents

the change in phosphate concentration at the river mouth. Phosphate concentration decreases mainly due to the use of detergents without phosphates in some EU Member States.
Despite the population and welfare growth – both factors amplifying emissions - the nutrient concentrations are shown to decrease in all European rivers.

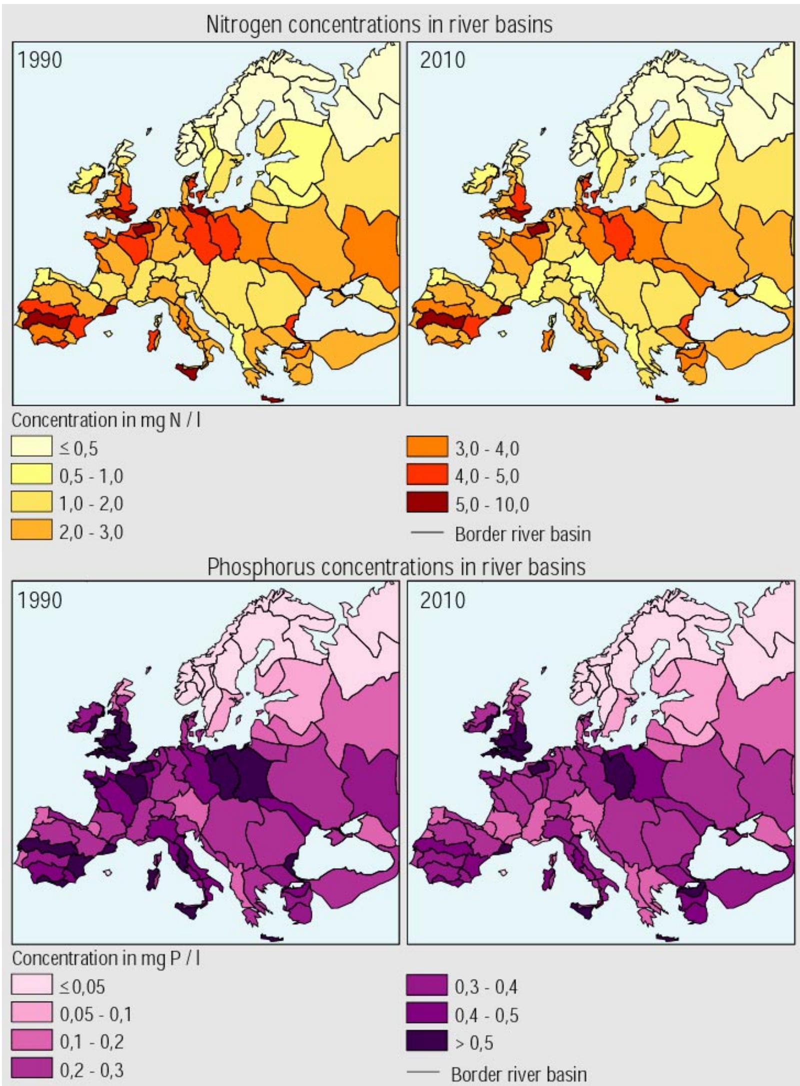

Figure 1.1 Nitrogen and phosphorus concentrations in river basins by 1990 and 2010, as projected for the river mouth for the Baseline scenario. In the EU, the agricultural sector is the main supplier of nitrates. The introduction of new household detergents has/ will have a marked effect on the phosphate load. For the Central and Eastern European countries, the Baseline scenario projects show convergence to EU standards, implying an enhancement of sewerage system connections and sewage treatment (see section on enlargement, scenario A).

1.2.3.3 Measures related to agriculture.

Transporting manure out of 'hot spots'
Current legislation with regard to manure disposal limits the application of manure on agricultural land to 170 kgN/ha. When at a farm, the amount of manure exceeds this limit, the excess manure has to be transported or treated. Up to now treatment costs are much larger than transport. Therefore, we only considered transport of manure collected at intensive husbandry farms holding pigs, poultry and dairy cattle. Figure 1.2 shows the transport costs in the EU-15 for various limits of the manure disposal limit given the 1990 split of regional animal densities (Eurostat, 1995) as evaluated by TME.

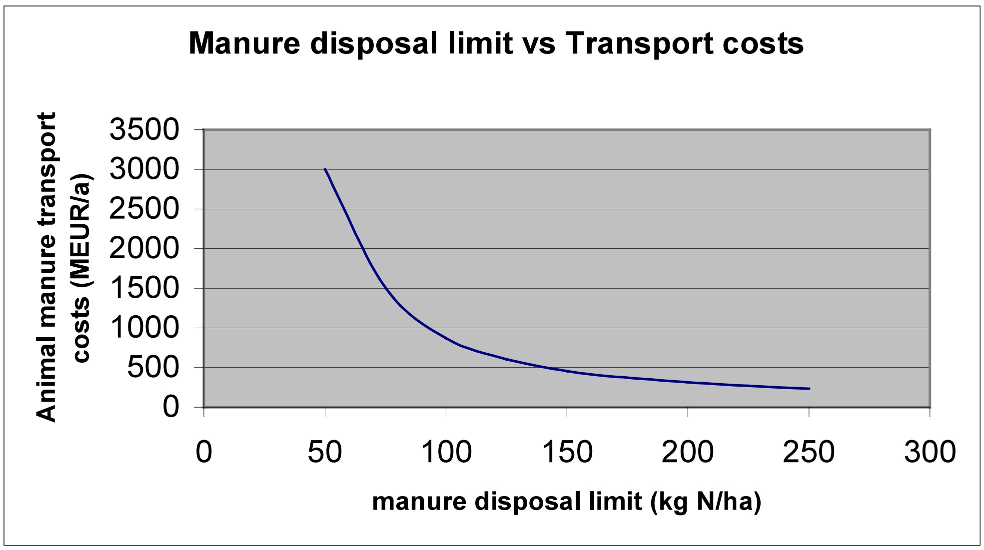

Figure 1.2 Animal manure transport costs (million € per year) as a function of introducing manure disposal limits.

Reducing animal density
Relaxing the measure to 1 LU/ha nationally, or 1.5 LU/ha on farm level or regionally would relax the pressure on the agricultural sector. Table 1.4 shows the impact of the four alternatives and the impact on the manure arising. Aggregating the impacts of these measures to EU-15 level hides the impacts on regional agriculture and environment. Appendix C provides details for the 'hot spots' of intensive animal husbandry in the EU: Belgium, Denmark, Niedersachsen, Nordrhein-Westfalen, Galicia, Cataluna, Loire, Bretagne, Lombardy, and the Netherlands, all regions with a livestock density higher than 1.4. Over 50% of the pigs husbandry is concentrated in these 'hot spots', while their arable area makes only 13% of the total arable area.

Table 1.4 Impact of various volume measures regarding animal densities in Livestock Units (LU) in the EU-15 compared to the 1995 situation.

	Cattle million	*Pigs million*	*Poultry million*	*density LU/ha*	*Manure kg N/ha*
1.0 LU/ha farm	62	5	40	0.41	39
1.5 LU/ha farm	73	10	91	0.49	46
1.0 LU/ha region	80	21	182	0.56	50
1.5 LU/ha region	84	42	364	0.64	56
1995 situation	88	115	1010	0.90	61

source LEI, 1999, adapted by RIVM

We made the following full assessments integrating the issue of acidification and eutrophication and the issues of water management and coastal zones for the agricultural sector in relation to the control of ammonia emissions:

	scenario	*technology costs*	*volume impact*	*Environment impact*
BL	no policies in place	small	small increase	Small
TD	‘end of pipe’	high	small increase	Medium
AP-techno	‘end of pipe’	medium	small increase	Small
AP-volume	1 LU/ha on farm	small	large reduction	Large

The AP-techno scenario assesses cost-effective solutions to both acidification and ozone exposure and does not limit emissions as such. The scenario assesses a package of measures for SO_x, NO_x, NMVOC and NH_3 emission abatement. The impact of the volume measure (in AP-volume) is quite extreme: pigs and poultry populations reduce by a factor of 10. However, in acidification abatement this drastic scenario - with its NH_3 emission reduction - will allow for larger emissions of SO_x and NO_x, as both packages, techno and volume, meet the same emission target. By and large the AP volume will bring about a transfer of costs from the transport, industry and energy sectors to the agriculture sector. Also, there are more spillover effects. The AP-volume has fewer emissions of methane (climate change), and particulate matter PM_{10} (chemical risks, urban stress).

Table 1.5 reviews the obtained results aggregated to EU-15 level. Emission reduction to 60% of 1990 levels are obtained by both this AP and the TD scenario at the expense of reducing the pigs and poultry sector by a factor of 10 or of increasing control costs of €11 billion per year.

Table 1.5 Comparison of the various scenarios.

EU-15 (figures from RAINS)	*1990*	*2010-BL*	*2010-AP-techno*	*2010-AP-volume*	*2010-TD*
Cows (millions)	91	84	84	73	84
Pigs (millions)	117	118	118	15	118
Poultry (millions)	929	1004	1004	100	1004
Total CH_4 emission	9652	8985	8985	7682	8985
Total N production	8973			7738	
Total PO_4 production					
NH_3 emission (kton)	3576	3153	2969	2206	2156
NH_3 abatement costs (M€/yr)		317	1405	49	11471
Total abatement costs (G€/yr)		53	59	61	87
Area exceed acid.	25%	5%	3%	3%	2%
Area exceed eutrop.	55%	41%	36%	28%	24%

1.3 Conclusions

KEY MESSAGES

- water quality protection represents the key attribute, although water supply is an important issue in Mediterranean countries.
- water use will probably remain relatively stable, if not decline, for some sectors through more efficient management and pricing during the outlook period.
- Wastewater from point sources will most likely be effectively treated by the end of the assessment period. This will significantly reduce nutrient impacts on downstream rivers and marine areas.
- the most important aspect to be addressed by new policy measures relates to agricultural runoff.

Current EU directives have addressed adequate water quality management. This applies particularly to the Nitrates Directive and the Urban WasteWater Treatment Directive, which have been entered in the Baseline assessment. Implementation of these directives is on its way, though large implementation failures have been reported. Implementation costs are:

The water stress problem requires that local conditions will be taken into account. The proposed Water Framework Directive, that should encompass all previous directives into one binding legal framework, takes this into account as it refers to local ecological targets and abatement action plans. However, models are not yet instrumented to assess the impact of this proposal due to lacking consistent regional data. Hence, no TD or AP policy scenarios have been evaluated. The introduction of a monitoring system – EUROWATERNET - should alleviate this data problem in the near future (EEA, 1998). It is recommended to ensure that an associated assessment model will be developed.

In the UK, the data availability situation did warrant a cost benefit analysis of the proposed Water Framework Directive (WRc, 1999). Potentially, costs range from £3 to £11 billion, which are dominated by the costs of controlling municipal point sources and agricultural diffuse sources. Benefits are expected to range from £2 to £6 billion. The largest benefits arise from improved amenity for owners of riparian homes, and from improving low flow regimes in rivers. However, many uncertainties remain with regards to assumptions and sensitivity analyses stressed the limitation of the above-mentioned results.

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2. Benefit assessment

2.1 Water stress

2.1.1 Public opinion

Eurobarometer 1995 and 1992 both rank 'water management' as one of the top three most serious environmental problem. Variance about this ranking is quite narrow: there is a 1st ranking in Ireland and Denmark and a 2nd ranking in England and the eight[2] European countries covered in the ISSP (1993) survey.

2.1.2 Expert opinion

GEP et al. (1997) report rankings for 'water scarcity and pollution'. The pollution is due to agricultural practices, air pollution (acidification) and industrial waste. Eutrophication is not given a separate category. Water management is ranked third, with 21.6% of researchers citing it as a first or second problem. As is expected, water scarcity and pollution is ranked first by South European countries. Most probably this reflects concern about water availability rather than pollution. It is ranked third by North European countries.

2.1.3 Benefit estimation

The benefit estimate results for the Baseline scenario only are summarised in Table 2.1.

Table 2.1 Summary of monetised environmental damage due to Baseline

Environmental damage due to Baseline scenario € million		
	1990	2010
Availability: not calculated[3]		
Quality-1: areas where 50 mg/l NO_3-N target in groundwater is exceeded	602 - 5,716	397 - 4,594
Qualtiy-2: river stretches of a certain quality		
Average EU quality	152	146
Poor quality	245	212
Total	397	358

Suitable indicators and valuation estimates for further research in this area are as follows:

Availability:
- population at risk of discontinuous supply;
- water consumption data, and
- WTP data for water in Europe.

Quality:
- volume of water used per unit of value added in agriculture, and
- Nitrate and pesticide concentration in ground water.

2 ISSP (1993) survey, includes: The Netherlands, Ireland, Italy, Great Britain, Norway, Spain, West Germany, East Germany.

3 not calculated due to incompatibility of key indicator data with WTP data.

Water management indicators

Water management addresses two issues: water availability and water quality. As far as cost-benefit analysis is concerned the following are the relevant indicators:

Water availability: is defined with respect to ground water availability and water stress. Where water stress is defined as a situation in which there is less than 200 litres/cap/day availability. Two variants of this are given by RIVM: the first indicator (ground water availability) assumes 1/4 of groundwater recharge is available for consumption. The second indicator (water stress) assumes that 1/4 groundwater recharge is available and 1/10 of total upstream discharge is available. However, these data are available for the Baseline scenario only.

Water quality: the indicators are; i) areas where 50 mg/l NO_3-N target in groundwater is exceeded and ii) river stretches of a certain quality.

There are studies of the WTP to avoid nitrates in groundwater, however the indicator is not currently available at the EU:15 level. N and P concentrations in coastal areas are covered in the eutrophication analysis of the Baltic Sea dealt with in the benefit assessment of 'Coastal Zones'.

Water availability analysis

Table 2.2 and 2.3 report the water availability (or water stress) indicators.

Table 2.2 Groundwater availability and stress

Country	fraction of stressed population (1990)	fraction of stressed population (2010 - baseline)
Austria	0.14	0.16
Belgium + Luxembourg	0.38	0.46
Denmark	0.31	0.31
Finland	0.33	0.33
France	0.29	0.31
Germany	0.37	0.37
Greece	0.52	0.52
Ireland	0.14	0.14
Italy	0.45	0.43
The Netherlands	0.20	0.30
Portugal	0.49	0.49
Spain	0.45	0.45
Sweden	0.18	0.24
United Kingdom	0.58	0.58
EU - 15	0.41	0.41

Table 2.3 Surface and groundwater stress

Country	fraction of stressed population (1990)	fraction of stressed population (2010 - baseline)
Austria	0.00	0.00
Belgium + Luxembourg	0.26	0.26
Denmark	0.26	0.26
Finland	0.00	0.00
France	0.03	0.04
Germany	0.07	0.07
Greece	0.41	0.41
Ireland	0.00	0.00
Italy	0.14	0.14
The Netherlands	0.20	0.20
Portugal	0.34	0.34
Spain	0.18	0.19
Sweden	0.00	0.05
United Kingdom	0.22	0.22
EU - 15	0.13	0.14

The fraction of the population with water stress stays constant for EU-15 in the Baseline scenario for one indicator and increases slightly for the second indicator. However, since the population grows from 365 million in 1990 to 389 million in 2010, the result is an increase in the absolute number of people subject to water stress. The estimates are:

	1990	2010 Baseline
Number of people subject to water stress	millions	millions
Groundwater availability	150	159
Water (surface and ground water) stress	47	54

The indicator showing lower stress relates to surface and groundwater together whereas the higher stress indicator relates to groundwater only. The low stress indicator would seem preferable since water supplies come from groundwater and surface water in most countries, although, clearly, the situation varies by region.

The data indicates that most 'stress' occurs in Portugal, Spain, Italy, Greece and the UK. The UK is the most stressed country if only groundwater is considered, but Greece is the most stressed if we consider surface and ground water together.

2.1.4 Water availability analysis

The correct approach to estimating the economic value of water stress is to applying individual's willingness to pay to secure an improved water supply. While there are numerous studies of the demand for water in developing countries, clearly because of problems of water scarcity, there is a surprising dearth of studies for Europe. Table 2.4 provides a survey of *residential* demand in developed countries, utilising the excellent study for Sweden by Hoglund (1997), the guideline document by Young (1996) and the earlier synthesis of Gibbons (1986).

Table 2.4 Studies of water demand in developed countries

Country and study	Mean quantity litres per day per person	Mean price in 1980 US$/m^3	Elasticity of demand[1]	
			marginal price	*average price*
SWEDEN				
Hanke and deMare, 1982	137	0.88	-0.15	
Hoglund, 1997	190	0.45 to 0.61	-0.11	-0.20 to -0.41
FINLAND				
Laukkanen, 1981	na	na	-0.11	
USA				
Danielson, 1979	254	0.4 to 0.8	-0.27	
Foster and Beattie, 1979	197	0.36		-0.12
Carver and Boland, 1980	170 to 821	0.19 to 1.04		-0.70
Cochrane and Cotton, 1985	469	0.31		-0.40
Griffin and Chang, 1985	627	0.31 to 0.43		-0.19 to -0.37
Nieswiadomy and Molina, 1989	Na	0.27	-0.36 to -0.86	
Schnedier and Whitlach, 1991	Na	na	-0.26	
Nieswiadomy, 1992	340 to 660	0.20 to 0.35	-0.11 to -0.17	-0.22 to -0.60
Martin and Wilder, 1992	Na	0.35 to 0.60	-0.32 to -0.60	-0.49 to -0.70
Nieswiadomy and Cobb, 1992	443 to 522	0.22 to 0.28	-0.17 to -0.29	-0.46 to -0.64

Source: Hoglund (1997), and Young (1996). For a more extensive compendium of US price elasticities see Pezzey and Mills (1998).

Note. 1. Water tends to be priced with a fixed charge and a variable charge, the latter varying with consumption. As such, an *average price* is the fixed charge plus the variable charge divided by quantity. The *marginal price* is the extra charge for consuming an extra unit of water. Analysts dispute which is the most relevant price. The average price is widely used because there is some evidence that consumers respond more to changes in the average price than to changes in the marginal price.

A cost-benefit model for increased water supply

To compute the costs and benefits benefit of reducing water stress requires a model along the following lines:

Step 1: compute the extra quantity of water required to reduce water stress, according to the chosen scenario;

Step 2: find the willingness to pay of an increment in water quantity;

Step 3: multiply the WTP per unit water availability by the quantities required to obtain the *aggregated gross willingness to pay* for the increment;

Step 4: estimate the cost of providing the extra quantities of water;

Step 5: deduct the cost of provision from the gross willingness to pay to obtain *aggregated consumers surplus*, which is then the measure of net benefit of the provision policy.

Such a model is problematic at an EU-wide level. First, data on actual consumption of water are limited. Some of the EUROSTAT data, for example, relates to consumption in 1980 as the last date of information. Second, as Table 2.4 shows, there are very few data on the willingness to pay for water in Europe. Third, while it is possible to estimate the cost of new investments for water supply, the pricing of water does not obey the requirements of optimal private cost pricing in all countries.

There are hidden subsidies and substantial regulation which prevents water being priced at marginal supply costs everywhere. This means that estimating the costs of new supply is not always relevant: policy measures to price water correctly, even at private cost levels, will often be more appropriate. But estimating the costs of such policies requires analysis and information beyond the scope available for this study. Fourth, 'water stress' is itself a 'fuzzy' concept. Water is used in various ways: for domestic consumption, for agriculture, industry, and for environmental purposes. The concept of 'stress' implies that any one of these sectors can be, in some sense, 'short' of water. But shortages in one sector could be relieved by reducing supplies in another sector. Economic analysis would suggest that water should be allocated to the sector with the highest WTP so that the final allocation of water supplies meets the condition that marginal WTP is equal in all uses. But this is not how water is allocated, so that there are considerable allocation distortions. In the presence of those distortions, 'water stress' becomes a concept of doubtful validity.

Data on water consumption

Data on water consumption are unreliable. EUROSTAT (1995, Table 5.3) reports consumption by sector for most EU European countries, but it is not possible to make the data there compatible with the RIVM 'water stress' indicators[4]. Thackray (1996) has produced the figures for *residential* consumption shown in Table 7.5. Table 2.5 also shows some price information taken from sources utilised in Pezzey and Mill (1998).

Table 2.5 Water consumptions and prices in selected countries

Country	Consumption in cubic metres/yr per capita, 1986*	Price in € per m^3
Belgium	38	88
Austria	47	69
England and Wales	49	na
Germany	53	100
France	54	73
Denmark	70	na
Sweden	73	na
Italy	80	na
Switzerland	95	44

Source: Thackray (1996), Pezzey and Mill (1998). *Thackray produces 1995 data which show some changes in the numbers shown here. However, the price data are for 1986 so we have used 1986 consumption data.

Willingness to pay estimates for increased water supply

To find the WTP for an increment in water supply use can be made of the formula (Gibbons, 1986, p17):

$$GWTP = (P.Q_2^x)/(1-x).[Q_2/Q_2^x - Q_1/Q_1^x]$$

Where, GWTP is gross willingness to pay, P is the ruling price, Q_1 is the quantity consumed, Q_2 is the new quantity, and x = 1/e where e is the elasticity of demand (independent of sign).

Application of the formula to the data in Table 2.5 gives the following results (see Table 2.6) assuming an elasticity of -0.2 (Hoglund, 1997).

4 For example, France has 31.3 billion m^3 year consumption according to EUROSTAT. But this figure relates to 1981. Even if it is approximately correct, average consumption would be $31.3x10^9$ x 2.738 x $[56.74 \times 10^6]^{-1}$ = 1510 litres per day, more than 7 times the 'water stress' threshold. If only domestic consumption is taken, then the figure is 90 litres per day, less than 50% of the water stress threshold. The conversion factor 2.738 converts cubic metres per year to litres per day.

Table 2.6 Willingness to pay for water in selected EU countries

Country	GWTP per capita for 1 m^3 increase in supply of water 1986 base €	GWTP per capita for10% increase in supply of water 1986 base €
France	0.51	3.36
Germany	1.05	6.83
Belgium	0.94	4.37
Austria	0.73	5.08

The results are reasonably uniform for the countries where estimates are possible, suggesting that a 10% increase in supply would be valued at € 3.4 - 6.8 per person. Taking average consumption to be 70 m^3 year in 1997, this means that an extra m^3 would be valued at € 0.5 - 1 per m^3 in 1986 prices, or, assuming WTP rises proportionally with income, some € *0.7 - 1.4 per m^3 in 1997 prices*.

Willingness to pay here has been derived from actual market data. A different approach is to conduct a stated preference survey whereby customers are asked to indicate their WTP for an improvement in service. The UK water regulatory agency conducted such a survey, though unfortunately the sample chosen was not random (Bolt, 1993). Out of the three kinds of service improvement, one related to reductions in the risk of supply interruptions. Households were WTP £28.8-32.7 per annum for this benefit. In 1997 price terms this is £32.4 to £36.8, or € 48.6 - 55 per household per year. Averaged over average consumption in England and Wales of about 54 m^3/ year, this is € 1 per m^3 which is surprisingly consistent with the market demand estimates.

Costs of water supply

As noted above, comparing GWTP measures with new supply costs is somewhat misleading in policy terms because new supplies may not be the most efficient means of meeting a demand increase. Other policies, such as leakage reduction and full cost pricing may be preferred. But it is possible to compare the WTP numbers derived above with costs of new supply. Table 2.7 presents some typical cost data for new supply options. Self evidently, costs of supply vary enormously with site and type of scheme, but the figures shown in Table 7.2 are broadly representative of new schemes in Europe.

Table 2.7 Costs of water supply

Type of new supply scheme	Unit costs € per liter per day
Groundwater schemes	0.46
Re-use schemes	1.04
Transfer schemes	1.34 to 2.44
Reservoir	2.15
Barrage	2.28
Desalination	4.55
National grid	5.46

Source: Halcrow and Partners (nd). UK£ estimates multiplied by 1.3 to give €.

Cost benefit analysis of water supply

The data suggest that, if WTP for increased water supply in the EU follows the higher levels estimated, € 1.4 per m^3, then only some new supply schemes would be justified in cost-benefit terms: i.e. groundwater schemes and re-use schemes and perhaps some transfer schemes. New reservoirs, desalination and nation-wide water grid schemes would not be justified. However, if WTP is at the lower end of the range, i.e. € 0.7 per m^3, then only groundwater schemes would make economic sense. This is a useful conclusion since it suggests that water stress problems are not best resolved by new schemes, a conclusion that is in fact in keeping with much of the 'new philosophy' about water supply in Europe, a philosophy that focuses more on leakage reduction and demand management first and new supplies only last. Once again, it is important to understand that these are very broad averages: it does not mean that a new reservoir in some location is never justified.

Conclusion on water availability

There is a clear WTP on the part of EU residents for increased water supply but comparison of this WTP with the costs of new supplies suggests fairly strongly that new schemes are not generally justified in cost-benefit terms. The focus of policy should be on correct water pricing and demand management schemes (see the policy discussion).

2.1.5 Water quality analysis

The second feature of water management is water quality. Two quality indicators are available: (i) areas where 50 mg/l NO_3-N target in groundwater is exceeded; and (ii) river stretches of a certain quality.

Water quality 1: areas where 50 mg/l NO_3-N target in groundwater is exceeded: Table 2.8 shows the fraction of area above this groundwater quality guideline.

Table 2.8 Groundwater quality

Country	Fraction of area above guideline (1990)	fraction of area above guideline (2010 - baseline)
Austria	0.02	0.02
Belgium + Luxembourg	0.13	0.13
Denmark	0.07	0.07
Finland	0.00	0.00
France	0.18	0.15
Germany	0.06	0.04
Greece	0.02	0.01
Ireland	0.21	0.22
Italy	0.00	0.00
The Netherlands	0.19	0.08
Portugal	0.00	0.00
Spain	0.00	0.00
Sweden	0.00	0.00
United Kingdom	0.09	0.07
EU – 15	0.05	0.04

Hanley (1989) estimates the WTP for groundwater quality improvement, using a contingent valuation survey based on the population in the Anglian Water Services area. This value is € 20 per person per year (1997 prices) for ensuring that water quality does not go below the limit of 50mg/l nitrate concentration.

The RIVM scenario data are presented in terms of fraction of area above the guideline value of 50 mg/l, while the only valuation study for nitrate pollution is expressed in units of WTP per person per year. Therefore, a conversion of the units of the physical data is necessary.
For this, the following assumptions are made:

- for the lower bound estimate we assume that only those who live in the area with groundwater above the guideline would be concerned about this problem and will have a WTP to avoid it. Under this assumption the correct way of finding this population would be to take the population figures in the grid squares which are affected by the poor groundwater quality. However, for now, we choose a short cut and assume that the fraction of area affected in a country is identical to the fraction of population affected in that country. This implies a uniform population distribution, which is not strictly correct, but is best that can be assumed under the circumstances.
- for the upper bound estimate we assume that the whole population in a country is concerned about this problem and hence will have a WTP to avoid it regardless of the fraction of the total area affected or the level of pollution. Therefore, we aggregate the individual WTP estimate over the whole population.

Table 2.9 shows the damage valuations for the baseline scenario in 1990 and 2010.

Table 2.9 Damage costs in 1990, 2010 in the Baseline: € million (1997 prices)

Estimate	**1990**	**2010**
lower bound	602	397
upper bound	5,716	4,594

Note; Baseline scenario relates relates to areas where 50mg / l NO_3-N target in groundwater is exceeded.

Water quality 2: river stretches of a certain quality: There are two sets of physical data for this indicator. The first one shows the fraction of surface area with average (EU15) conditions (1.7mg/l). The second one shows the fraction of surface area with poor quality river water. Table 2.10 presents both sets of data.

The valuation study we have chosen to use for this is Green and Willis (1996). They estimate two non-use value estimates, i.e. WTP unrelated to actual use of the rivers.

- € 0.0025 per household per km of freshwater body per year for improving water quality from 'medium' to ' good' quality. This value is used for the data based on the average EU quality, and
- € 0.0047 per household per km of freshwater body per year for improving water quality from 'poor' to 'medium' quality. This value is used for the data for poor quality river water.

In both cases we assume that an area of x km^2 affected by each pollution level is representative of x km (i.e. the square root of x km^2) of water body affected by the same pollution. The number of households concerned with this problem and which have a WTP to avoid it in any given country is the total number of households in that country. This is because the WTP estimate is based on non-use value which can, in principle, be applied to the whole population. In practice, there is evidence that WTP declines with distance from the site, so the measures here are overestimates.

Table 2.10 River stretches of a certain quality

Country	EU average quality		poor quality	
	1990	2010 - baseline	1990	2010-baseline
Austria	0.06	0.06	0.00	0.00
Belgium + Lux.	0.76	0.76	0.61	0.65
Denmark	0.52	0.3	0.11	0.05
Finland	0.00	0.00	0.00	0.00
France	0.41	0.38	0.16	0.12
Germany	0.67	0.51	0.15	0.04
Greece	0.39	0.36	0.18	0.17
Ireland	0.08	0.09	0.03	0.03
Italy	0.45	0.41	0.24	0.23
The Netherlands	0.61	0.56	0.15	0.15
Portugal	0.34	0.3	0.12	0.11
Spain	0.59	0.56	0.26	0.24
Sweden	0.01	0.02	0.00	0.00
UK	0.43	0.38	0.27	0.21
EU – 15	0.37	0.33	0.15	0.12

Table 2.11 shows the damage values for (a) fraction of surface area with average (EU15) conditions and (b) fraction of surface area with poor quality river water.

Table 2.11 Damage cost in 1990, 2010 due to water quality, Baseline scenario: € million

Estimate	1990	2010
Average EU quality	152	146
'poor quality'	245	212
Total	397	358

The damage associated with 'poor' quality is greater than that associated with average EU quality. This is because the unit WTP to improve the former is greater than that for the latter.

Cost-Benefit analysis of ground water quality

Significant work has now been done on the economic value of water as an environmental service. Table 2.12 reports on the value of water in three UK studies. In each case a contingent valuation methodology was used. Significant issues arising in the studies were: (a) the statistical non-significance of the non-use values in the S W England study, i.e. the mean value reported is not statistically significant; (b) the major problem of aggregating individual or household values, i.e. over what population are the values representative?; (c) the significant proportion of zero WTP responses in the Ouse case, the mean values being 'driven' by the one third of the general public giving positive WTP numbers. The studies do show, however, that economic values for water quantity (and, thereby, water quality since flow regimes affect quality) can be derived. How meaningful it is to convert these values into 'per m^3' or 'per miles' numbers is far more doubtful, even though several 'manuals' of assessment have attempted to make this transition (Foundation for Water Research, 1996).

Table 2.12 Economic value of environmental functions of water, UK

Study	Location	Value of water: WTP per person to:		
Garrod and Willis 1996	River Darent, England		Maintain flow	Improve flow
		Visitors Residents Non-users	£7.2 £10.2 £3.9	£4.9 £6.3 £3.0
ERM and University of Newcastle, 1997	Rivers in S W England	Value of water: Marginal WTP per person per mile per annum		
		Users Non-users	£0.0760 £0.0435	
EFTEC and CSERGE, 1999	River Ouse, England	Value of water: WTP per household per annum to restore river to pre-abstraction levels (5cm increase)		
		General public Users	£4.7 £5.6	

Groundwater contamination can arise from various chemical risks, including pesticide risks and from nitrate contamination due to fertiliser run-off. There are only a few benefit valuations for Europe but more from the USA (see Table 2.13). These are fairly consistent in suggesting that WTP to avoid contamination could be very high, reflecting households' concern over drinking water. The two European studies, Hanley (1989) for England and Press (1998) for Italy, suggest a range of valuations of one order of magnitude, but the Italian estimate could reflect the existence of a prior major incident with the quality of Milan's groundwater.

Table 2.13 Studies of WTP per household per year to avoid water contamination

Study	WTP per household per year € 1999
USA:	
Jordan and Edwards (1993)	832 - 1118
Schultz and Lindsay (1990)	356
Edwards (1988)	610 - 3042
Power (1991)	68
Mitchell and Carson (1986)	4 - 63
UK:	
Hanley (1989)	49
Italy:	
Press (1998)	490
Costs of control	31
B / C ratio (low ratio)	1.5: 1
B / C ratio (high ratio)	15: 1

Costs of decontamination are quoted by Söderqvist as some $36 per household per annum (€ 31 ph pa) for a clean-up programme in Venice. Comparing benefits and costs and assuming values are transferable between sites, suggests that benefit - cost ratios could be 1.5:1 even for the low benefit estimates. At the high WTP value, benefit - cost ratios could be 15:1.

2.2 Coastal zones

2.2.1 Public opinion

Eurostat (1992 and 1995) both rank the degradation of coastal zones first, thus making it the most important environmental problem in Europe. Coastal zone pollution is also ranked first in the UK national study (DoE 1993) and third in Denmark (1995). Overall, taking all the surveys into consideration, the final ranking again suggests that coastal zones degradation is top priority.

2.2.2 Expert opinion

GEP et al (1997) have a category 'seas and coastal areas' which would seem to directly reflect this environmental issue. Sea and coastal pollution is the main issue, together with the degradation of coastal scenery and the overexploitation of marine resources. It is ranked last out of 11 categories with a world score of only 5.7%. Surprisingly, South Europeans gave the same score as the North with the ranking and with the score, 5% (0.05). One would have expected a higher score based on the direct concern for their coastal areas.

2.2.3 Benefit estimation

A methodology to value coastal zones is given, however final valuations are omitted due to the lack of suitable indicators for coastal zone management and the consequent lack of scenarios and targets. Table 2.2.1 suggests the type of indicators needed and valuation estimates required for future research into the valuation of coastal zone degradation.

Table 2.2.1 Indicators and valuation estimates needed for future research

Bathing Water quality: % municipalities complying with Bathing Water Directive
Clean Beaches: Blue Flag indicators
Area of marine areas protected
'Hot spots' coastal zones

An obvious objective would be full compliance with the Bathing Water Quality Directive. EFTEC estimates the monetised environmental benefits (in 2010) of the EC Directive on Quality of Bathing Water. The analysis assumes a 'maximum feasible' scenario that all coastal zones meet the quality standards in 2010, whilst in the Baseline scenario 4% remain below standard[5]. The environmental benefits are estimated to be € 84.1 million Full details are given below.

Definition of coastal zones

With no unique definition of what constitutes a 'coastal zone' it is important to establish a workable definition in order to set parameters for the area of concern. A workable definition is as follows:

> 'the part of the land affected by its proximity to the sea; and that part of the sea affected by its proximity to the land as the extent to which man's land-based activities have a measurable influence on water chemistry and marine ecology'

(US Commission on Marine Science, Engineering and Resources, 1969)

The definition suggests an intuitive division of coastal zones into three categories.

Coastal zone type 1: the sea area in close proximity to land that is affected by land based activity, for example, pollution, eutrophication;

Coastal zone type 2: the coastline itself including beaches and cliffs, and

Coastal zone type 3: coastal / saline wetlands that lie behind the coastline.

Note that wetlands, both freshwater and saline are included in biodiversity loss. To avoid double counting, coastal wetlands are omitted from this analysis.

Total economic value of coastal zones

In economic terms natural assets in the coastal zone can yield both use value and non-use value. Use value includes direct use value, indirect use value and option value. The non-use value includes existence value and bequest value. The total economic value (TEV) of coastal zones should include all of these elements.

$$TEV = DUV + IUV + OV + BV + XV$$

where:

DUV = direct use value e.g. fishing, recreation
IUV = indirect use value e.g. flood control, storm protection
OV = option value e.g. insurance value of preserving options for future use
BV = bequest value e.g. value of passing on natural assets 'intact' to future generations

[5] Scenario data is not given for this environmental issue, the MFR and Baseline scenarios used here are authors own suggestions.

XV = existence value e.g. value derived from just knowing a species/system is conserved

In terms of placing monetary values on coastal zones and analysing the benefits of any given scenario it is convenient to follow the approach of dividing coastal zones into the three sections detailed above.

An additional threat to coastal zones is the potential sea level rise resulting from global warming. Valuations for coastal damage due to sea level rise are included in 'climate change', thus they are omitted from this analysis.

The contingent valuation method (CVM) can provide valuations of both use and non-use value depending on the design and location of the questionnaire. For this reason we concentrate on CVM valuation studies to derive valuation for coastal zone damage.

A specific problem concerning coastal zone valuation is the process of aggregation from individual WTP measures to Europe wide valuations of coastal zones. For example, a number of studies provide WTP values for bathing water quality. However, all studies concentrate on direct use value by questioning local residents and holiday-makers at coastal resorts. To aggregate such valuations would require information on the proportion of the European population living in coastal areas and the number of people taking holidays in European coastal resorts. Even if such data is available the valuations will still ignore non-use values. Issues concerning aggregation will be highlighted. We proceed by considering the coastal zone type 1, sea area in close proximity to land.

Coastal zone type 1: sea area in close proximity to land

To value the damage caused by pollution of coastal water this study concentrates on valuations connected with bathing water quality. As a result of the public health risks from sewage contaminated coastal waters the EC introduced a Directive on the Quality of Bathing Waters (CEC, 1976). The Directive specified numerical quality standards for a range of physico-chemical, bacteriological and aesthetic criteria. Table 2.2.2 shows the percentage of sites that did not meet these standards in 1993 for each EU12.

Table 2.2.2 Percentage of sites not meeting Directive on Bathing Water standards - 1993

	No of sites monitored	% not meeting standard
Belgium	39	3
Denmark	315	0
Germany	399	11
Greece	1240	2
Spain	1399	3
France	1690	4
Ireland	90	1
Italy	4017	4
Luxembourg	n/a	n/a
The Netherlands	39	0
Portugal	208	8
UK	457	9
EU12	9893	4

Source : Eurostat

A study by Georgiou et al (1996) investigates the WTP for EC standard bathing water quality in the UK. The study was conducted at two sites - Lowesoft and Great Yarmouth. Lowesoft had already achieved the Directive standard and the WTP relates to maintaining the standard. Great Yarmouth had not reached the Directive standard and the WTP relates to improving bathing water quality to achieve the standard. The WTP values are broken down into holiday makers, day trippers and local residents. For the purpose of this study the mean WTP for local residents and day-trippers is used as it conforms with the method of aggregation. The results are converted to Euro 1997 using the Euro inflation rate of 6.9% between 1995 and 1997. It is also assumed that environmental quality has a rising relative price through time that is linked to growth in EU per capita GNP. The per annum adjustment factor is calculated at 1.005. Table 2.2.3 shows the results from the Georgiou et al study.

Table 2.2.3 WTP for achieving / maintaining EC Directive bathing water quality in the UK, (1993).

	WTP /pp/pa € 1997	Sample size
Lowesoft (maintain)	18.08	203
Great Yarmouth (improve)	12.80	197

A study by Le Goffe (1995) investigated the WTP for 'without risk' bathing water quality in the natural harbour at Brest, France. The WTP value is € 35.74 (1997 price) per household per annum. When converted to a per person value the figure is some € 14 per person, very much in line with the Georgiou et al results

A separate study for Portugal (Machado and Mourato, 1998) uses both contingent valuation and contingent ranking procedures. The CVM study seeks individuals' WTP to avoid a number of health states associated with declining coastal water quality. From the CV study, the mean WTP to avoid an instance of gasroenteritis = Pte 7782. To convert the health effect into a per visit valuation, this value is multiplied by the probability of contracting gasroenteritis (0.058 to 0.087) = Pte 451-677 (€ 2.25 - 3.38).

A separate survey showed that there were 2760 million season visits to Estoril beaches, implying Pte 1.24 - 1.87 billion for cleaning up Estoril bathing water. The CR study suggests that WTP for an improvement of bathing water quality from 'bad' to 'average' quality is € 9.6 and that it is € 6.6 for an improvement from 'average' to 'good'. Table 2.2.4 reports the results.

Table 2.2.4 WTP for coastal water quality improvement in Portugal (1997)

WTP pp / pa € 1997	Sample size	Effect captured
48	195 to 401	Mainly recreation

Interestingly, the WTP in Portugal is above that in the UK. This may reflect the fact that more bathing will take place in the warmer climate of Estoril compared to the East coast of England.

The process of aggregation is not simple. Estimates for the proportion of the European population living within 50km of coastal waters in 1992, is 28% (RIVM 1998). This is equivalent to a population of 102.8 million living with 50 km of coastal waters. (This assumes a population of 367.43 million in EU15 in 1992). It is assumed that, this population use the coastal waters for bathing at least once a year as local residents or day trippers. This conservative assumption allows for reduced visits in colder waters and higher visit rates in warmer waters (e.g. in Estoril, average visits are 5 per year). Therefore, the WTP figures for direct use value, from Table 2.2.3, can be applied directly to this population as UK GNP per capita is broadly in line with the EU15 average. The higher WTP for Portugal probably reflects the high density tourism in that area.

A further breakdown of the population can be introduced by using the average EU12 percentage of coastal sites not meeting the Directive standard of 4% detailed in Table 2.2.2 and applying it to the EU15 population. We therefore have a population of 4.11 million living within 50km of bathing water below Directive standard and 98.69 million living within 50 km of bathing water above Directive standard in 1992. Clearly there may be some over-lap as some proportion of this population will live within 50 km of more than one coastal area

A total direct use value for maintaining bathing water at the EC Directive standard for EU15 in 1992/3 can now be calculated:

98.69 million *x* € 18.08 = € 1784 million

As said above, this value ignores direct use value from long-staying tourists and also non-use value. Thus, this value clearly under-estimates total benefits.

Benefits of full compliance with the EC Directive on Quality of Bathing Waters: an example

The methodology given above will also make it possible to develop scenario analysis if data on 2010 scenarios becomes available. As an example, it is assumed that the 'MF' scenario is that all coastal water will reach Directive standard by 2010 and the baseline is that 4% of coastal water will remain below Directive standard by 2010. Following the same procedure as above and taking account of the predicted population growth to 389 million in EU15 in 2010, then a population of 109 million will live within 50km of coastal waters in 2010. Under the MF scenario all will have Directive standard bathing water whilst under the baseline a population of 104.64 million will have Directive standard bathing water. Under the additional assumption that non-Directive standard bathing water has a zero value then the benefits of MF over baseline in 2010 can be estimated for EU15.

4.36 million *x* € 19.29 = € 84.1million

Where, 4.36 million is the additional population that benefit from Directive standard bathing water under MF and 19.29 is the adjusted WTP 1997 to maintain Directive standard bathing water due to the rising relative price of environmental quality (i.e. 18.08 x 1.005^{13}). Again this figure ignores a significant part of tourist direct use value and also non-use value.

Benefit of full compliance with the EC Directive for EU15 is given as the benefit to EU15 from maintaining currently compliant sites with EC Directive standard, plus the benefit of raising currently non-compliant sites to EC Directive standards, € 1.86 billion from € 1784 million + € 84.1million. The benefit estimates are likely to be an underestimate due to the omission of benefits related to tourism.

The Bathing Waters Directive dates from 1976 and hence predates the benefit - cost requirement in the Single European Act and Maastricht Treaty. Unfortunately it is not possible to estimate a B/C ratio for the full compliance to the Directive, due to lack of information on costs at the EU level and to the incomplete estimation of benefits. I
In 1994 the Directive was revised and the UK House of Lords (House of Lords, 1994, 1995) estimated the compliance cost to meet the original Directive for the UK only, to be UK £ 1.7 billion, € 2.6 billion (1997 prices) over 10 years, or roughly € 0.26 billion per annum. Whilst costs of the proposed revisions to the EC Directive range from:

- A Secondary treatment and filtration of effluent and dis-infection by UV radiation: € 310 - 759 million per annum
- B, C Different levels (B, C) of secondary treatment and UV radiation for discharges within a certain distance of bathing waters: B would cost € 215 - 510 million per annum and C would cost € 85 - 200 million
- D insignificant new requirement, negligible cost

B/C analysis of nutrient load reduction to the Baltic Sea

One major research project has conducted extensive valuation studies of eutrophication in the Baltic Sea (Turner et al. 1995, 1997). The study involved several contingent valuation and travel cost studies in the context of an assumed 50% nutrient load reduction programme which can legitimately be regarded as an 'MFR' scenario since such a programme has substantial costs of nearly € 4 billion pa. The results are set out in Table 2.2.5. The Polish CVM studies involved a beach survey (Zylicz, 1995a) and a household survey, with, in each case, respondents being asked their WTP for clean-up programme (Zylicz, 1995b). The first two surveys give very close results, but the surprising feature is the sheer size of the WTP figures. Taking the Swedish results as being typical of the west European economies and the Polish results as being typical of the EITs, Turner et al.(1997) estimate Baltic basin wide benefits from a clean-up programme to be some SEK69 billion per year, or some € billion 8.0 per annum compared to costs of € billion 3.6 pa. If non-respondents are treated as having implicit zero WTP for the clean up programme, then benefits reduce to SEK billion 37.9 pa or about € billion 4.4: benefits are still greater than costs though by a fairly narrow margin.

Table 2.2.5 Median Willingness to Pay to Improve the State of the Baltic Sea. (1994US$)

Study	Country	WTP	% Income
Zylicz et al, 1995a Beach survey, CVM	Poland	US$65-122 pp.pa	3-5%
Zylicz et al, 1995b household survey, CVM	Poland	US$30-73 pp.pa	1.3-3.3%
Mail survey, CVM	Poland	US$104-211 pp.pa	4.6-9.5%
Sandstr∧m, 1995, TCM beach use	Sweden	US$32 per trip	
S∧derqvist, 1995, CVM	Sweden	US$333 pp.pa	

Table 2.2.6 shows the costs and benefits estimated in the study, broken down by country.

Table 2.2.6 Costs and benefits of a 50% nutrient load reduction in the Baltic

Country	Nutrient Load Reduction %	Costs SWkr10^6 pa	Benefits SWkr10^6 pa	Net Benefits SWkr10^6 pa
Sweden	42	5300	20723	15423
Finland	52	2838	10799	7961
Denmark	51	2962	12376	9414
Germany	39	4010	8369	4359
Poland	63	9600	11631	1761
Russia	44	586	3488	2902
Estonia	55	1529	418	-1111
Latvia	56	1799	583	-1216
Lithuania	55	2446	923	-1523
Total	50	31070	69310	38240
(Total in €)		*€ 3591*	*€ 8011*	
Total when non-respondents given WTP=0		31070	37892	6822

Source:Turner et al. 1997. Assuming SEK 8.65117 = € 1 (1997 prices).

Total N and P loads to the 'Baltic proper' (the geographical zone used in the study) in 1993 were 860,000 tN and 33,000 tP per annum. Allocating benefits to N and P in the load ratios would then give the following WTP per tonne N and P[6]:

[6] Where; total nutrient load = 893,000 tonnes per annum. Benefits relate to a 50% reduction of nutrients, therefore we consider 446,500 tonnes nutrients per annum, of which N loads represent 96% and P loads are 4%. Total benefits are assumed to be € 8011 million and using the load ratios we estimate the benefit due to N is roughly 7690.5 x 10^6, whilst damage due to P is about 320.4 x 10^6. Dividing by the tonnes of N and P released to the Baltic, gives a WTP of € 17,885 per tN and for P € 19,421 per tP.

€ t/N WTP = € 17,885 / t
€ t/P WTP = € 19,421 / t

These are averages for the countries combined. It makes little sense to express WTP per tonne for individual countries since the questionnaire approach made it clear that the benefits would accrue from a combined programme rather than any unilateral action.

Overall, if we take a 50% nutrient load reduction as being an ambitious programme consistent with MFR assumptions, then an MFR scenario addressing eutrophication from all sources, suggests that benefits exceed costs.

3. Policy assessment

3.1 Policy package water stress

3.1.1 Key issues

Water availability

Data for water management suggest that significant numbers of people within the EU suffer 'water stress'. The cost-benefit analysis (see section 2: Benefit Assessment) indicated that WTP in the domestic sector was greater than the cost of new supplies for only the cheapest sources of supply, such as new groundwater schemes and re-use schemes. It was suggested there that this indicated the need for policy measures that did not focus on new investments in supply, but which stressed demand management and leakage control. The discussion is relevant to the *Water Resources Framework Directive* which, would aim to set a framework for:

> *integrated water management and coherent policies, e.g. between groundwater and surface water, and taking into account future water demands and the burden they will impose on the environment, network for the exchange of information, identifying water resource issues that are likely to emerge in the future, and collecting information.*

The focus is on river basin management planning and ecological and hydrological monitoring. As noted by Kopke (1996) future policy must embrace the reduction of wasted water as well as increasing new supplies.

Correct water pricing

The main requirement is to price water at long run marginal cost, (LRMC) as part of a longer run commitment to full social pricing. The correct pricing of water must account for the (i) extraction and distribution costs (ii) the environmental costs of low flow regimes and (iii) the opportunity costs of water use.

The basic principle of economic pricing of water is given by the formula:

$$P = MC + MEC + MUC$$

where MC is the (long run) marginal cost of supply (LRMC), MEC is the marginal external cost associated with water abstraction and disposal, and MUC is the marginal 'user cost' which is designed to reflect the cost of foregone future benefits because of use of the water today. MUC would tend to zero in a fully sustainable water supply and use system.

The first step towards correct water pricing is the removal of subsidies. Where cross-subsidisation of low income consumers is required this can be achieved by 'lifeline' tariffs, i.e. charging low prices for low consumption and very much higher prices for higher consumption.

Once the above is achieved, the recommended policy options are as follows:

3.1.2 Recommended policy initiatives for water availability

Water trading rights

Water should be allocated to its 'highest' value uses through market mechanisms involving, where possible, trading of water rights (assuming water is priced at LRMC). Such trading regimes may be localised but could also include broader national and even transboundary trades.

There seems to be no experience of water trading in the EU. A consultation process in the UK (results reported in ENDS report, June 1998) showed that none of the interested parties (industry, regulators, academia and general public) thought that a system of tradable rights was a viable instrument for water policy mainly for administrative reasons. However, water trading systems exist in USA, Australia, Mexico and Chile. Such measures are likely to show, that some agricultural uses of water in Europe are not commercially justified (Pearce, 1999). For example, if irrigation water was priced in this way, we doubt if some of the agricultural activity in water scarce areas would have an economic justification.

It is clear that tradable rights systems do not 'bite' in terms of economic and environmental efficiency until there is real scarcity. Trends in global warming, growth of population, and growth of tourism, all suggests this will be a serious problem in the future for parts of Europe, especially Mediterranean Europe. Implementing a system of tradable rights on an experimental basis, perhaps for irrigation water only (which is how the schemes largely operate in Australia and California), appears to be a potentially most effective measure.

Abstraction / use charge based on correct water pricing

Based on the above observations, we can conclude that after achieving full private costs (removing subsidies), demand management schemes, such as abstraction / use charge could be used to achieve the target of 200 litre per capita per day. These charges would need to have the following characteristics:

- to cover at least the clean-up and future investment costs, if the ideal case of estimating environmental damage costs proves impossible, charges should be differentiated by the location of abstraction (high for environmentally sensitive areas lower for less sensitive areas), differentiated by timing of abstraction (high in dry seasons lower in wetter ones), differentiated by unit of abstraction or consumption (progressive rates), and the revenues should be earmarked especially for the industrial users in order to encourage water saving technologies.

- the more detailed the instrument, the more difficult it will be to implement it. However, given the current discussions on the Water Resources Framework Directive, location specific instruments can be dealt with (the Directive suggests water basin based management tools). Whatever the design, the most crucial issue remains to be that of price elasticity of demand for water. In various studies over the years, the demand is found to be inelastic (*refer to Benefit Assessment Section 2*). This implies one of two policy solutions: (i) a very high charge so that consumption behaviour can be changed or (ii) a well designed earmarking scheme for the revenues of a lower charge system. The first option would be politically difficult to get acceptance, while the second one may face the opposition of the Ministries of Finance who would prefer to add the revenues to the general budgets.

Water meters

Metering of the water use by the domestic sector is the first step in charging for water use. Metering provides the basic information on the extent to which domestic use is a stress factor and helps with measuring the effectiveness of any policy instrument.

In addition, the experience with metering shows that the mere existence of meters and meter-based water rates encourage users to reduce their consumption with attached costs and benefits.

Pezzey and Mill (1998) report that metering in Europe is widespread, with 66% of households having meters in Denmark, 84% in Hungary, Spain and Sweden. But the UK has only 10% of households subject to metering (although this is rapidly changing) and Norway has only 5%. A policy of increasing the level of metering does not therefore appear to be called for outside of the UK. The general effect of introducing meters is to reduce consumption by 10% and perhaps more (Pezzey and Mill, 1998; Herrington, 1997).

The major water pricing and water metering initiative, however, must be in the irrigation sector (see correct water pricing discussion).

Harmonised VAT rates for water

VAT rates could be harmonised within each Member State so that water is not treated differently to other commodities. VAT is not a good proxy for environmental taxes, but if environmental charges are not considered feasible, VAT is a potential basis for second best environmental charging.

Moret, Ernst and Young (1996) note that water to domestic consumers is often supplied with a lower rate of VAT than the standard rate. This suggests, for example, that water consumption in France is 1-4% higher than would be the case if the standard French rate of VAT was charged. They cite Denmark and Finland as the only countries with VAT rates equal to the normal internal rate.

3.1.3 Recommended policy initiatives for water quality

N-fertiliser tax with payments to organic crops

Due to the inelasticity of demand for fertilisers, it is likely that a fertiliser tax will probably not change the quantity of fertiliser used. Therefore we recommend a hypothecated tax on fertilisers, where the revenues raised are used to clean up the water. We also suggest giving payments to farmers to switch crops, i.e. to organic produce. Thus, a suitable policy option to address one cause of water contamination would be a fertiliser tax plus payments to organic crops.

Some evidence of the effects of a nitrogen fertiliser tax can be gauged from the experience of Austria's levy on fertilisers 1986-1994 (it was abolished on Austria's entry to the EU in 1995). Hofreither and Sinabell (1998) show that the levy was first introduced to secure revenues to assist with the financing of net grain exports from Austria in the 1980s. They find a price elasticity of demand for N-fertilisers of –0.2. After the introduction of the levy, fertiliser consumption fell in Austria, but Hofreither and Sinabell show that the direct price effect of the levy contributed to only part of this change. Moreover, much of the impact of the levy was offset by reductions in the base price of fertilisers. What effect there was, they argue, arose from the fact that the levy acted as a signal to farmers, and the general public, about the 'scarcity' of the Austrian environment. The Austrian experience confirms the wider view that fertiliser taxes are likely to be imperfect instruments for securing reductions in nitrogen use so long as the focus is on the price elasticity alone. But it also suggests that economic instruments can have a 'signalling' effect, which leads to changes in behaviour.

Waste water effluent charges

In most EU Member States, effluent charges are implemented, even though in most cases these charges aim to recover the cost of monitoring or clean-up rather than to change polluting behaviour. Effluent charges do not therefore currently cover the 'full costs' of discharge (i.e. LRMC + MEC, see correct water pricing discussion).

In a 1988 survey of industrial polluters in the Netherlands, about 66% said that the reason for their reduction of effluent discharges was the effluent charge. Between 1975 and 1990, the government effluent charge increased by 250%. The discharge of oxygen-demanding materials decreased by 25% between 1975 and 1990, while heavy metal discharges decreased by 45% on average during the same period. Correspondingly, the total income from the charge decreased over the same period.

We assume that 66% of the decrease in effluents between 1975 and 1990 was due to the charge, i.e. 16.5% decrease in oxygen-demanding materials and 29.7% decrease in heavy metal discharges. Considering that this is the result of an increase in the effluent charges of 250% during the same period, the elasticity is -0.07 and -0.12 for oxygen demanding materials and heavy metals, respectively.

3.1.4 Multiple benefits

The correct pricing of water at long run marginal social cost will help to improve the issue of water quality by reducing chances of low flow. Water quality will also benefit from policies targeted at other environmental policies, including: i) methane emissions from livestock (see Technical Report on Climate Change), ii) ammonia tax (see Technical Report on Acidification, Eutrophication and Tropospheric Ozone); iii) agricultural policy reform, that leads to a reduction in the use of fertilisers and pesticides, iv) pesticides tax with a subsequent fall in the use of toxic pesticides.

3.2 Policy assessment water stress

Water availability: i) Demand management schemes i.e. correct water pricing by abstraction / use charges with water metering and ii) tradable water rights for agricultural sector (irrigation water)

Groundwater quality with known contamination: There is no need to introduce a tax because the contamination has already happened. There are a number of well defined technological options available to clean the water. Although, there may be some instances where the issue cannot be resolved. A monitoring system is necessary.

Ground water quality with threat of contamination: In order to regulate by means of a tax it is necessary to identify the cause of the contamination. I.e if it is due to nitrate run off, then a fertiliser tax is recommended. Thus a good monitoring system must be in place in those areas where the threat of contamination is greatest.

3.2.1 Causal criterion

Table 3.2.1 lays out the driving forces behind the problem of water management, the underlying causes of these driving forces are also identified.

Table 3.2.1 Driving forces and underlying causes of the water management problem

	Driving forces	Underlying causes		
	U = related to water use, Po = relates to pollution	MF	IntF	ImpF
D1	Poorly managed tourism (U + Po)			
D2	Dependence on groundwater for drinking (65% in '90)(U)		X	
D3	Irrigation, especially in Southern Europe agriculture (U)		X	
D4	Excessive use of N and P fertilisers and manure in agriculture (Po)	X		
D5	Use of persistent pesticides in agriculture (Po)	X		
D6	High livestock density (Po)	X		
D7	Surface water pollution by untreated sewage (nutrients, heavy metals, and organic matter) from industry (Po)			X
D8	Surface water pollution by untreated sewage (nutrients and organic matter) from households, (Po).			X

X = main underlying cause, MF = market failure, IntF = intervention failure, ImpF = implementation failure. Note that for driving force D1, the main causes are due to the growth in population and real income.

Water availability: the correct pricing of water, at long run marginal cost ensures that the cause of excess demand is addressed. If irrigation water is priced in this manner, we doubt if some of the agricultural activity in water scarce areas would have an economic justification.

Water quality: the fertiliser tax may not address the cause. This is because the tax will probably not change the quantity of fertiliser used, due to the inelasticity of demand. Unlike pesticides, fertilisers are generally uniform in their effect on the environment, so there is no basis for a fertiliser tax based on toxicity to reduce the contamination due to fertilisers. In this instance we argue for a hypothecated tax on fertilisers, where the revenues raised are used to clean up the water. We suggest giving payments to farmers to switch crops, i.e. to organic produce. Thus a fertiliser tax plus payments to organic crops policy option will address one cause of water contamination. The pesticide tax discussed in 'chemical risks' is also relevant for water quality.

If the pesticide tax reduces use of highly toxic and persistent pesticides, then such a measure can be said to address the underlying cause.

3.2.2 Efficiency criterion

Benefit-cost ratio

Due to the absence of scenarios for the issue of water management, B/C ratios are given only for the recommended policy actions.

Water availability: B / C ratios for new supply schemes are as follows:

Ground water schemes:	1.5 - 3.0	Reservoir:	0.3 - 0.6
Re-use schemes:	0.6 - 1.3	Barrage:	0.3 - 0.6
Transfer schemes:	0.5 - 1.0	Desalination:	0.15 - 0.3
		National grid:	0.12 - 0.25

There is a WTP on the part of EU residents for increased water supply but comparison of this WTP with the costs of new supplies suggests fairly strongly that new schemes are not generally justified in cost-benefit terms. The low B/C ratios suggest that the only option that makes economic sense are groundwater schemes. The high B/C ratios indicate that groundwater schemes, re-use schemes and some transfer schemes are justified in cost benefit terms. New reservoirs, desalination and nation-wide water grid schemes would not be justified[7].

The above observations suggest that water stress problems are not best resolved by new supply schemes. Instead, the focus of policy should be on correct water pricing, demand management schemes and leakage reduction.

Water quality: B / C ratios for groundwater contamination clear up range from 1.5:1 to 15:1 (*refer to Section 2: Benefit assessment*)
Benefits are an underestimate because clean-up of water will also have major effects on the improvement of bathing water.

Cost effectiveness

Correct pricing of water will alleviate water stress and it will also help to improve water quality by reducing chances of low flow (see above for the low B / C ratios of new supply schemes). Correct pricing of water with trading of water rights is a cost effective policy option that ensures water will be allocated to 'highest' value use.

Public opinion

Eurobarometer (1995) shows that 69% of respondents believe that decisions of the environment should be taken at the Community level, rather than at national level. It also shows that most Europeans are prepared to change their consumption behaviour as a step to slow down or perhaps even stop the deterioration in the environment as a whole. Despite this, there is currently strong opposition to the compulsory introduction of water metering by households.

3.2.3 Administrative complexity

Water availability: administrative complexity for abstraction / use charges is low as regulators already exist. There is some conflict however, between the role regulators are playing in lowering prices due to the liberalisation of the market and the need to raise prices due to environmental concerns.

[7] It is important to understand that the B/C ratios are based on very broad average costs and benefits. It does not mean that a new reservoir, for example, in some location is never justified

The most serious problem with a system of tradable water rights is that where there is a tradition of riparian law, it effectively means that the legal system needs to be changed considerably. However, where there is a tradition of appropriative water rights, as in California, introducing tradable water rights is not a serious issue.

A consultation process in the UK showed that none of the interested parties (industry, regulators, academia and general public) thought that a system of tradable rights was a viable instrument for water policy mainly for administrative reasons (results reported in ENDS report, June 1998).

Water quality: paying farmers to switch crops, i.e. changing an output subsidy to a payment for an environmental service seems to fit with recent changes in the CAP. Thus, administrative complexity is considered to be low as the institutions are already in place.

3.2.4 Equity criterion

Water availability: experience with proper pricing of water suggests that the 'poor' need not suffer as water costs may decline as more efficient use is practised. If distributional problems arise, 'lifeline' pricing can be practised whereby a rising tariff for high consumers is used to subsidise low volume (users paying a below marginal cost tariff).

Water quality: the distributional incidence of the fertiliser tax is not known, but the occupational group will be affected (i.e. agricultural sector). In the long run the fertiliser tax will affect the price of food. As a proportion of individual's income it is expected to be negligible.

3.2.5 Jurisdictional criterion

Water quality: if Member States have the freedom to manage their own distribution of subsidies then the payments to farmers to switch crops may not need be a centralised issue afterall. However, competition could be affected which then suggests there is an argument for centralisation.

Water availability: correct water pricing will affect competition. Water is a major cost to some sectors, i.e. agriculture. Thus this policy option may be a centralised issue even though water scarcity is a localised environmental problem.

Macroeconomic effect

Details provided in *Technical Report on Socio-EconomicTrends, Macro-Economic Impacts and Cost Interface.*

3.3 Policy package coastal zones

3.3.1 Key issues

Beach quality is generally covered by existing Directives, i.e. Urban Waste Water, Bathing Water, but the issue of implementation is outstanding.

Fisheries suffer the classic problems of open access resources, this suggests the eventual introduction of tradable fishing quotas is required. This implies wholesale reform of the CFP, which currently provides subsidies to fishing.

Development of coastal areas: the introduction of a development tax for marinas, sports complexes with requirements to offset biodiversity and habitat damage or a system of transferable development rights. See also *biodiversity loss*.

3.3.2 Recommended policy initiatives

Land use planning

Land use planning which contains regulations on what type of activity or structures can and cannot take place has been thought of as the principle mechanism for coastal zone management. Such plans exist in most EU countries, however, their implementation differ between different sites and in connection with other related policies.

The recent report on the EC's Integrated Coastal Zone Management (ICZM) project remarks that land use planning alone is not a sufficient vehicle for ICZM due mainly to (EC, 1999):

- their lack of scope: narrow focus on terrestrial parts of the coastal zone; narrow focus on structures rather than activities; weak linkages between the land use planning and sectoral planning systems and limited applicability of controls outside urban areas;
- their lack of flexibility: difficulty of adapting the plans to local needs; difficulty with renewing plans due to resource constraints and appeal processes being time consuming and costly

In order to guarantee adequate coastal zone management other mechanisms are required, recommended actions are listed below. Measures targeted at biodiversity loss are also relevant for coastal zone management.

Transferable development rights

The use of transferable development rights provides a market mechanism for simultaneously protecting landscapes and natural endowments and encouraging development in appropriate locations.

The key to achieving development in an efficient and equitable fashion seems to lie in separating the market for the right to develop from the right to use land. Transferable development rights (TDRs) operate in the following way: in a certain jurisdiction (i.e. country, region), 'sending' areas where development should be limited or prohibited are identified according to some criteria (e.g. landscape uniqueness, species protection, conservation of ecosystem processes, role in flood control, conservation of farmland). Also, 'receiving areas' are identified, these are areas where development is to be facilitated and encouraged. Development rights are identified as follows: the land owners in the sending areas are given development units, designated per unit area, e.g. one residential unit of a certain size per 10 acres, but these can only be exercised in the receiving zone. If developers in the receiving zone wish to exceed the standard development densities (or other norms to be defined) , then they must purchase the right to do so in the development rights market. The legal instrument developed to transfer development rights in the US is an easement document and a deed of transfer which are recorded and approved by the Planning Board.

TDRs are in existence on in the US. Montgomery County, Maryland and the Pinelands, New Jersey have successfully implemented a TDR programme. Pizor (1986) summarises their experience and suggests that the success of TDRs depends on (a) an institution, e.g. planning department, with responsibility for explaining and promoting the concept, providing price data and facilitating trades; and (b) the receiving areas must be well sited for immediate development. Necessary infrastructure should be in place and the receiving sites should be in market areas which, from a market perspective, are most suitable and attractive for development; and (c) guarantee of delivery, i.e. if the developer pays for rights then s/he can exercise them; and (d) prohibitions on development in the sending areas must be comprehensive and mandatory, and the TDR must be mandatory on lands suited for preservation; and (e) provide a bank as a buyer of last resort for TDRs of sending landowners in the event of recession; and (f) reduce regulatory complexity, i.e. if developers have confidence that they can use the rights profitably, this will increase the probability that they will be used; and lastly (f) in so far as it is possible, design TDRs to fit the needs of the development chain, rather than the needs of planners.

Owner liability and performance bonds against oil spills

Although issue of oil spills is not thoroughly investigated in this study an initial discussion is presented in *Technical report on Nucclear Accidents*. For this reason, we present potential measures that could reduce the risk of oil spills.

Where tanker and installation owners are clearly identifiable, and cause and effect can be established, there is scope for the use of *liability* mechanisms, and *performance bonds*. These are discussed more fully in Annex 3: major accidents. Other measure to encourage oil spill prevention include:

Oil pollution combating charge: Finland levies an oil pollution combating charge on imported oil. The charge is levied on imported petroleum in the amount of FIM 2.20 (€ 0.39) per tonne of oil and is differentiated on the basis of the environmental safety of the cargo boats. For instance, the charge rate is double for oil transported in cargo boats without double hulls. A cargo boat of 100,000 tonnes is liable to a charge of about FIM 160,000 (€ 28,013) per journey; for vessels of the same size and double hulls, the charge will be about FIM 80,000 (€14,007). The charge is earmarked for environmental measures, such as cleaning oil spills (North Council of Ministers, 1994).

The oil pollution tax rebate (from one cargo trip) is sufficient to cover the extra interest expense incurred in producing a vessel with double hulls (which is about 15% more expensive than producing a vessel with a single hull). The revenue of this charge, which amounted to FIM 36 million (€ 6.9 million) in 1992, accrues to the oil-protection fund and is earmarked to buy equipment for cleaning oil spills.

Similarly, in Sweden, Gothenburg harbour levies harbour charges differentiated according to type of ballast tanks. Since 1993, cargo boats above a certain tonnage with segregated ballast tanks and/or double hulls arriving at the harbour have been granted rebates in the harbour charge, this is calculated on the basis of gross tonnage excluding segregated ballast tanks.

The rebates in Sweden, which range from 15 to 20%, provide incentives to construct vessels with segregated ballast tanks. Currently, the dominant proportion of petroleum waste in the North Sea is reported to stem from illegal discharges from vessels without segregated ballast tanks rinsing out their oil tanks (North Council of Ministers, 1994).

Since 1979, all operators on the UK continental shelf are required to have oil spill contingency plans approved by the Department of Transport and of Trade and Industry. Within 25 miles of the coast and in environmentally sensitive areas, operators must be prepared for rapid response to a spill and to minimise impact on shore. All operators are members of the Offshore Pollution Liability Association, which makes funds up to US$ 100 million (€ 91.1 million) per incident available to pay the operator's liability for damage and remedial action. The UK supported increases in the liability and compensation ceilings in the framework of the 1984 and 1992 protocols following the Amoco Cadiz and Exxon Valdez disasters (OECD, 1994).

3.3.3 Multiple Benefits

Policies targeted at coastal zones specifically, such as land use planning, tradable development rights and owner liability / performance bonds against oil spills, will also benefit biodiversity loss. Whilst, coastal zones will be affected by varying degrees from policies targeted at other environmental issues. We list a selection of policy options recommend for other environmental issues, expected to provide multiple benefits to coastal zones:

i) N fertiliser tax (see *water management*);
ii) effluent charges (see *water management*);
iii) agricultural policy reform (see *biodiversity loss*);
iv) mitigation banking (see *biodiversity loss*);
v) owner liability and performance bond schemes against oil spills (see *major accidents*);
vi) methane tax on livestock (see *climate change*);
vii) ammonia tax on livestock (see *acidification*).

Eutrophication in regional waters, such as the Baltic Sea can be reduced by policy initiatives that ensure nutrient reduction. The preliminary cost benefit analysis of a 50% nutrient load reduction in the Baltic Sea shows benefits exceed costs, with a ratio of 2.2. This suggests that the seemingly ambitious programme is justified in cost benefit terms.

These policies will benefit coastal zone areas due to expected reduction of nutrient loads to waters and the protection of coastal lands. Unfortunately, there is insufficient data to estimate what this spill over effect would be.

3.4 Policy assessment coastal zones

Land use planning is already widely used by most Member States. However, in order to guarantee adequate coastal zone management other mechanisms are required, such as transferable development rights, (TDR); This scheme provides a market mechanism for simultaneously protecting landscapes and natural endowments and encouraging development in appropriate locations.

Policy initiatives targeted at other environmental concerns will also contribute to coastal zone management, these include:

- agri-environmental schemes (see biodiversity loss);
- fertiliser tax with payments to switch to organic crops (see water management);
- increased 'owner liability' to include not just loss of livelihood and property but also disamenity and, ultimately, non-use values (see major accidents).

3.4.1 Causal criterion

Table 3.4.1 presents the driving forces behind the coastal zone management problem, the underlying causes are also identified.

Table 3.41 Driving forces and underlying causes of coastal zone management problems

		Underlying causes		
		MF	IntF	ImpF
D1	Recreation / tourism, hunting	X		
D2	Urbanisation and traffic			
D3	Industrial, fishery and aqua-culture growth			
D4	Shipping and offshore activities			X
D5	Agricultural intensification, fertiliser and pesticide use		X	

X = main underlying cause, MF = market failure, IntF = intervention failure, ImpF = implementation failure. Note that for D1, D2 main causes also include population growth and growth in real income, whilst for D3, growth in real income is the main cause.

The policies recommended for coastal zone management, biodiversity loss and water management, are measures which can be designed especially to target the most important sources of pressure to coastal resources: urbanisation, transport, industry, agriculture, fisheries, tourism. Oil spills are dealt with directly in 'major accidents' through the 'owner liability' legislation, whereby the scope of 'damage' is increased to include property and direct economic losses to disamenity and ecological damage and even to non-use values. Clearly, the analysis presented above mainly focus on water quality aspects related to eutrophication and bathing, and does not adequately cover fishing issues. Further work would be required on these issues.?

3.4.2 Efficiency criterion

Scenarios are not established for coastal zone management in this study. However, B/C ratios nutrient control initiatives near Baltic Sea are provided.

B/C analysis for the EC Directive on Quality of Bathing Waters

Although limited costs and benefits information exist, to estimate the B/C ratio for full compliance of the EC Directive bathing water quality would be too arbitrary and uncertain. Thus, such analysis has not been undertaken in this report.

B/C ratio for policy initiatives to reduce nutrient loading to Baltic Sea

Eutrophication in regional waters, such as the Baltic Sea can be reduced by policy initiatives that ensure nutrient reduction. The preliminary cost benefit analysis of a 50% nutrient load reduction in the Baltic Sea shows benefits exceed costs, with a ratio of 2.2. This suggests that the seemingly ambitious programme is justified in cost benefit terms.

Cost-effectiveness:

Transferable development rights; TDRs ensure development takes place in an efficient manner by separating the market for the right to develop from the right to use land. In return, for development restrictions on private holdings, landowners are awarded with alternative rights in designated growth areas, located outside the environmentally sensitive areas. These rights can be used for development or if a market exists, they can be sold for profit to other developers who want to develop in this zone.

Fishing quotas: one of the main objectives of quota-based fisheries management is to improve economic efficiency by reducing over capitalisation. The quota is set according to total allowable catch. Participants then use the most optimal configuration of inputs (i.e. capital, labour, time) to achieve their individual quota. Fishers will chose to fish when the price is high opportunity cost of time low and the climatic conditions safe, such behaviour contributes to higher socio-economic gain from the resource in the long run, OECD (1996). Trading quotas means the more efficient operators buy out the less efficient operators, thus the fleet will be composed only of the most efficient operator. This reduces the unit costs of operation, whilst also compensating the marginal producers.

Countries that implement individual transferable quotas for fisheries management combine them with input restrictions to prevent certain types of environmentally damaging gear to be used. The introduction of quotas may lead to increased technological efficiency that could threaten marine ecosystems, but on the other hand, greater gear efficiency may result in more investments to reduce by-catch.

Public opinion

Although public opinion regarding the issue of coastal zone management is not known with certainty, the results of the Eurobaromter (1995) suggest that the European population may be in favour of the suggested measures as a means to managing coastal regions in a sustainable manner.

3.4.3 Administrative complexity

Transferable development rights. Institutions are most likely to be in place, for example, those responsible for land use planning could oversee this scheme. It would be responsible for explaining and promoting the concept, providing price data and facilitating trades. Areas for development should have the necessary infrastructure in place and should be most suitable and attractive for immediate development. It may be necessary to set up an appropriate legal structure to zone land and enforce restrictions.

Fishing quotas: Any tradable quota system needs to be flexible to allow review and modification according to biological and economic (equity) requirements. Institutions must oversee the initial distribution of quotas through grand-fathering and auctioning. Quota management can be subject to cheating and / or under reporting when local conditions make it difficult to monitor catches. The current EU Common Fisheries Policy suffers from poor monitoring at the national level.

3.4.4 Equity criterion

Transferable development rights: These schemes have the potential to operate in an equitable manner if a mature market exists for the land with development rights, such that, compensated land owners have the option to sell their rights at a profit.

Fishing quotas: Trading quotas automatically compensates the marginal producers removed from the fleet. In reducing the fleet to the most efficient operators a quota scheme may have beneficial effects on population management. But experience has shown that quota systems can exclude small-scale and independent fishers from fisheries, which fall increasingly under the control of large corporations.

Schemes for conflict resolution between fishermen and wildlife could be based on capturing some of the WTP for wildlife. For example, case studies estimating WTP for Cornish grey seals and Mediterranean monk seals exist, if their WTP is captured it could be used to compensate fishermen in order to avoid 'hot spots' of conflict, i.e. a kind of marine set-aside.

3.4.5 Jurisdictional criterion

Coastal zone management is predominantly a local issue. Centralisation of policies would only be necessary if some Member States were reluctant to act on their own.

Macroeconomic effect

Details of macroeconomic effects are presented in Technical Report on Socio-Economic Trends, Macro Economic Impacts and Cost Interface.

Appendices: technical documentation of the methodology and tables with Member State specific information

Annex A: Water supply and sectoral demand in the EU-15, 1995 - 2010.

1995	Inh.	Inland resources	irrigation area	Total abstraction	public	irrigation	industry	cooling
	Mio	Mio m³	km^2	Mio m³	%	%	%	%
Austria	7.8	84000	40	2360	33	9	21	38
Belgium	10.1	16500	10	7015	11	0	3	86
Denmark	5.2	6115	4750	915	49	38	9	4
Finland	5.1	110200	640	3345	13	2	33	52
France	58.2	188000	15250	40641	15	12	10	64
Germany	81.2	164000	4750	58862	7	3	11	79
Greece	10.3	60500	13400	5040	12	83	4	1
Ireland	3.6	52198	0	1212	39	15	21	26
Italy	57.4	175000	27100	56200	14	57	14	14
Luxembourg	0.4	3205	0	57	59	0	25	16
The Netherlands	15.4	91000	5700	12676	8	1	4	87
Portugal	9.9	65305	6320	7288	8	53	3	36
Spain	39.3	117290	36600	33289	12	60	4	24
Sweden	8.8	173975	1150	2709	35	6	55	4
United Kingdom	57.9	145038	1080	12117	52	14	7	27

2010	Inh.	Inland resources	irrigation area	Total abstraction	public	irrigation	industry	cooling
	Mio	Mio m³	km^2	Mio m³	%	%	%	%
Austria	8.2	84000	40	2244	32	10	20	37
Belgium	10.5	16500	10	6609	11	0	3	86
Denmark	5.4	6115	4750	1051	48	39	13	0
Finland	5.4	110200	640	3151	13	2	33	52
France	62.5	188000	15250	38812	15	13	10	63
Germany	86.9	164000	4750	56098	7	4	11	78
Greece	11.1	60500	13400	4718	13	83	4	1
Ireland	3.8	52198	0	1209	38	18	19	24
Italy	56.8	175000	27100	52724	14	58	14	14
Luxembourg	0.4	3205	0					
The Netherlands	16.8	91000	5700	12003	8	1	4	86
Portugal	10.2	65305	6320	7646	7	57	3	32
Spain	40.2	117290	36600	31624	12	61	4	24
Sweden	9.4	173975	1150	2623	35	8	53	4
United Kingdom	59.4	145038	1080	11476	51	15	7	26

Source: ETC-IW, 1999.

Water demand for irrigation relative to the amount of available net precipitation and to the amount of annual resources (net precipitation and import)

	Precipitation	evaporation	Net precip	Allocated to irrigation	irrigation to net precip	Irrigation to Resources
	mm	mm	mm	mm	%	%
Austria	1169	513	656		0	0
Belgium	819	444	375		0	0
Denmark	667	526	141	74	17	17
Finland	657	340	317	125	0	0
France	816	490	326	322	6	5
Germany	790	490	300	384	3	2
Greece	849	492	357	312	20	16
Ireland	1150	450	700		0	0
Italy	983	428	555	1188	28	27
Luxembourg	818	444	374		0	0
The Netherlands	737	477	260	22	17	2
Portugal	886	474	412	607	18	9
Spain	662	432	230	546	25	25
Sweden	700	320	380	151	0	0
United Kingdom	1077	484	593	1593	2	2

Source: EEA, 1999.

Annex B: Waste water treatment

In line with [EEA, June 1999], the baseline scenario for wastewater treatment in the EU-15 is the full implementation of the Urban wastewater Treatment (UWWT) Directive [CEC 1991]. This directive describes the requirements for the collection and treatment of urban wastewater for the EU-countries. Table B.1 shows that the Directive distinguishes sizes of agglomerations and type of areas (sensitive versus non-sensitive).

Table B.1 Main requirements and moments of implementation for urban waste water collection and treatments according to the UWWT-Directive

Agglomeration (x 1,000 inhabitants)	Non-sensitive areas: Sewerage, Connection before:	non-sensitive areas: secondary treatments before	sensitive areas: sewerage, connection before:	sensitive areas: secondary treatment before	sensitive areas: allowed maximal concentrations (mg/l)	
					Ntot	Ptot
< 2	Not required	not required	not required	not required		
2 – 10	2006	not required	2006	2006		
10 – 15	2006	2006	1999	2006	15	2
15 – 100	1999	2001	1999	1999	15	2
> 100	1999	2001	1999	1999	10	1

Source: [CEC 1991]

For the implementation of the UWWT-Directive it was assumed that the whole EU-15 consists of sensitive areas. Regarding the year 2010, this focus on sensitive areas has the following implications:
- The agglomeration category 2,000 – 10,000 inhabitants should have secondary treatment in 2010;
- The agglomeration categories larger than 10,000 inhabitants should meet specific Ntot and Ptot concentrations in 2010. This implies certain forms of tertiary treatment.

Unlike [EEA, 1999] where results have been reported by national contacts to the European Waste Water Group, the presented UWWT-Directive implementation costs are constructed by taking the following steps:

a. Population data. The following population data have been gathered:
   1 Total, urban and rural population data: [NTUA, 1998], [UN, 1996], [World Resources, 1997]
   2 Data on population living in cities larger than 100,000 inhabitants: [UN, 1996]
b. The gathered data (total, urban, rural, larger than 100,000 inhabitants) are distributed over the 5 relevant agglomeration categories listed in Table B.1.
c. Sewerage connection data: [OECD, May 1999], [OECD, 1997]. Distribution of the number of unconnected people over the 5 relevant agglomeration categories. It is assumed that the possibility to encounter households without sewerage connection in densely populated agglomerations is much smaller than for the rural agglomeration category.
d. Treatment (primary, secondary and tertiary) data: [OECD, May 1999], [OECD, 1997]. The distribution of the treatment over the agglomeration categories starts with tertiary treatment in the largest agglomeration category (> 100,000 p.e.)

With this distribution of the total populations over 25 sources (5 agglomeration categories multiplied with 5 'treatment' options) simulations have been carried out in the computer model MOSES. In order to carry out simulation the following extra steps had to be taken:

e. Emission factors. Dutch emissions have been coupled to the number of inhabitants (p.e.). The following emission factors have been used: 45.4 kg COD/p.e./year, 0.7 kg Ptot/p.e./year, and 4.0 kg Ntot/p.e./year. In terms of concentrations, the values are: 558 mg COD/l, 9 mg Ptot/l, and 49 mg Ntot/l
f. Emission growth. Emission growth has been coupled to the varying number of inhabitants. It has been assumed that the emission factors will not change in time and that the change of the total population in the period 1995-2010 is distributed equally over the 25 sources
g. Emission abatement database. In order to use MOSES the 25 different sources have to coupled to strings of technology presented in the three databases presented in Annex II.

In tables B.2, the results for the EU-15 in 1995 and 2010 are presented. The treatment costs are presented in prices 1997 and include the costs for sewerage connection.

Table B.2a Implications of the implementation of the Urban Waste Water Directive (common in the BaseLine, TD, and AP): 1995, before implementation.

	1995			
Country	Annual costs ECU/year/inh	COD efficiency	N efficiency	P efficiency
Austria	31	0.35	0.24	0.22
Belgium	40	0.19	0.14	0.10
Denmark	49	0.78	0.54	0.48
Finland	41	0.65	0.46	0.54
France	41	0.54	0.36	0.34
Germany	46	0.68	0.48	0.50
Greece	25	0.09	0.06	0.05
Ireland	25	0.09	0.07	0.05
Italy	42	0.43	0.29	0.27
Luxembourg	0	0.00	0.00	0.00
TheNetherlands	49	0.80	0.57	0.48
Portugal	23	0.12	0.08	0.07
Spain	34	0.35	0.23	0.19
Sweden	49	0.80	0.57	0.68
United Kingdom	45	0.68	0.47	0.40
EU tot	41	0.54	0.37	0.35

Table B.2b Implications of the implementation of the Urban Waste Water Directive (common in the BaseLine, TD, and AP): 2010, after implementation.

	2010			
Country	Annual costs ECU/year/inh	COD efficiency	N efficiency	P efficiency
Austria	46	0.72	0.48	0.53
Belgium	54	0.83	0.69	0.77
Denmark	52	0.79	0.64	0.70
Finland	44	0.69	0.55	0.59
France	43	0.63	0.43	0.48
Germany	48	0.78	0.63	0.70
Greece	39	0.67	0.43	0.47
Ireland	34	0.48	0.26	0.29
Italy	55	0.82	0.69	0.77
Luxembourg	0	0.00	0.00	0.00
The Netherlands	49	0.80	0.65	0.62
Portugal	48	0.75	0.35	0.39
Spain	39	0.54	0.49	0.54
Sweden	51	0.80	0.67	0.74
United Kingdom	53	0.79	0.70	0.78
EU tot	48	0.73	0.59	0.65

Annex C: Animal Husbandry

Based on livestock units (LU), animal density in the EU is almost 1 LU per ha. As a rule of thumb one LU produces 60 to 80 kg N per year, cattle produce more than pigs and poultry. Hence, the requirement of the Nitrate Directive – not more than 170 kg N/ha manure should be spread on the land – should not pose the EU as a whole a problem. However, if measured at farm level, about half of the livestock population in the EU is kept at holdings exceeding 1.5 LU per ha. Also, due to the benefits of economies of scale, these intensive animal husbandry holdings tend to concentrate in a few European 'hot spots'. In these 'hot spots' manure production often exceeds the carrying capacity of soil and surface water.

Manure releases ammonia and methane, which are agents for acidification, eutrophication, and climate change. The Technical Report on the acidification issue discusses several technological solutions that minimise air emissions from stables and fields after manure spreading. This Annex assesses alternatives to these by setting limits to animal densities on farms and in regions.

Agro-economic investigations provided the regional data for these 'hot spots' of EU intensive animal husbandry (LEI, 1999). The database was also used to assess the impact of a series of limitation scenarios. Table C.1 gives the 1993 animal population in the 'hot spot' regions and the estimated manure disposal. The regions account for more than 50% of the EU-15 pigs population, while their utilised agricultural area (UAA) make only 13% of the EU-15 total of 128 million ha. Manure disposal exceeds the Nitrate Directive limit in Belgium and the Netherlands.

Considering the additional artificial fertiliser application of another 75 kg N/ha, the 170 kg N/ha limit of the Nitrate Directive is exceeded in all of the regions. This limit only applies to animal manure. One should bear in mind that the limit is a generic one and does not relate to ecological or human toxicological risks. Groundwater below sandy soils is most vulnerable to nitrate contamination. Application of fertilisers to the level of 170 kg N/ha will result in groundwater concentrations exceeding the limit values in the EU Drinking water directive.

Table C.2 through C.5 provide a quantification of the consequences of four restrictive policy options. Limiting the animal densities at farms stops intensive pigs and poultry husbandry, as manure disposal is only allowed on the farmer's own land. Only mixed farming sustains such restrictive policy options. Limiting the animal densities in regions allows for the transportation of manure to nearby agricultural land. However, broadening the scope of manure disposal is associated with extra transportation costs.

The policy options are presented in a declining order of impact. However, limiting animal density to 1.5 LU/ha on a regional level – the mildest restriction – would still stop pigs and poultry husbandry in Belgium and the Netherlands.

Table C.1 Hot spots of intensive animal husbandry: livestock population, utilised agricultural area (UUA) and animal density in the EU-15 in 1993.

Country/region	*Cattle*	*Sheep goats*	*Pigs*	*Poultry*	*Livestock*	*UAA*	*Animal density*	*manure disposal*
		Million animals			Million LU	Million ha	LU/ha	kgN/ha
Belgium	3,2	0,2	7,1	28,6	4,2	1.3	3,2	232
Denmark	2,2	0,2	11,6	19,8	4,6	2.7	1,7	105
Niedersachsen	3,0	0,3	7,2	39,1	4,3	2.7	1,6	113
Nordrhein Westfalen	1,8	0,2	5,8	9,9	2,8	1.6	1,7	118
Galicia	0,9	0,3	0,6	7,6	0,9	0.6	1,4	122
Cataluna	0,6	0,9	4,5	38,8	2,1	1.1	1,9	109
Loire	2,8	0,4	1,5	56,7	2,9	2.2	1,3	110
Bretagne	2,4	0,2	7,7	100,4	4,7	1.8	2,6	171
Lombardy	1,9	0,2	2,9	19,0	2,2	1.1	2,0	155
The Netherlands	4,8	2,1	15,0	98,4	8,5	2.0	4,3	284

Source: Eurostat, LEI; adaptation RIVM

Table C.2 Hot spots of intensive animal husbandry: livestock population, utilised agricultural area (UUA) and animal density in the EU-15 in 1993, under limitation scenario A: limit animal density to 1 LU/ha on farm level

Country/region	*cattle*	*Sheep goats*	*pigs*	*poultry*	*livestock*	*UAA*	*Animal density*	*Manure disposal*
		Million animals			Million LU	Million ha	LU/ha	kgN/ha
Belgium	1.9	0.2	0.0	0.0	1.1	1.3	0.9	90
Denmark	1.7	0.2	0.3	0.6	1.1	2.7	0.4	42
Niedersachsen	2.4	0.3	0.2	1.2	1.6	2.7	0.6	59
Nordrhein Westfalen	1.4	0.2	0.3	0.5	1.0	1.6	0.6	58
Galicia	0.6	0.3	0.0	0.1	0.4	0.6	0.6	64
Cataluna	0.2	0.9	0.0	0.0	0.2	1.1	0.2	18
Loire	2.2	0.4	0.0	0.6	1.4	2.2	0.6	66
Bretagne	2.0	0.2	0.0	0.0	1.2	1.8	0.7	71
Lombardy	1.0	0.2	0.0	0.2	0.6	1.1	0.6	59
The Netherlands	2.3	2.1	0.0	0.0	1.5	2.0	0.8	78

Source: LEI; adaptation RIVMTable C.3 Hot spots of intensive animal husbandry: livestock population, utilised agricultural area (UUA) and animal density in the EU-15 in 1993, under limitation scenario B: limit animal density to 1.5 LU/ha on farm level

Country/region	cattle	Sheep goats	pigs	poultry	livestock	UAA	Animal density	Manure disposal
		Million animals			Million LU	Million ha	LU/ha	kgN/ha
Belgium	2.5	0.2	0.1	0.3	1.6	1.3	1.2	123
Denmark	2.0	0.2	0.8	1.4	1.6	2.7	0.6	53
Niedersachsen	2.8	0.3	0.8	4.3	2.2	2.7	0.8	75
Nordrhein Westfalen	1.6	0.2	0.6	1.1	1.3	1.6	0.8	73
Galicia	0.7	0.3	0.0	0.4	0.5	0.6	0.8	78
Cataluna	0.3	0.9	0.0	0.0	0.3	1.1	0.2	22
Loire	2.7	0.4	0.0	1.1	1.8	2.2	0.8	80
Bretagne	2.2	0.2	0.1	1.0	1.5	1.8	0.8	82
Lombardy	1.3	0.2	0.1	0.4	0.8	1.1	0.7	76
The Netherlands	3.0	2.1	0.0	0.0	2.1	2.0	1.1	104

Source: LEI; adaptation RIVM

Table C.4 Hot spots of intensive animal husbandry: livestock population, utilised agricultural area (UUA) and animal density in the EU-15 in 1993, under limitation scenario C: limit animal density to 1.0 LU/ha on regional level

Country/region	cattle	Sheep goats	pigs	poultry	livestock	UAA	Animal density	Manure disposal
		Million animals			Million LU	Million ha	LU/ha	kgN/ha
Belgium	2.5	0.2	0.0	0.0	1.3	1.3	1.0	114
Denmark	2.2	0.2	4.4	7.5	2.7	2.7	1.0	75
Niedersachsen	3.0	0.3	2.1	11.3	2.7	2.7	1.0	87
Nordrhein Westfalen	1.8	0.2	1.3	2.3	1.6	1.6	1.0	86
Galicia	0.9	0.3	0.0	0.0	0.6	0.6	1.0	100
Cataluna	0.6	0.9	1.8	15.1	1.1	1.1	1.0	68
Loire	2.8	0.4	0.5	19.3	2.2	2.2	1.0	93
Bretagne	2.4	0.2	0.5	7.0	1.8	1.8	1.0	94
Lombardy	1.7	0.2	0.0	0.0	1.1	1.1	1.0	102
The Netherlands	3.2	2.1	0.0	0.0	2.0	2.0	1.0	104

Source: LEI; adaptation RIVM

Table C.5 Hot spots of intensive animal husbandry: livestock population, utilised agricultural area (UUA) and animal density in the EU-15 in 1993, under limitation scenario D: limit animal density to 1.5 LU/ha on region level

Country/region	*cattle*	*Sheep goats*	*pigs*	*poultry*	*livestock*	*UAA*	*Animal density*	*Manure disposal*
	Million animals				Million LU	Million ha	LU/ha	kgN/ha
Belgium	3.1	0.2	0.0	0.0	2.0	1.3	1.5	154
Denmark	2.2	0.2	9.3	15.8	4.0	2.7	1.5	95
Niedersachsen	3.0	0.3	6.3	34.4	4.1	2.7	1.5	109
Nordrhein Westfalen	1.8	0.2	4.4	7.4	2.4	1.6	1.5	108
Galicia	0.9	0.3	0.6	7.6	0.9	0.6	1.4	122
Cataluna	0.6	0.9	3.3	28.3	1.7	1.1	1.5	91
Loire	2.8	0.4	1.5	56.7	2.9	2.2	1.3	110
Bretagne	2.4	0.2	2.8	36.1	2.7	1.8	1.5	118
Lombardy	1.9	0.2	1.1	7.4	1.7	1.1	1.5	132
The Netherlands	4.2	2.1	0.0	0.0	3.0	2.0	1.5	148

Source: LEI; adaptation RIVM

In table C.6 the costs to transport animal manure in/from the hot spots is presented for the actual situation (1993) and for the four restrictive policy options.

Table C.6 Costs for transport of animal manure in the hot spots of intensive animal husbandry: Actual situation (1993), under limitation scenario A, B, C and D

Country/region	*Animal manure transportation costs (M€ per year)*				
	Actual situation (1993)	Scenario A	Scenario B	Scenario C	Scenario D
Belgium	95,3	3,4	5,6	4,8	7,7
Denmark	47,4	3,3	5,5	18,0	37,1
Niedersachsen	37,6	4,7	8,3	14,0	33,2
Nordrhein Westfalen	25,4	3,0	4,8	7,7	19,2
Galicia	4,6	0,9	1,3	1,8	4,6
Cataluna	19,2	0,2	0,3	6,7	13,1
Loire	15,2	3,8	5,0	8,4	15,2
Bretagne	47,9	3,3	4,2	6,6	17,0
Lombardy	17,0	1,6	2,3	3,2	8,6
The Netherlands	254,2	4,1	6,4	6,5	11,4

Source: TME, 1999

Based on table C.6 the following can be stated about the animal manure transport costs in the hot spot regions in 1993 (actual situation) and under the four scenario's:

- The more animals are allowed in the hot spot regions, the more animals will be produced resulting in higher animal manure transport costs;
- Under the scenario's A-D, the manure disposals (kg N/hectare) are lower than the,. according to the Nitrate- Directive, permitted amount of 170 kgN/hectare. Therefore only intra-regional transport costs are taken into account. In the actual situation (see table C1), the manure disposals in Belgium, Bretagne and the Netherlands are higher than the permitted 170 kg N/hectare resulting in both intra-regional and interregional transport of animal manure.

The intra- and interregional transport costs of animal manure have been determined by taking the following steps:

Step 1 Amounts of animal manure (Pigs, poultry, cattle) per region

The following data have been combined to determine the total amount of N-manure per region:

- Number of animals (cattle, goats, pigs, poultry and sheep) per region. Numbers of animals are presented in table C1-C5;

- Specific animal manure production factors (kg N/animal per year, kg manure/animal per year).

The applied emission factors are presented in table C7

Table C7 Applied emission factors for cattle, pigs, poultry and goats and sheep

	Emission factor (kgN/animal/year)	*Emission factor (gN/kg animal manure)*
dairy cattle	86	5.6
other cattle	59	2.7
Cattle	75	4.3
Sheep, goats	7.5	
Pigs	10.1	6.8
Poultry	0.725	35.6

Step 2 Amount of N-manure allowed per region

The following data have been combined to determine the amount of N-manure allowed per region:

- The Nitrate Directive permits the use of 170 kg N from animal manure per hectare per year;
- Area of agricultural land (hectare). The areas of agricultural land (UAA) for each regions are presented in table C1.

Step 3 Distinction between inter- and intra-regional transport

The following data have been combined to make a distinction between inter- and intra-regional manure transport:

- The results of step 2 are the permitted amounts of N-manure per region;
- The results of step 1 are the produced amounts of N-manure per region;
- The ‘acceptance factor’ tells something about the percentage of agricultural land to be filled up with N from animal manure. Here an acceptance factor of 100% has been assumed.

Step 4 Determination of final amount of animal manure to be transported (‘for money’) intra-regionally

The following data have been combined to determine the final amount of animal manure to be transported intra-regionally:

- The results of step 3 are the amounts of N-manure to be transported intra-regionally. However based on Dutch statistical data on manure transport [CBS, 1995] could be derived that certain amounts of the produced manure is used ‘for free’ at the farm and does not need to be transported for money: 95% of cattle manure, 37,5% of pig manure and 0% of poultry manure is applied at the farm and is excluded for intra-regional transport;
- Conversion factors for cattle, pigs and poultry to convert N-manure into manure. The applied conversion factors are presented in table C7.

Step 5 Determination of the transport distance for intra-regional manure transport

The following data have been combined to determine the transport distance for the amount of animal manure to be transported intra-regionally:

- Distance indicator per region. By considering the region as a circle, the distance indicator (km) has been determined by taking the radius of the circle. In table C8 the areas and distance indicators of the hot spot regions are presented.
- ‘Go round factor’. A ‘go round factor’ of 1.41 (root 2) has been taken because the road between producer and consumer of animal manure is not straight. However the larger the distance indicator, the smaller the ‘go round factor’ becomes. Therefore a linear relation between the ‘distance indicator’ and the ‘go round factor’ has been assumed. The two boundaries are: ‘distance indicator’ > 500 km, ‘go round factor’ =1, and ‘distance indicator’ = 0 km, ‘go round factor’ = 1,41;
- ‘Available space indicator’. The more space there is to transport the manure to in the region, the shorter the transport distance will be. Therefore the ‘available space indicator’ has been introduced,

defined as (produced Ntot – excess Ntot)/permitted Ntot. The value of the 'available space indicator' varies between 0 (no production of animal manure) and 1 (production of animal manure is equal or larger than the permitted amount of animal manure).

Table C8 Surfaces and distance indicator of the hot spots regions in the EU-15

Country/region	*Surface (km^2)*	*Distance indicator (km)*
Belgium	30517	99
Denmark	43080	117
Niedersachsen	47349	123
Nordrhein Westfalen	34068	104
Galicia	29434	97
Cataluna	31930	101
Loire	32082	101
Bretagne	27208	93
Lombardy	23859	87
Netherlands	41007	114

Step 6 Determination of the transport distance for interregional manure transport

The following data have been combined to determine the transport distance for the amount of animal manure to be transported inter-regionally:

- The results of step 5 are the transport distances for intra-regional manure transport. In case animal manure will be transported inter-regionally, the 'available space indicator' from step 5 should be 1;
- 'Extra transport distance indicator'. This indicator is determined by the square root of the ratio between produced and permitted Ntot.

Step 7 Determination of the transport costs of animal manure

The following data have been combined to determine the total costs to transport animal manure intra- and inter-regionally:

- The results of step 3 and 4 are the amounts of animal manure to be transported inter- and intra-regionally respectively;
- The results of step 5 and 6 are the transport distances for intra- and interregional transport respectively;
- Loading costs. The applied loading unit costs are €1.79 per tonne animal manure;
- Transport costs. The applied transport costs are €21.2 per ktonne animal manure/km.

Sensitivity analysis

In the determination of the animal manure transport costs seven steps have to be taken. In all steps uncertainties occur. These uncertainties can be classified into two groups, namely:

- Data uncertainties. In step 1, 2, 4 and 7 (Dutch) data have been used that can be discussed. Examples are the (Dutch) loading and transport unit costs;
- Assumptions. In step 3, 5 and 6 assumptions have been made to determine for example the amount of animal manure to be transported intra-regionally (acceptance factor) and the intra-/interregional transport distance.

Here attention is paid to the 'distance factor' of the Netherlands. Only data are available for the whole of the Netherlands. This results in a relatively large 'distance indicator' and therefore in relatively large costs related to transporting the animal manure (excluding the loading costs). In case data were available at regional instead of national level, the 'distance indicator' and therefore intra-/interregional transport distances would have decreased substantially. In table C9 the overall transport costs for the actual situation in the Netherlands are presented following the two (regional and national) approaches.

Table C9 Costs for transport of animal manure of intensive animal husbandry in the Netherlands in 1993 (actual situation) following the regional and national approach

Country/regions	*Costs for transport ($M€_{97}$/year)*
12 provinces	197
4 regions	211
1 nation	242

Source: TME.

Ergo, computation of manure transport costs based on national data leads to an overestimation of some 20% in relation to summed computations based on provincial data. And, regarding the uncertainties in animal densities due to definition problems, manure N production per animal category, and utilised agricultural area UUA), this model imperfection seems acceptable.

Annex D: Methane Emission Reductions

This Annex assesses the impact of volume and technological measures to restrict the emissions of methane. Methane is an important greenhouse gas. Its potential to contribute to climate change exceeds carbon dioxide by a factor of 21. Hence, policies to restrict the emission of methane contribute to the abatement of the climate change problem. Capture of methane provides a useful source of energy, as methane is equivalent to natural gas, which is considered to be a clean fossil fuel.
Technical measures to restrict methane emission (2010-BL) have been treated in the Technical Report on Climate Change section non-carbon dioxide gases. The volume policy assessed here relates to the severe intensive animal husbandry restriction of 1.0 LU/ha at farm level (see Annex C: Measures to restrict animal husbandry to carrying capacity). This restriction effectively eradicates intensive pigs and poultry husbandry in the EU.
In agriculture methane emissions originate from enteric fermentation and manure. Cattle, sheep, goats, and pigs digesting foodstuffs produce methane by enteric fermentation. Enteric fermentation is the process of anaerobic decomposition of polysaccharides and other feed components in the gut of animals. Methane is a waste product of this fermentation process. This decomposition process continues in animal manure. Manure management – storage or spreading over the agricultural land as a fertiliser - should ensure either aerobic degradation or avoid methane evolution.

The table below shows that a restrictive animal density policy would bring about a further reduction of agricultural methane emissions of 18% equivalent to 52 ktonnes of carbon dioxide.

	scenarios			emission factors		emissions		
	1990 heads 10^6	2010-BL heads 10^6	2010-vol heads 10^6	Enteric fermentation CH_4 kg/head	Manure management CH_4 kg/head	1990 sum ton	2010-BL sum ton	2010-vol Sum ton
dairy	27	26	16	100.0	20.6	3226	3113	1930
other cattle	65	62	62	48.0	6.3	3510	3388	3367
sheep	115	108	108	8.0	0.4	965	903	907
pigs	120	116	15	1.5	7.0	1023	983	128
poultry	951	967	100	0.0	0.1	95	97	10
LU	115	111	65					
total						8819	8483	6341

Annex E: Coastal Zones

Context

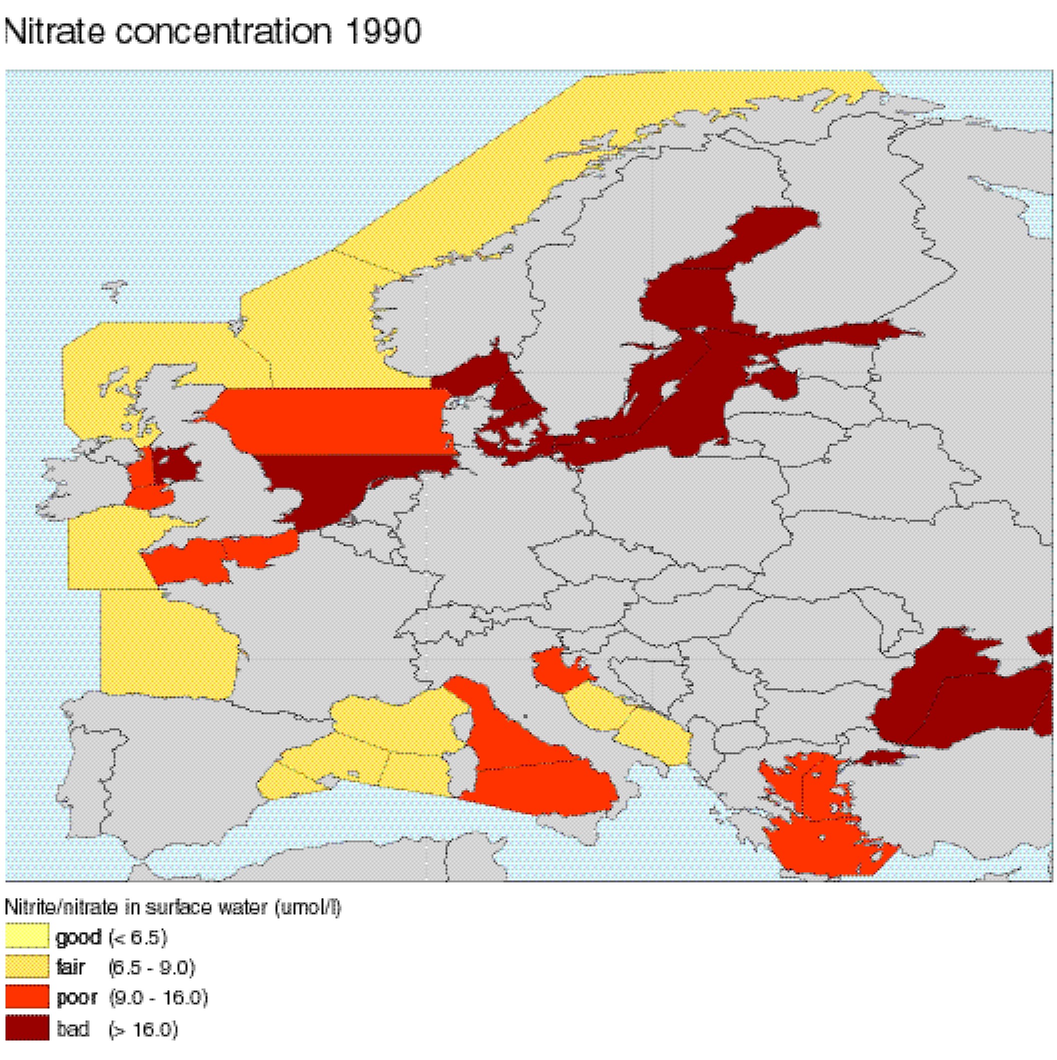
In this assessment, coastal zone issues are only represented through the analysis of nutrient loadings. Other coastal issues, such as urbanization, tourism, and over-fishing are not taken into account.

In many coastal areas, poor water quality and distorted marine ecosystems are attributable to excess nitrogen, particularly along the margins of enclosed seas which have relatively little connection to the open ocean. Water quality is generally well covered by existing EU directives, including the Bathing Water Directive and the UWWTD. However, full implementation has yet to be achieved. The most appropriate indicator to measure the impact on bathing water quality is agricultural nitrate load, which reflects the largest pollution source for coastal seas. However, policy targets for this area have not generally been established.

In the context of this study, there is a close linkage to water management. The biodiversity of marine ecosystems benefits from actions to limit excessive nitrogen releases to surface water.

Assessment and Trends

The agricultural intensification since 1950 has brought about an increase in the amounts of excess nutrients entering the surface waters and the coastal seas. *Figure 3* presents the nitrogen enrichment in coastal zones. The indicator is potential nitrate concentration. The index is labelled *potential* and represents all nutrient inputs to the seas, including air deposition, oceanic influx and exchanges. The index does not account for removal processes in the seas themselves.
In general, there will be little change in the location or degree of enrichment between 1990 and 2010 under the continuation of existing policies (EEA, 1998). This can be considered as a positive result of these policies as the impact of population and welfare growth – both factors amplifying emissions – have been mitigated. The enrichment trend, which started since 1950, has come to a halt. To reverse the trend, additional policies are required.
The Nitrate Directive has some positive impact on coastal zone eutrophication, but has not been designed for coastal zone management. The Nitrate Directive refers to manure and not to artificial fertiliser, which contributes to some 50% of the total agricultural N input. Hence, overall reductions due to the Nitrate Directive are small.
Trends for surface freshwater are provided in Section 2.4.2.


Nitrate concentration 1990
Nitrite/nitrate in surface water (umol/l)
good (< 6.5)
fair (6.5 - 9.0)
poor (9.0 - 16.0)
bad (> 16.0)

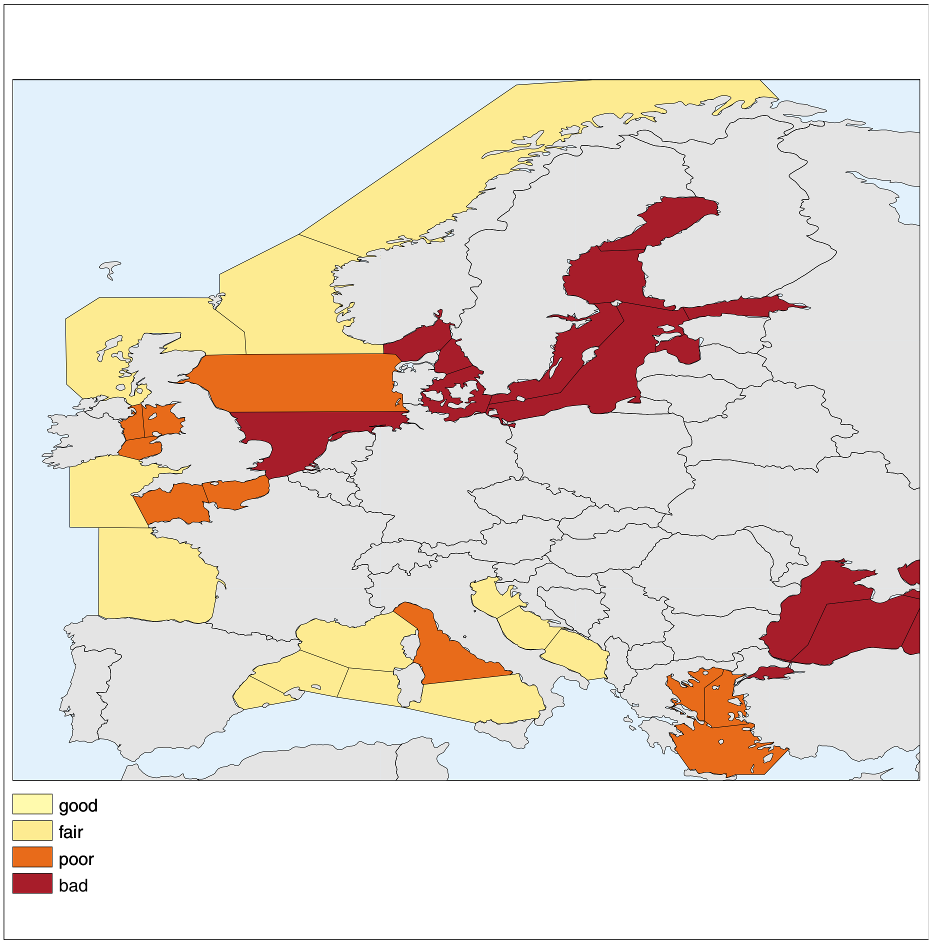


Figure 3. Baseline projection of potential nitrate concentration by 1990 and in 2010. CARMEN model calculations, which account for riverine and atmospheric input, and mixing with neighbouring waters, provide so-called potential nitrate concentration for coastal zones. Due to limited mixing with fresh ocean water, the Baltic Sea and the Black Sea are sensitive to nutrient enrichment. Despite ongoing population and welfare growth, the trend in nutrient enrichment, which started in fifties, seems to be halted.
N.B. No marine denitrification processes have been modelled.

Legend	Quality status	Nitrite/nitrate in surface water (μmol/l)
	Bad	> 16.0
	Poor	9.0 – 16.0
	Fair	6.5 – 9.0
	Good	< 6.5