ECORISK

Ecosystem services valuation for environmental risk and damage assessment

EPA ERTDI STRIVE Programme

Prepared for the Environmental Protection Agency

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Executive Summary

The Environmental Liability Directive (2004/35/EC) applies a common liability approach to instances of environmental damage throughout the European Union. It aims to prevent and remedy environmental damage by holding those responsible financially liable for remediation.

ECORISK has the objective of exploring methods whereby the valuation of social and economic impacts can be used to supplement established methods of environmental damage assessment for the purposes of remediation.

The procedures of the Environmental Liability Directive (ELD) apply to:

- The contamination of land presenting a significant risk to human health
- Impacts to water bodies sufficient to cause a change in quality status as defined by the Water Framework Directive (2000/60/EC)
- Impacts to protected species and natural habitats covered by the Habitats Directive (92/43/EEC) and the Birds Directive (2009/147/EC) sufficient to cause a change in "favourable conservation status".

Where damage has occurred the ELD allows for three types of remediation:

- Primary remediation to restore a damaged resource or impaired service to its baseline condition
- Complementary remediation in cases where primary remediation would fail to fully restore a site to its baseline conditions using primary remediation alone
- Compensatory remediation where there are interim losses until primary or complementary measures take effect.

Complementary remediation could include an improvement to habitat at another site which is geographically linked in terms of species/habitats or human interactions. Compensatory remediation addresses interim losses and aims to compensate for the temporal loss of ecological functions until recovery is achieved. This could involve a prolonged period.

Definitions of complementary and compensatory remediation have been informed by EU Member States' experience of implementing the Habitats Directive (92/43/EEC) and the Birds Directive (2009/147/EC). However, the scope of the ELD is potentially greater in that it applies to impacts to protected species and natural habitats listed by these two directives, but found both within and outside of designated sites. Furthermore, of relevance to the objectives of this report, the ELD acknowledges that as well as species-to-species interactions, losses of ecological functions can impact on social and economic welfare too.

Ecological functions underpin ‘ecosystem services’ of benefit to human beings. These include fundamental ecosystem supporting services, but with benefits realised as regulating services, i.e. services which maintain environmental quality (e.g. waste assimilation), provisioning services (e.g. the supply of natural products such as food) and cultural services (e.g. settings for recreation and various non-use social or cultural values) (MA, 2005; TEEB, 2010; Haines-Young and Potschin, 2013a).

Remediation requires that the operators responsible for an environmental impact provide an equivalent amount of the environmental good and level of functions to that which has been lost. Recognition, and potentially valuation, of ecosystem services losses can complement methods of resource equivalency, specifically habitat equivalency analysis which focuses on providing an equivalent amount of ecosystem processes or functions. The most important requirement is to identify the type and scope of ecosystem services that have been lost or damaged. The next step is to value these services where possible in economic terms.

ECORISK has explored the potential to value ecosystem services in the context of the ELD by using the example of water. Water is one of the three foci of the ELD, but water quality also impacts directly and indirectly of many protected species and natural habitats. The project has identified the principal ecosystem services provided by freshwater, estuarine and inshore coastal water bodies. Estimates of their value were informed by a modest case study of the River Suir and Waterford Harbour.

The project demonstrated that the value of water for abstraction, recreational use and fisheries could potentially be estimated in monetary terms. However, in many cases, data would need to collected locally and, in other instances, is not readily available. The level of direct use can be modest, in particular for rivers, and is mostly represented by angling. Passive use and the value placed on water in the landscape is significant, but more stated preference (i.e. survey-based) studies are needed before value estimates can be reliably transferred to specific cases. The project also found that one of the key ecosystem services provided by water bodies, namely the assimilation of waste, would be difficult to value directly, although its importance could be valued indirectly through the amounts spent on wastewater treatment to maintain water quality and assimilative capacity.

Despite these issues, an identification of ecosystem services can form an important element of the Beneficial Use Index that is being developed by the EPA for the purposes of directing investment to improve water quality in line with the objectives of the WFD. This could provide the rationale for initiatives to collect data to contribute to ecosystem services valuation, enhanced by further primary studies. As the index evolves, it would then be possible to incorporate more quantitative monetary estimates of the value of ecosystem services along with existing physical indicators. An adjustment of values could be needed for the ELD as the directive addresses environmental losses rather than gains.

This process could also inform more formal approaches to resource equivalency. Remediation under the ELD could form one component of a biodiversity offsetting programme that would not be limited to existing sites designated for nature conservation. The valuation of ecosystem services could also contribute to biodiversity banking, an extension to offsetting that offers more flexibility that offsetting applied to pairs of sites using like-for-like remediation. Recognition of ecosystem services would help to ensure that a well managed scheme would deliver "no net loss" as a minimum and, preferably, distinct gains to the wider environment.

In brief, the main findings for the study are listed in the box below:

Overall findings and recommendations

- Remediation should take account of impacts on ecosystem services of value to human beings. It should aim to restore these ecosystem services or compensate for interim losses.
- In some cases the value of these ecosystem services can be quantified in monetary terms.
- Various economic valuation methods are available including cost-based methods, revealed preference and stated preference techniques. As the last of these can be time-consuming, benefit transfer methods are also recommended if the source study has been applied to a similar Irish or UK environment.
- Many ecological functions are not well understood, but often data on distinct environmental changes in outputs (e.g. in fish stocks, bird populations, etc.) is sufficient for environmental valuation.
- When valuing environmental damage, ecologists, the public and specific stakeholders are most likely to value avoidance of dangerous environmental thresholds or tipping points.
- Where monetary quantification is difficult or data unavailable, the scale of these ecosystem services should still be assessed along with the number and identity of recipients. Where ecosystem service losses have occurred in an interim period but cannot be quantified remediation should aim to exceed a no net loss situation.
- Procedures should be put in place to improve the availability of data for local impact assessment, for example data on public and private water abstraction (location, quantity, recipients), data on water and waste water treatment costs, and data on visitor and tourist numbers. Public bodies should be obliged to collect this data and to make it more freely available.
- More primary economic surveys are needed to establish the value that the public places on the quality of freshwater and coastal water bodies and on wildlife and wildlife habitat.

On freshwater bodies:

- Rivers and lakes supply a key ecosystem service in the form of waste assimilation and other service benefits in the form of water supply, angling and various types of recreation.
- A water body's capacity to assimilate waste is strongly related to water quality and is best valued through primary stated preference valuation or benefit transfer. Population is a factor, but it is important to define the extent of the spatial catchment in which values are held.
- Angling and recreation values can be measured through a combination of production function methods and revealed preference, i.e. participation, fishing permit sales, boat hire, travel cost and local expenditure.
- Some local authorities have insufficient data - or insufficiently accessible data - on water abstraction, waste water treatment and respective costs.

On estuarine and inshore coastal water bodies

- Estuaries and coastal areas supply key ecosystem service benefits in the form of waste assimilation, fin fish, shellfish, and recreation, including wildlife related recreation. However, ecosystem services valuation can be challenging because many of the relevant ecosystem functions are still poorly understood.

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1 The Environmental Liability Directive, its application in Ireland and a general introduction to the valuation of ecosystem services

1.1. Introduction

ECORISK has the objective of recommending methods whereby ecosystem services valuation can be used to inform risk assessment and environmental damage assessment in the context of the Environmental Liability Directive (ELD). Chapter 1 one this report reviews the content of the ELD together with its implementation in Ireland. The chapter also introduces the types of remediation, the concept of ecosystem services, various methods of environmental valuation and the approaches used to date.

1.2. The Environmental Liability Directive

1.2.1 Key elements of the ELD

The Environmental Liability Directive (ELD 2004/35/EC) applies a common liability approach to instances of environmental damage throughout the European Union. It aims to prevent and remedy environmental damage by holding those who have caused such damage financially liable for remediation.

The provisions of the Directive are distinct from impacts of civil liability which do not necessarily cover environmental damage. The Directive applies to:

- Protected species and natural habitats covered by the Habitats Directive (92/43/EEC) or Birds Directive (2009/147/EC)
- water bodies as defined by the Water Framework Directive (2000/60/EC)
- contamination of land presenting a significant risk to human health.

Schedule III of the Directive lists specific activities which are considered to present a higher risk to the environment. These include, amongst others, activities requiring licenses, permits authorisations, consents or other instruments permitted under the 1996 Integrated Pollution Prevention and Control (IPPC) Directive[1], various waste management operations, authorised discharges to surface or groundwater, water abstraction and impoundment, the manufacture of dangerous substances, the transportation of dangerous goods, air pollution, the continued or deliberate release of genetically modified organisms (GMOs), transboundary shipment of waste and the management of mining or extractive waste.

Activities that are not listed in Schedule III can be held liable if operators were at fault or negligent but only in relation to damage to protected species and natural habitats listed in the annexes of the Habitats Directive and the Birds Directive. The ELD does not apply to instances of diffuse pollution where it is impossible to identify the polluter, to damage that occurred prior to April 2009, or to events that fall within the scope of international conventions such as pollution of the seas.

The schedules of the ELD determine whether strict or fault based liability should apply.

[1] Shortly to be superseded by the Industrial Emissions Directive.

- Strict liability applies to the operations listed in Schedule III of the Directive for damage to land, water or protected species and natural habitats.
- Fault based liability applies to damage to protected species and natural habitats where an operator is judged to have been at fault or negligent with regard to an activity that is not listed in Schedule III, i.e. mostly agricultural or land use related activities relating to impacts on protected species or natural habitats.

The risk of liability itself acts as a deterrent against those operating with inadequate environmental safeguards. It is intended that this risk of liability will provide an incentive for businesses or organisations to proactively assess the level of environmental risk presented by their operations and to take "preventative actions" where there is an imminent threat of damage (EPA, 2010b)

Amongst the costs of implementation of the ELD that were identified by the Regulatory Impact Assessment (Environ, 2008) were arguments of compliance burden, competitiveness and social exclusion. For these reasons, Member States had the opportunity to choose from a list of discretionary provisions at the time of the Directive's transposition into national law. Member States could choose from:

- extension of the ELD to nationally protected species or natural habitats beyond those covered by the Birds and Habitats Directives,
- 'permit defence', i.e. exemptions where actions had been permitted by a regulatory authority in situations where operators can be argued to have acted according to the best scientific knowledge at the time,
- instances where operations can be argued to have been carried out according to the best scientific knowledge of the time (state-of-the-art defence),
- the restriction of requests for action from third parties to instances of actual damage (rather than imminent threat)
- the listing the spreading of treated sewage sludge from urban waste water treatment plants under Schedule III.

The benefits of these discretionary provisions were identified as being better environmental compliance, prevention and remedial measures, and an expectation of more routine preventative actions by operators, reducing the incidence of remediation costs being realised by state agencies using public funds.

The Directive complements existing environmental protection instruments that are provided by Member State's own environmental protection legislation. A weakness of earlier liability regulations was that they often failed to include an obligation to remediate damage.

1.2.2 The ELD in Ireland

In Ireland, the Environmental Liability Regulations (S.I.547/2008) resulting from EU Directive 2004/35/EC were transposed into law in 2008. Full implementation will apply following the Environmental Liability Bill which is currently proposed for 2013.[2]

The ELD is supported by a guidance document (EPA, 2011a) which explains how operators can proactively assess the environmental risk of their activities. The document describes strict and fault based liability using examples pertinent to Ireland. It explains how the Directive applies a common framework to the prevention and remedy of environmental damage, but also to the imminent threat of damage. Operators whose activities have the potential to cause an imminent threat of environmental damage are expected to apply risk management, specifically to work through a Risk Hierarchy commencing with 1) risk management, followed by proactively responding to 2) an

[2] A draft Environmental Liability Bill was prepared in 2008.

imminent threat of damage, dealing with 3) actual environmental damage, and 4) providing environmental remediation as a final step.

The basic steps are described by the Environmental Management System ISO14001, alternatively known as the Plan-Do-Check-Act (PDCA), which requires operators to minimise any risks identified at the planning stage, to undertake routine checks and to act on risks that are subsequently identified. The Environmental Liability Risk Assessment (ELRA) (EPA, 2006) required for IPPC and waste licences defines this process. Step 1 requires a Screening and Risk Assessment under which the complexity is considered along with environmental sensitivity and the operator's compliance record. For known liabilities, residual management plans have been replaced by the CRAMP procedure involving closure, restoration and an aftercare management plan. For unknown liabilities operations are assessed according to three risk categories which along with an assessment of probability of occurrence and potential severity, determine the range of cost implications. A review of the 2006 ELRA Guidance and CRAMP requirements is currently underway and will be amended to reflect experience to date.

In addition to the requirements of the ELD existing national liability regimes apply to impacts to water and Nutrient Management Plans, to the transport and disposal of waste, IPPC licensing, air pollution, GMOs and damage to nationally designated species and habitats, e.g. in Natural Heritage Areas (NHAs).

1.2.3 Definition of Environmental Damage

The ELD has the objective of preventing damage due to a) land contamination that could give rise to significant health risks, b) the qualitative status and ecological potential of waters, and c) the conservation status of protected species and natural habitats.

1) Land damage

Land damage is any damage that creates a significant risk to human health. Environmental risk assessment is carried out in the form of a Contaminated Land Exposure Assessment (CLEA). This in turn replies on a source-pathway-receptor (SPR) approach in which evidence of all three constitutes an impact under the Directive. Remediation is deemed to have occurred if the SPR chain is broken.

2) Water damage

Damage must be significant enough to affect the quality status of the water body as defined by the WFD. If the impact is less than this it may still be addressed under the Water Pollution Act. Under the WFD, the water quality status of surface bodies is determined through three criteria, namely ecological status, ecological potential and chemical status.

- Ecological status is the biological status or structure and functioning of the aquatic ecosystem, depending on the presence of specific pollutants and on general physico-chemical conditions including oxygen, nutrients, transparency, temperature, acidity, salinity.
- Ecological potential is applicable to modified or artificial waterbodies and describes the ecosystem's capacity to achieve its potential ecological quality.
- Chemical status refers to the presence of priority (polluting) substances.

Ecological status and ecological potential are scored on a five point scale from Bad to High, while Chemical status can be either Good or Failure to Achieve. Where damage has occurred a screening assessment is undertaken to determine if the damage falls within Schedule 3 of the regulations and whether it is significant. (nb. more detail on water quality will follow in WP2).

3) Damage to protected species and natural habitats

Protected species and natural habitats as defined by the ELD and as transposed into Irish law include Birds Directive Annex I species, other regularly occurring migratory species and their habitats; and the Habitats Directive – Annex I habitats (see Appendix 2 of this report), Annex II(b) species and their habitats, and Annex IV species and their breeding sites and resting places (wherever they occur).

Operators are required not to cause environmental damage that would undermine the achievement or maintenance of "favourable conservation status" of protected species or natural habitats within or outside of Natura 2000 sites The potential spatial scope of the ELD is therefore wider than for the Habitats Directive in that it applies to species and habitats protected by the Birds or Habitats Directives wherever they occur and not just within the confines of a designated sites. However, Ireland chose not to include, species and habitats that are designated at a national level from among the aforementioned list of discretionary measures (although impacts may still be covered by national legislation, namely the Wildlife Acts, 1976-2010 and 2011 Habitats Regulations).

Specific detail on favourable conservation status is provided in the 2011 Habitats Regulations. For protected *species* conservation status is favourable if:

- the range of the species is not reduced (and the range is stable or increasing),
- that the population dynamics permit the species to maintain itself on a long-term basis
- that there is, and will probably continue to be, a sufficiently large habitat for the species to maintain its population

For natural *habitats*, conservation status is favourable if

- the area is stable or increasing
- structure and functions necessary for long-term maintenance are likely to continue to exist
- conservation status of typical species is favourable.

In addition to the summarised criteria above, favourable conservation status is acknowledged to require the maintenance of "ecological processes" (structures and functions). Examples of threats are provided by the EC (1996) and include damage such as that from drainage or habitat fragmentation inside or outside of designated sites.

Schedule 1 of the ELD defines the species and habitats data needed to determine baseline condition and the criteria needed to determine the significance of an impact. For many species and habitats in Ireland, one challenge to the legislation is that detailed assessments of baseline condition are always not available for Natura 2000 sites. The NPWS (2008) provides information on the status of protected species and natural habitats. [3] As conservation status is already deemed unfavourable in many cases further losses may be significant by default. There is an interaction with water quality in that the favourable status of many Natura sites depends on the quality of connecting surface or groundwaters.

1.2.4 Liability measures and responses

Each year the EPA publishes a report on prosecutions in which typically around a dozen instances are described. Most of these are related to pollution from poultry or pig farms, from meat or dairy processing plants, and from waste collection facilities. Amongst the more significant incidents of

[3] The first round of Article 17 reports on conservation status was in 2007 and the second round is due this year.

recent years was the illegal dumping of hazardous waste at Coolamandra in County Wicklow and a fire at a landfill at Kerdiffstown near Naas in which local water bodies were threatened with pollution from leachate (SKM Enviros, 2010)

The ELD contains no requirement for mandatory insurance. However, some Member States, e.g. France and Spain, have required insurance or for companies to make payments into a collective clean-up scheme. All member States are called upon to promote the uptake of such protection. There have been examples of the use of bank guarantees and bonds or other market based instruments. Insurance companies themselves have been gradually introducing policies such as Environmental Impairment Liability or extending the cover available particularly in relation to potential risks to health. However, they have been more reluctant to provide protection against instances of gradual damage (although there is at least one such policy) or for habitat or compensatory remediation due to the difficulty in quantifying risk and potential losses (EC 2010). As of 2008, there were no products to address the risk from the release of GMOs. By comparison, in the United States where the Company Environmental Response, Compensation and Liability Act was passed in 1980, environmental insurance has matured into an industrial sector that is now worth over $2 billion, although such insurance is still mostly availed of by larger business (Bio Intelligence Service, 2008).

1.2.5 Remediation

Types of remediation

The ELD allows for three types of remediation:

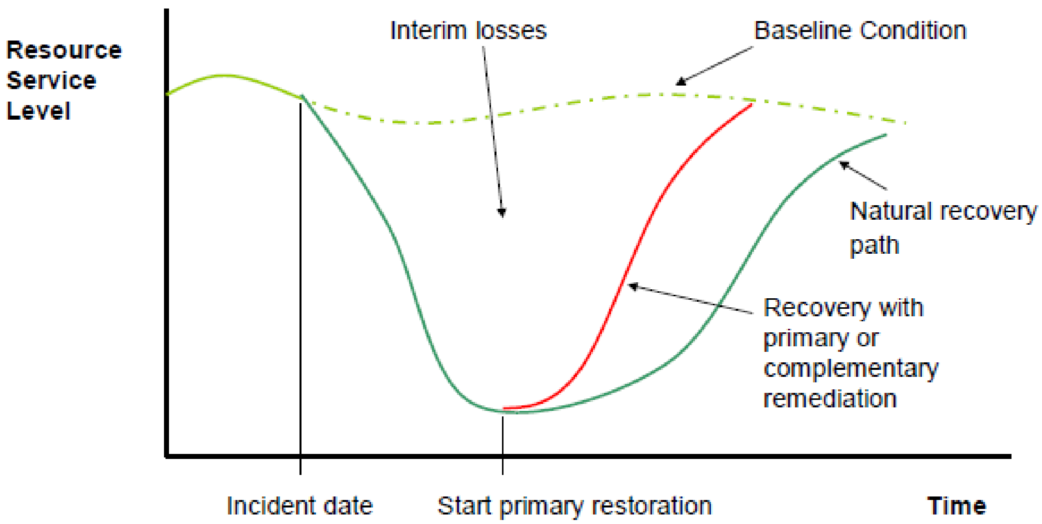
- Primary remediation where it is required to restore a damaged resource or impaired service to its baseline condition

- Complementary remediation which involves additional enhancement in cases where primary remediation would fail to fully restore a site to its baseline condition. This could include habitat enhancement at another site which is geographically linked in terms of species or habitats.

- Compensatory remediation for instances where there are interim losses until primary or complementary measures take effect. This includes temporal loss of ecological functions. Compensatory remediation could involve additional improvement to the site or another site. As defined by the ELD compensation consists of improvements to a natural habitat at the impacted site or the alternative site, or to reimbursement of a government agency's restoration or administrative costs. This report, however, examines compensation with respect to the value of losses of services provided by natural resources where physical remediation is prolonged or not possible.

The nature of complementary and compensatory measures has been informed by EU Member States' experience of implementing the Habitat Directive. Guidance provided by the Commission (2007) on Article 6(4) of the Directive describes both measures under the generic heading of "compensatory measures". Rather, the Directive refers to "extra compensation" for interim losses, but provides no definition of what these are. Both types of measures involve an "operator" identifying remediation options and "scaling" the level of remediation to compensate for the loss of environmental resources over time. Compensatory measures are independent and additional to any mitigation required for a project and are intended to offset certain negative consequences while restoration is undertaken to return a site to the reference biological integrity that justified its designation. In practice, where planned impacts are unavoidable or where an adverse environmental impact has occurred for which restoration is not feasible, complementary remediation has involved habitat restoration, enhancement or creation elsewhere (Mayes, 2008). Guidance supplied by the UK Department for Environment, Food and Rural Affairs (Defra, 2012) refers to measures decided on a case-by-case basis with environmental protection agencies agreeing on the level of remediation that is required. Clearly, if interim losses are substantial, compensatory remediation on site could be difficult. Many habitats, such as peatland, would have evolved over hundreds or even thousands of years and a full restoration in any meaningful time-frame could be impossible. Compensating for long-term or major interim losses could present "disproportionate costs" as acknowledged by the ELD (although the directive does not define what these are). Figure 1 provides an illustration of how significant interim losses can be.

Figure 1.1 Interim losses (source REMEDE Lipton et al., 2009).



Under case law of the European Court of Justice (CJEU) sites classified under the Birds Directive are expected to receive an equivalent level of protection to that under the Habitats Directive. The Habitats Directive requires operators to bear the cost of any impacts and restoration or reinstatement in line with the Polluter Pays Principle.

Operators who cause adverse environmental impacts to protected species or natural habitats as these are defined by the Habitats Directive and Birds Directive are expected to bear the cost of any remediation in line with the Polluter Pays Principle. In the case of the ELD impacts would have been unplanned.

The EPA Guidelines on the ELD provide a detailed description of primary, complementary and compensatory remediation as they apply to Ireland. These note that primary remediation can include natural recovery supported by monitoring if this will provide the best route to recovery. Limited remediation may be all that is required if the ecosystem is capable to re-establishing itself with some support.

For complementary remediation, the Guidelines emphasise the need for a physical link to the damaged habitat. Compensatory remediation is described as involving physical resources in response to interim losses. Although the ELD excludes financial compensation to members of the public, it allows for financial compensation where in-kind approaches are not possible. Interim losses are scaled to compensate for the resources that have been lost over time, but need not be restricted to an adjacent site. The use of equivalence analysis for either complementary or compensatory remediation is discussed below.

Resource Equivalency

For damage to water, habitats or species, the ELD adopts a *resource equivalency analysis* (REA) to estimate the amount of remediation or restoration required through the provision of an equivalent amount of the environmental resource. Resource equivalency has particular relevance to this report in that it goes beyond physical restoration or replacement by acknowledging both the importance of environmental functions (or services between species) *and* services to human beings, i.e. environmental benefits or 'ecosystem services' (described in more detail below). The Directive notes that where restoration of these services "is not possible, alternative valuation techniques shall be used". This distinction is important in that in principle it permits resource equivalency to be pursued through a hierarchy of *resource-to-resource*, *service-to-service* or *value-to-value* approaches. The last of these requires an economic quantification of services.

Where restoration of complementary remediation is considered the emphasis is placed on providing equivalent levels of ecological functions. The resource-to-resource approach aims to identify ecological functions, although for practical reasons straightforward counts of species may be used. The service-to-service approach is often described under the heading of *habitat equivalency analysis* (HEA) in that the objective is to scale the remediation to account for the loss of ecosystem type, quantity or quality. The calculation is made in physical terms, for example where two unit areas of a newly created wetland are assessed to produce the same level of natural resource functions or species-to-species services as the damaged wetland. Both approaches have a requirement for adequate data and a common metric.

A value-to-value approach becomes a possibility through the alternative metric of a monetary valuation of the ecosystem services element. If the monetary valuation of ecosystem services cannot be performed within a satisfactory timetable, a value-to-cost approach may be implemented where the cost of remedial measures is made equivalent to an estimate of the value of the lost natural resource/services.

The regulations require that remediation is based on the following criteria:

- Effect on human health and safety
- Cost
- Likelihood of success
- Prevention of future or collateral damage
- The extent of the benefit to the natural resource or service
- Social, economic and cultural concerns
- Length of time for restoration to be effective
- Geographical linkage to the damaged site.

Compensatory Measures as defined by the Habitats Directive and ELD

In practice, complementary and compensatory measures are often discussed together under the heading of "compensatory measures". Neither is the notion of compensatory remediation specific to the ELD or Habitats Directive. Compensatory measures have, for example, been used within the context of environmental impact assessment where mitigation has either failed or is not possible.[4] The EIA Directive (85/337/EC) defines mitigation as "measures envisaged in order to avoid, reduce and, if possible, remedy significant adverse effects." Commission Guidance on Article 6(4) of the Habitats Directive (EC, 2007) similarly defines mitigation as measures to "avoid, reduce or even cancel" the negative impacts on a site that are likely to arise as a result of the implementation of a plan or project. By comparison, compensatory measures involve replacing lost or adversely impacted environmental goods with others that provide similar functions of equal environmental

[4] Directive 85/337/EC defines mitigation as "measures envisaged in order to avoid, reduce and, if possible, remedy significant adverse effects."

value. Cowell (2000) defines compensatory measures (here taken to include complementary measures) as “the provision of positive environmental measures to correct, balance or otherwise atone for the loss of environmental resources”. If the resources are irreplaceable, then compensation seeks to create new environmental goods of equivalent value and functionality.

Compensation and enhancement appear last in a *mitigation hierarchy* that commences with avoidance of impacts, i.e.

a. Avoidance.
b. Minimisation
c. Restoration
d. Compensation
e. Enhancement

Although this hierarchy is lauded, Rajvanshi (2008) makes the point that mitigation has its own problems and is sometimes regarded sceptically as a afterthought in an impact assessment process whose primary objective is to allow a development to proceed [5] To promote best practice, the European COST Action 341 clarifies the procedures for avoidance, mitigation and compensation, specifically in relation to road and rail projects (IENE, 2011).

Compensation and enhancement appear last in the mitigation hierarchy because priority is given to avoidance at source in line with the precautionary principle which is enshrined within the Maastricht Treaty (amongst other international conventions). Compensation and enhancement emerge much later in the hierarchy as they are thought to embody risks that the compensated or enhanced resource will not be equivalent to that which has been lost.[6]

Typically, areas of new habitat are created to compensate for the loss of other areas. However, the only country with formal rules for compensatory measures (outside of protected areas) is Germany under the Federal Environmental Impacts Compensation Rule. Some other countries allow for compensation within protected areas. COST 341 (ibid) states that under international or national legislation, compensatory measures should address both physical and (ecological) functional aspects whereas compensation of ecological values or financial compensation may be sufficient for areas addressed by national policy.

Compensatory measures have been taken at higher levels. For example, the redevelopment of Cardiff Bay to allow for the regeneration of the city’s waterfront involved the barraging of the city’s two rivers to form a permanent lake. The project was opposed by conservation organisations as it involved the flooding of mudflats used by wading birds. Compensation was provided in the form of new created wetlands to the south of the city.

For the purposes of valuing environmental liability, the protected species and natural habitats element of the ELD applies to all the relevant habitats listed in the Habitats Directive wherever they occur as well as to species protected under the Birds Directive. Article 6(4) of the Habitats Directive is therefore relevant to the interpretation of the scope for compensatory measures under the ELD. The EC guidelines on Managing Natura 2000 Sites (EC, 2000) emphasise that compensatory measures should be “additional” to the normal process of implementing the Directive and only be considered in the context of appropriate assessment associated with a new or amended plan or project. The Habitats Directive concedes a case for compensatory measures only at the point where there is acceptance that a plan or project that is damaging must proceed for imperative reasons of overriding public interest (IROPI) in the absence of reasonable alternatives following an appropriate

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[5] On the basis of his experience with numerous environmental impact assessments this is an argument with which the author confers.
[6] Cross-reference to comments on preceding page.

assessment process. The precise definition of what comprises compensatory measures and their scope is currently being considered by the European Court (Case C-521/12).[7]

Article 6(4) outlines the imperative reasons that can be invoked. The opinion of the Commission must first be sought in situations where a priority habitat would be adversely affected other than for reasons of public health or safety.[8] Although it is open to a country to disagree within this opinion, it must be prepared to defend its standpoint in the European Court. In all Article 6(4) cases, the Member State must inform the Commission of the compensatory measures that will be taken to protect the overall coherence of a Natura 2000 site. In practice, the requirements severely restrict the scope for this where impacts are adverse or long lasting, although (in the UK) instances of successful Article 6(4) have arisen in relation to coastal defence, flood control and some infrastructure developments.

Habitats Directive - Article 6(4)
"If, in spite of a negative assessment of the implications for the site and in the absence of alternative solutions, a plan or project must nevertheless be carried out for imperative reasons of overriding public interest, including those of a social or economic nature, the Member State shall take all compensatory measures necessary to ensure that the overall coherence of Natura 2000 is protected…"

The Habitats Directive is strict, i.e. no adverse effect on the integrity of a European site, i.e. an SAC or SPA covered by Natura 2000, is permitted except in exceptional circumstances. The Directive requires compensation so that the integrity of the Natura 2000 network is maintained with respect to interim losses (as referenced by the ELD) and these should never be equivalent to the total loss of the ecological resource. However, the Commission advice "Assessment of Plans and Projects Significantly Affecting a Natura 2000 Site (EC 2001) takes a pragmatic perspective. In this document "compensatory measures" (for interim losses) can take a variety of forms, i.e. of restoration, creation, enhancement or preservation of habitat, so long as this compensatory habitat has comparable properties and functions to that lost. These compensatory measures should be in place and functioning before original areas are lost. Payments into a conservation fund would not suffice.

Consequently, there is a potential contradiction between the very limited compensation that is allowed for by the Habitats Directive and that which is acknowledged by the ELD. Key distinctions for the ELD are that relevant impacts are not restricted to Natura 2000 sites and are not planned, but rather (by their nature) have already occurred. Furthermore, interim impacts can be significant given the long (sometimes very long) time that it would take habitats to recover or be restored. The potential duration of interim impacts means that compensatory measures must address both ecological functions (i.e. species-to-species) as well as ecosystem services. The latter is a distinction of the ELD which addresses the implications of environmental loss for human beings whereas the Habitats Directive concerns itself only with ecological functions as they impact on a species, habitat or site.

In all cases, the notion of compensatory remediation involves "no net loss" of ecological integrity. Although conditions are not strictly specified in the guidance on Managing Natura 2000 Sites, it is generally assumed that actions should veer towards over-compensation (EC 2007). Under the Habitats Directive, compensation can only be contemplated where it represents re-creation or improvement of sub-standard area or at an external site within the same "biogeographical region",

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[7] CJEU is currently reviewing a case (Briels and others vs. Minister van Infrasructuur) that could help to define where compensatory measures should occur. The deliberations are relevant to the Galway City Outer Bypass, the preferred route of which was rejected on the basis that a small area of priority habitat would have been was impacted and could not be replaced. The relevance of compensatory measures to non-priority habitats (such as the wintering grounds of protected species in the case of the on-going Galway Port development) still has to be demonstrated (www.lexology.com).
[8] Priority habitats are those in danger of extinction at a national level.

migratory route or wintering area and preferably as near to the original site as possible. The site should also match the ecological structures or functions within the existing site or be of a level to justify the new site's designation. Where permitted for purposes of interim losses, like-for-like compensation requires that the new habitat be of at least a comparable size with comparable ecological functions. The DG Environment website provides various examples relevant to Article 6(4).

These requirements are relevant to compensatory measures under the ELD, although the potential application is not limited to ecological functions alone and extends beyond Natura sites. This question of equivalence is critical in that compensatory measures can be defined in terms of either units of resources (REA) or of ecosystem functions and services (HEA). The process is far from straightforward. In practice, an environmental assessment will need to consider such issues as baseline conditions, spatial extent and the damage-recovery trajectory (Lipton et al., 2008).

1.2.6 Ecosystem Services

Definitions and typologies

Terrestrial, aquatic and marine ecosystems have been described as *natural capital* as they are inputs to processes and goods that advance economic welfare along with financial capital and labour inputs. Natural capital has been the subject of many environmental economic valuation studies since the 1960s with the number of such studies having risen rapidly since the mid 1980s, particularly in relation to individual species or environmental goods of amenity value. Increasingly, however, attention has focused on a definition of the types and range of *ecosystem services* that 'flow' from 'stocks' of natural capital and which provide for human well-being and even human survival (CBD, 2006; Fisher and Turner, 2008). A conceptual framework for ecosystem services was introduced by the Millennium Ecosystem Assessment (MA, 2005) which categorised these flows into supporting, regulating, provisioning and cultural services.

Of these services, *supporting services* underpin other ecosystem services and include such primary ecological functions as soil formation or the role of deep sea reefs in the marine food chain. *Regulating services* are those which maintain the quality of the environment and include such benefits as the contribution of aquatic biodiversity to water quality and waste assimilation or a complexity of ecological contributions to human health such as the natural control of pathogens. *Provisioning services* are readily understandable as the supply of fish, crops, timber or other renewable materials. *Cultural services* include the more habitual targets of environmental valuation such as direct or indirect benefits in the form of amenity and recreation as well as certain non-use goods that are valued for their pure existence or which are perceived to contribute to quality of life. Provisioning and cultural ecosystem services both involve distinct interactions with human beings. Recent definition have refined cultural ecosystem services to be the "physical settings, locations and situations" (Haines-Young and Potschin, 2013b) provided by natural functions that contribute to human well-being

The MA typology is not the only available. In a preceding paper by De Groot et al (2002) ecosystem functions are identified as the natural processes and components that provide services to human beings. On this basis, they identify regulatory functions (e.g. gas transformations, climate regulation, disturbance prevention, water supply, water regulation, soil formation, soil retention, nutrient cycles, bio-control, etc), habitat functions (ecosystems as habitats for living space, refuge and nurseries), production functions (including functions equivalent to the supporting services, but extending to provisioning services as referred to by the MA) and information functions (reference points for human wellbeing). The first two of these four function groups are described as being conditional to the availability of the other two. From the De Groot et al perspective, functions become reconceptualised as ecosystem services when human valued are attached. They add that the physical scale of ecosystem functions need not necessarily correspond with the scale at which ecosystem services are valued by human beings.

The definitions offered by The Economics of Ecosystems and Biodiversity (TEEB, 2010) also identify habitat services rather than supporting services as a means to avoid double counting by distinguishing these from ecosystem functions that (as with de Groot et al) "underpin the capacity of an ecosystem to provide goods and services". A recent definition proffered by CICES (Haines-Young and Potschin, 2013b) on behalf of the European Environmental Agency retains supporting services but emphasises the need to concentrate on final ecosystem services for the purposes of environmental accounting.[9] The report distinguishes final ecosystem services that in turn provide goods and services to human beings. It also identifies cultural services as being physical setting for recreational activity and for cultural values. The concept of settings is also adopted by the UK National Ecosystem Assessment (NEA, 2011). The treatment of cultural services corresponds to the notion that the environment is both a natural (biotic and abiotic) and cultural phenomenon in which symbolic meanings and cultural values are attached to wild and semi-natural landscapes.

The value of ecosystem services

Ecosystem services are needed - often consciously demanded - by human beings and will have an economic value that is most apparent when demand exceeds supply. For some services, including many provisioning services, this value is realised, at least partially, through the market place. However, the value of many other supporting, regulating and cultural services is not usually realised through such transactions. These are *non-market goods* that are not traded in any marketplace and where the absence of price signals fail to communicate possible scarcity or to signal vulnerability.

Many non-market goods fall into the category of public goods for which there are no property rights and for which use may be "non-excludable" in that others cannot be excluded from consumption. One important cause of market failure is information failure arising from the public good characteristics. Frequently, the value of non-market environmental goods, along with the ecosystem services that contribute to them, is not appreciated or cannot be assigned to any single organism. Often this value is poorly understood by science let alone by resource managers. Even amongst marketed agricultural products the role of underlying ecological processes can be poorly understood, for example the role of soil microbes. Consequently, there is no guarantee that land managers will take account of these processes and protect them even where this is in their best interests. Indeed, the absence of information from which to derive values or by which to chart scarcity can result in the unsustainable use of resources. This misuse can extend to the trade offs that are made between different ecosystem services. For example, natural forests which are of value for regulating services such as the maintenance of water quality, flood moderation or erosion control, may be cleared to provide land for provisioning purposes for crops or commercial forestry that are of reduced or no regulating service value.

Market failure also extends to temporal and spatial discrpencies. The uncertainty that characterises the interactions between ecological functions and their relationship with human beings applies especially over longer time frames. The implications of ecosystem loss are often not appreciated in the short term and flows of longer term benefits may be discounted by standard accounting procedures. Uncertainties also apply at different spatial and social scales in that ecosystem losses at one location can have implications for other locations or for other population subsets (Carpenter et al., 2009). These uncertainties often provide tacit justification for policy inertia. A very pertinent example is the role of forests and peatlands in carbon sequestration where the value of the regulating service will only be realised in the decades to come and where a spatial and temproal mismatch exists both now and in the future between the recipients of the associated private benefits and social costs. For this reason some environmental scientists, e.g. Grecham Daly, identify a further ecosystem service in the form of an *option value* (an accepted element of Total Economic Value) favouring the preservation of ecosystems until such time that their true value is realised.

[9] With a view to the System of Economic and Environmental Accounts (SEEA) being developed under the European Biodiversity Strategy.

In the same manner that the functions which support many ecosystem services are not fully understood, there is also a mismatch between how the concepts of natural capital and ecosystem services are understood by environmental scientists. This disparity extends to the practical application of this knowledge in policy and decision-making. Just as in the examples of above, some policies manage ecosystems from the point of view of natural resources, i.e. the management of forests as a timber resource, rather than for the full range of ecosystem services that they provide. At the opposite end of this spectrum, conservation policy began with the protection of threatened landscapes or species often without factoring in well-established interactions with human activity, including the social benefits that can provide for future protection. Conservation policy now recognises the dependence of target landscapes and species on the underlying ecological and social web that includes a wider concept of environmental protection extending to biomes and their full array of species. Increasingly it is understood that sites or species cannot be protected in isolation from the surrounding environment or an awareness of the needs of human beings. Nevertheless, much conservation and ecological research continues to address the consequences of anthropogenic impacts on biodiversity rather than of the benefits of biodiversity on ecosystem services and humans. This presents a challenge for ecosystem services valuation.

Information on the *marginal* value of changes in ecosystem services is of most relevance to decisions over natural resources or natural capital. Economic analysis is most productive where information on marginal change is available. Ideally, to understand the value of ecosystem services, economists need information on the additional contribution of each unit of ecosystem service at any one time. This informs us of how ecosystem services are affected by changing conditions and how they can be traded-off against other social priorities. It is in this regard that the deficiency of scientific information is most often revealed because we may not understand the marginal contribution of ecosystem functions and whether these would be vulnerable to the loss of keystone species.[i] In practice, environmental economics can usually tolerate information on discrete changes. Most important is an understanding of where the critical ecological thresholds are to be found. Awareness of these thresholds is pertinent to an assessment of liability. However, ecological thresholds can be difficult to predict in that they are frequently located outside of previous experience or can be determined by local conditions.

The relationship between biodiversity and ecosystem services

According to de Groot et al (ibid), ecosystem functions are the processes and components that provide for ecosystem services. Functional diversity has been described as the range of species' organisational traits that influence ecosystem properties in similar ways (after Tilman (2001)). While there appears to be a correlation between low levels of biodiversity and low levels of functional diversity (Hooper et al., 2002), a high level of biodiversity is not necessarily indicative of a wide variety of ecosystem processes or functions, let alone ecosystem services. There is generally a positive relationship, but the ecosystem contains a varying role for species redundancy and keystone species as well as a sizeable degree of context dependency (Naeem et al., 2002).

Redundancy occurs when the processes performed by one particular species can be replaced by another in the event of the first becoming more scarce or of the environment having changed. Although redundancy sounds like a negative term, it is this characteristic which provides for ecosystem adaptability and resilience. Keystone species, on the other hand, have a vital role to play in maintaining a particular ecosystem or ecosystem state. An identification of impacts on keystone species is therefore especially important to any evaluation of ecosystem service impacts and, by extension, to questions of liability..

Critical natural capital is a term often attached to certain important types of habitats or collections of species, for example Ireland's remaining raised bogs, where these have evolved over centuries. These are defined as priority habitats and are protected in that they provide a core reserve of biodiversity in the event of environmental change or losses elsewhere. The precautionary principle implies that we should protect the best examples of natural ecosystem given our uncertain knowledge of ecosystem processes. These habitats may contain certain keystone species that perform, or are thought to perform, critical processes, the loss of which could undermine the stability of the ecosystem or of wider environment along with the sustainability of human activities that depend on it.

Rather than just biodiversity, it has been suggested that we should be looking to protect the diversity of ecological functions (Perrings et al., 2010). There is evidently a need to raise the resilience of natural systems and the ecosystem functions they perform, but also to ensure the sustainability of interactions between natural systems and human beings. However, only recently have ecologists begun to pay attention to ecological functions (Mooney, 2002). Our understanding of functional diversity is in its infancy and we have little information on the ecosystem processes that occur between trophic levels or at a the landscape levels (Mooney, 2002). Different scenarios of biodiversity loss affect functional diversity in different ways depending on what else is present (Solan et al., 2004). Not all ecological functions are of value as ecosystem services, but we might expect that the more functions there are the more likely there will be ecosystem service benefits. An ecosystem function could contribute to more than one ecosystem service (e.g. waste assimilation to drinking water and water-based recreation), but also various functions could contribute to a single ecosystem service (e.g. functions at work in salt marsh, mudflats or dunes and benefits to coastal hazard reduction). There are also ecological functions that have an *option value* in that they could be valued in the future.

The existence of critical trade-offs and thresholds has long been known to ecologists (Ostrom, 2009), but the fundamental ecological mechanisms behind these are less well understood (Bennett et al., 2009; Nicholson et al., 2009). The protection of priority habitats may not be enough to maintain species as favourable concservation levels and an interconencted spatial network consisting of a range of habitats, not exclsuive to priority habiats, may be needed. In resposne to the threat of climate change. modern portfolio theory has also been applied to conservation choicies (van Tefferlen and Moilanen, 2008; Ando and Mallory, 2012) to minimise the biological "portfolio's" vulnerability to exogenous change.

1.2.7 Economic valuation

Methods

The economic valuation of ecosystem services is warranted in that these services have an impact on the welfare of human beings. Remediation should seek to restore ecological processes and functions even in the context of complementary remediation. In doing so, it will restore ecosystem services, but interim losses will also have occurred. Monetary methods can be used to value final ecosystem services, be these regulating, provisioning or cultural services, Where they can be applied, these methods have the merit of expressing benefits in a common metric than can be compared between ecosystems and regions. A monetary metric can also be readily understood, including by policy makers, and can be compared with policy costs in a cost-benefit analysis, for example for environmental protection. In the context of the ELD, economic valuation is especially pertinent to the aforementioned value-to-value equivalency, although it has relevance to other resource equivalency approaches too.

Four principal approaches to the economic valuation of environmental goods are:

a) Productivity or production function methods
b) Replacement cost, avertive expenditure and avoided cost
c) Revealed preference
d) Stated preference.

a) Productivity or production function methods

Productivity or production function methods can be applied where market price data is available. For instance, a change in productivity approach uses market prices to demonstrate the additional resource cost needed to achieve the same output as would have occurred prior to a loss. Market prices can also be used within a production function approach to identify from the price of a marketed good, the indirect value of an ecosystem service if information is available on its contribution to the final product.

For example, timber has a market value that reimburses human and capital inputs such as labour, machinery and fertiliser. If allowance is made the value of these inputs during the growing phase, the standing value of trees can broadly be used to represent the value of the underlying environmental good. The production function approach therefore captures the net factor income or the specific contribution of the unpriced ecosystem service.

b) Damage avoided, replacement cost and avertive expenditure

Where the price data on the ecosystem service good is not available, *damage avoided* provides a measure of the benefit of an ecosystem service. *Replacement cost* or *avertive (defensive) expenditure* approaches may be used. The approaches attempt to reflect the benefit of environmental good by estimating the cost of the alternative or artificial mechanisms that would be needed to perform the same service. For example, where a beach and dune system prevents erosion, replacement cost could involve the construction of artificial defences such as sea walls. These might forestall the rate of inward erosion for a while, but could also have an impact on the attractiveness of the beach for leisure use. Avertive expenditure could include the cost of a strengthening of the dune system, for example by reseeding with marram grass. These costs in fact represent benefits in terms of costs avoided. These benefits could be extended to include the damage from flooding in the event that the dune system is breached or the loss of recreation value (for which see below).

c) Revealed preference

Revealed and stated preference methods apply particularly to direct benefits including those due to many provisioning and cultural ecosystem services. These methods respectively involve observations of human behaviour or expressions of willingness-to-pay/willingness-to-accept obtained from questionnaire based surveys. *Revealed preference* includes the example of the travel cost method (TCM) where data is collected on travel expenditure and/or journey time to determine the value that visitors place on a particular natural destination. Hedonic pricing is another example of revealed preference in which the value of natural assets is identified through the econometric analysis of property prices.

d) Stated preference

All valuation methods essentially depend on people's willingness-to-pay for a resource or service. Survey based *stated preference* asks people directly (in indirectly) to express this willingness-to-pay amount. The contingent valuation method (CVM) and discrete choice experiments (DCE) can be used to demonstrate people's willingness-to-pay to protect an environmental good.

CVM elicits direct expressions of value through a willingness-to-pay question. This can be an open-ended question, although (arguably) more accurate values can be obtained from closed-ended, or dichotomous questions in which different respondents are asked if they would be willing to pay a particular amount. This amount may be succeeded by a follow-up question with a higher or lower sum depending on the response to the preceding question. The aggregate responses can be used to construct a bid curve and to demonstrate the compensating variation value of a change in an environmental good or ecosystem service (for an improvement in the good).

In the DCE method, willingness-to-pay is derived indirectly by asking respondents to trade off the ecosystem service attributes against other attributes including a surrogate price. Respondents are presented with a number of choice sets comprised of the same attributes for two or more alternatives, but where the attribute levels are varied by means of an underlying statistical (factorial) design (or variations on this). The trade-offs permit estimates of the influence of an attribute level on the probability of a choice, while the payment attribute allows this probability to be expressed in terms of a monetary amount.

Stated preference methods have the virtue of capturing a larger proportion of the consumer surplus above that which would serve as the equilibrium price in the marketplace. However, the manner in which the survey and analysis is conducted is critical, in particular the context in which people are asked for their willingness-to-pay or their willingness to trade off environmental attributes. A respondent who does not accept trades-offs between income and environmental attributes or these attributes and a price attribute is said to have lexicographic preferences for a single output or combination of attributes.

All valuation methods must be applied with transparency and thoroughness. Many ecosystem services are realised as external benefits from one sector to another. As discussed above, forestry is most conventionally valued for its provisioning service of timber output, but also provides for various supporting, regulating and cultural services. With so many ecosystem functions providing multiple services, care must be taken to avoid double counting the benefits. There will also be instances where economic valuation is impractical. On such occasions, it may be possible to apply other strategies to demonstrate the scale of values, their significance and where they accrue. Qualitative techniques could be used or semi-quantitative approaches such as multi-criteria analysis (MCA).

Environmental values including losses and gains

A further issue of relevance to environmental impacts and liability is commonly referenced observation that people appear to value environmental losses more than environmental gains. Many applied studies reveal evidence of a kink in the valuation function. Related to this is evidence of status-quo bias in which people appear to value protection of the status-quo more than enhancements. Although at odds with classic economic theory, both behavioural preferences have been explained in psychology, notably through prospect theory (Kahneman and Tversky, 1979). Ecologists are often sceptical of the capacity of environmental restoration to perfectly replace an ecosystem that has been lost (e.g. (Woodcock et al., 2011)). While their view may be based on sound science, it appears the sentiment is also shared by the population at large. This would support the argument that remediation should go beyond replacement alone as more natural capital will be required to compensate for a loss. Loss aversion has relevance to both complementary and compensatory measures.

There are also issues over the comprehensiveness of values estimated using economic valuation. The question of intrinsic values often emerges as environmental goods are often argued to be of intrinsic or inherent value. Economic valuation only recognises anthropocentric values and not values of environmental goods in and of themselves. However, a dilemma is often revealed in relation to resources with particular or unique characteristics. Sometimes problems emerge from information failures in that the characteristics of some such resources may not be widely understood, for instance by public questioned in stated preference surveys. Additionally, issues arise from the aggregation of individual values in that these aggregate welfare benefits tend to be greater for environmental goods in the vicinity of areas where most people live. This is quite correct in that more people are available to value and visit such areas. However, according to these principles, pristine wilderness areas, or other areas of high ecological quality, are at risk of being undervalued. In fact, they could be valued more by people, including individuals who have rarely or never visited such areas (non-use or existence values), but direct use and numbers normally triumph in applications of economic valuation. In these circumstances, it could be argued that the wilderness has an intrinsic value that cannot be captured by economic methods in isolation.

In addition, economic methods can fail to articulate the full range of social and cultural values. Various ecosystem services are of importance in providing sustainability or security of livelihoods. For example, high crop yields are often achieved through capital intensive methods systems fuelled with fertiliser, herbicides and pesticides. Although ultimately fertiliser can only supplement natural processes, a system with a greater dependence on ecosystem services could appear less efficient and therefore less valuable. Such a system could represent a traditional practice that has communal values or is of cultural significance that cannot be captured by economic valuation methods where these are based exclusively on individual utilitarian benefits (Sagoff, 1994; Kumar and Kumar, 2008; Chan et al., 2012). In addition, the traditional practice could involve less adverse externalities (such as from pesticide contamination) and be more sustainable.

Benefits transfer

Valuation studies, particularly stated preference methods that are reliant on public surveys, are expensive to conduct on a routine basis. *Benefit transfer* (value transfer) is one way of getting around this barrier. Essentially, values derived from studies at one location are transferred to a similar location or similar natural capital asset for which a value estimate is sought. However, this assumes a correspondence between the nature of the two goods, of public preferences and of the socio-economics of the populations that value them. Typically, values are adjusted for relative national income levels and often for relative demographics (Navrud and Ready, 2007). However, such *unit value* transfers are often difficult to justify. Following their assessment, Rosenberger and Stanley (2006) are of the opinion that more reliable benefit transfer estimates are achieved by transferring the valuation function itself to the new site, i.e. *function transfer*. For this to succeed, they find that the valuation function must be comparable across space and time, be stable or vary in a systematic way, and be founded on "correct" primary data. The differences between the environmental goods must also be capable of being described by the estimated price vectors. In practice, the right conditions for benefit transfer are not always present. While there are transfers that are valid and well executed, there are enough less successful studies to undermine users' confidence. However, methods are improving. For example, Hynes et al (2010) have demonstrated the virtues of spatial micro-simulation methods to transfer value functions from one location to another.

Meta-analysis involves regressions of a number of studies to determine the circumstances in which values reliably correspond. In effect, meta-analysis is a more complex form of value function transfer estimated on the basis of multiple studies. In applying meta-analysis, though, it is important to note that studies can vary in their objectives.. Some primary studies seek to express only the benefits of a particular environmental good while others aim to value different management strategies. Some examine only particular aspects of the environmental good, while others seek to approximate its total economic value (e.g. Blomquist and Whitehead, 1998).

In a meta-analysis of wetlands, Brander et al (2006b) find that improvements in water quality are valued most by the public. They also find significant levels of correspondence between studies in relation to the effect of respondent income and of the anticipated relationship between value and wetland size (i.e. declining). Wetlands dominated by direct use values, such as agriculture or hunting, tend to have lower values than those valued for wildlife or amenity, but RAMSAR sites have rather low values which Brander et al postulate may be due to remoteness or limitations on access (implying that access to sites is valued rather highly).[10] Fundamentally, however, their data is not supportive of benefit transfer. A particular problem appeared to be a suspected lack of

[10] Note the comments on wilderness valuation made earlier

correspondence between sites with many studies providing only a crude definition of the environmental good.

Despite these issues, it is possible to test the validity of a benefit transfer exercise against the results of a well executed primary study. Transfer errors typically arise from issues of generalisation, measurement and publication bias (Rosenberger and Stanley, 2006). The first of these arises where the context or policy scenarios is different. Measurement errors arise from deficiencies in the reporting of data or methods in the original study such that, for example, a variable is excluded from discussion, due perhaps to insignificance, but which is of relevance to the benefit transfer exercise. Publication errors are due to the nature of academic publication whereby novel methods or applications are favoured over the consistent approaches needed to assess policy by standard criteria. This issue relates to the motivation for the study. In the academic literature to which most such studies are directed, the authors' interest is often less with the environmental good itself than with testing the virtue of a different survey approach or analysis method for the purpose of peer reviewed publication.

Despite these issues, it is possible to test the validity of a benefit transfer exercise against the results of a well executed primary study. On the whole, one would expect function transfers to out-perform unit transfers. However, there is much room for error, including the inclusion of data from inappropriate or inaccurate studies. Brouwer (2000) found some unit transfer errors in excess of 475%. However, Liu (2007) found that only 2.5% of published studies had transfer errors in excess of 100% and that 40% of studies performed satisfactorily with errors of less than 10%. The Brouwer et al analysis (ibid) found that less than one fifth of studies had a transfer error of less than 10% of value. Johnson and Rosenberger (2010) argue that there are often problems of insufficient commensurability between studies. Generally, the best transferability appears to occur where there is a similarity of site characteristics, population, and market structure.

Applications of valuation to natural capital and ecosystem services

Environmental economists have been applying each of the above valuation methods to environmental resources for over twenty years. Indeed, there is a large body of literature that has relevance to habitats, including various ecosystems that are at more acute risk of environmental damage such as wetlands. A few studies have attempted to value perceptions of biodiversity directly. Nunes and van den Bergh (2001) reviewed 61 studies that addressed biodiversity at one level of another, the majority of which applied CVM or investigated recreation issues. However, the authors found that the studies reviewed lacked a clear definition of biodiversity and failed to address the full range of biodiversity benefits.

To address some of the criticisms of how biodiversity should be described, Christie et al (2006) used focus groups to explore perceptions of biodiversity and followed these with both DCE and CVM. The DCE was used to examine attributes such as familiarity of species, rarity, habitat and ecosystem services. The last of these were not defined except to distinguish between those that benefit human beings and those beneficial to the wider environment. The CVM survey requested a willingness-to-pay for biodiversity enhancements, reporting values of €55 per household per year in increased taxation for enhancements achieved by habitat creation. The authors found that survey respondents were willing to pay for all ecosystem services, although they were especially motivated to do so where these were beneficial to human beings.

It is only in recent years that economists have turned their attention to ecosystem services valuation, mostly following the seminal global value estimate of biodiversity by Costanza et al (1997) and other reports for the Convention of Biological Diversity and the Millennium Ecosystem Assessment. Before the Costanza et al study economists variously referred to ecosystem services as ecosystem functions, ecological characteristics or to specific outputs. The aforementioned paper by De Groot et al (2002) also aimed to remove this inconsistency. Nevertheless, a search of the EVRI valuation database (www.envir.ca) returns 155 references with the term 'ecosystem services' in the title or abstract, although many of these are of only slight relevance. Very few references address environmental liability, although there are many applications of potential relevance such as impacts on water quality.

Over this period of burgeoning environmental economic research, a school of *ecological economics* has also emerged. Ecological economics applies a more transdisciplinary focus to a wider range of ecological values than the more utilitarian values identified by conventional environmental valuation. It can be distinguished from environmental economics by its rejection of the assumption of separability of economic values from ecological relationships. Fromm (2000) identifies a "valuation gap" between services that can potentially be monetised and those which are important in securing ecosystem services. Environmental economics sometimes acknowledges the limitations of valuation by appending sustainability rules or safe-minimum standards to the welfare-based values used in a cost-benefit analysis. However, Fromm warns against including arbitrary margins given the level of complexity and non-substitutability of ecological functions. The argument is analogous to that discussed above of uncertainty relating to ecosystem fragility and resilience and is one that has yet to be resolved. (see also (Straton, 2006).

Ecological economics makes a strong case for rejecting the assumption that natural capital can be substituted by human and financial capital. Similarly, many people are uneasy with the "commodification" of natural resources such as biodiversity. Nevertheless, at a practical level valuation is useful because decisions over resources must be made. The applied valuation of ecosystem services may often incomplete, but does have the merit of ensuring that natural capital is considered and not ignored in decision making. Potentially, there are opportunities to finance biodiversity protection and for payments for ecosystem services that could allow us to meet the objective of biodiversity protection more efficiently and effectively than statutory conservation or conventional command-and-control policies. The notion of "environmental protection being an opportunity rather than a cost"[11] was the dominant theme to emerge from the recent Rio+20 Earth Summit (Pearce, 2012). It is a philosophy that has yet to be tested.

1.2.8 Valuing Compensatory Remediation

Introducing environmental valuation to resource or habitat Equivalency

Increasingly, both the REA and HEA approaches to designing compensatory measures are merging and being applied with reference to an equivalent level of ecosystem functions or services under the generic title of REA. Both involve two steps, the first to quantify the resource loss, including the duration and extent of impact, and the second to identify an appropriate restoration or enhancement (Zafonte and Hampton, 2007). The choice of metric is very important to ensuring an equivalent scaling, i.e. that per unit losses or "debits" at one site are matched by per unit "credits" at another. In principle, where measures are implemented off-site, there is the potential for the use of value equivalency, i.e. the value-to-value approach. If the economic valuation is challenging to perform, the 'value-to-cost' approach can be considered as an alternative based on a remediation cost. The emergence of the concept of ecosystem services combined with advances in environmental valuation are also causing an examination of the nature of equivalency in that habitat, while protected primarily in recognition of its value to flora and fauna, also provides services of value to human beings.

The REMEDE Project (Lipton et al., 2008) describes five fundamental steps to determining the degree of compensation required:

1. Initial assessment.
2. Determine and quantify damage (the debit),

[11] Attributed to Sam Fankhauser of the London School of Economics.

3. Determine and quantify gains from remediation (the credit)
4. Scale the complementary and compensatory actions
5. Monitoring and reporting

An initial evaluation would need to consider factors such as the nature, degree, area and extent of damage. In addition, REMEDE proposes that potentially affected ecosystem services are identified *including* cultural services and social and economic issues. This stage must decide on whether there have been interim losses to date. It must also decide if primary remediation will be sufficient or if this will be incomplete in the short term. If a recovery time is involved, it must be decided if this is going to be of short or long duration and whether there will be interim losses of ecosystem services.

If it is decided that complementary or compensatory remediation is needed, then the first question is what resources or services should be remediated? From the perspective of equivalency, the need is to distinguish between species-to-species interactions and functions and ecosystem services of social and economic values. An assessment of remediation could proceed as follows:

Step 1

Initial assessment

Identify the nature and extent of the damage, the ecosystem services, whether these derive from specific species/species type (e.g. fish) or habitat (e.g. wetland), and their relationship with the environmental damage. Evidence of baseline conditions would be available for designated sites. For other locations, baseline data from similar reference sites may be needed. This could be ecological or social/economic.

Step 2

Determine and quantify the damage (debit)

This next step is to gather information on the baseline conditions, the location of ecosystem service (sources and benefits), the identity of receptors or stakeholders, frequency with which benefit realised (e.g. continuous or occasional), approximate scale, the exposure of receptors, interactions between services or between receptors/ stakeholders, and the potential for value estimation /monetisation. Together with the information from Step 1, this data can be entered into an assessment table. The determination of the causal link between impact and damage is a fundamental part of this step and may require specific technical expertise. The availability of data and the level/location of uncertainties should also be identified.

Step 3

Determine and quantify the gains from remediation (credits)

In the Step 3, the issue is the choice of remediation option and the question of how much complementary or compensatory remediation is enough? This required a calculation of the remediation credits. In essence, equivalency requires that the losses, i.e. debits, are divided by per unit remediation, i.e. credits, to determine the total remediation needed (Lipton et al., 2008).

The scaling of debits and credits is not always so straightforward. The metric should be a common to both debits and credits. If the metric is resource units, then the relevant approach is Resource Equivalency Analysis (REA). If the metric is habitats or ecosystem services, the relevant approach is Habitats Equivalency Analysis (HEA). In this case ecosystem services losses could be measured by habitat area multiplied by the percentage change in the ecosystem services. If the metric is monetary then Value Equivalency Analysis (VEA) is relevant, either a value-to-value approach or a value-to-cost approach.

The ELD indicates a preference for REA or HEA. The inclusion of human beings into the calculation begs the question as to whether either approach is sufficient based on the principles of economic welfare (Zafonte and Hampton, 2007). If not, then quantification of elements of Total Economic Value (TEV), based on utility and preferences associated with the direct use, indirect use, passive and non-use or option value of ecosystem services should enter the scaling process. Decisions may have to made on the relative importance of these different types of values. For instance, losses of income and threats to businesses or livelihood are likely to apply to direct use values (including provisioning services) and some indirect use values. These types of losses may merit compensation, but non-use values could be extensively held.

Figure 1.2 **Total Economic Value (e.g. forests)**

USE VALUE			NON-USE VALUE		
Direct use	**Indirect use**	**Option value**	**Passive use**	**Bequest**	**Vicarious**
Forest products Recreation (passive & active) Wildlife watching Hunting	Regulating ecosystem services (water protection, carbon sequestration, soil/erosion protection) Personal health or fitness Tourism income	Expectation of future personal use Value to society in future Quasi-option values/ preserving development option	Education. Landscape/ wildlife/ biodiversity appreciation. National or local identity	Pure bequest. Carbon storage	Value to others

VEA is likely to be most relevant where the nature, scale or location of remediation differs from the specific resources and services that have been damaged (Lipton et al., 2008). The very nature of interim losses introduces one difference in that society places a high value on losses/gains in the short term than in the future. This time preference is the rational behind discounting of values over time (nb. discounting is not equivalent to inflation). To estimate equivalent values for interim losses a *present value multiplier* might be needed. Information will also be needed on how long recovery is likely to take.

Use of a present value multiplier is not specific to VEA. The procedure can be applied to HEA by multiplying the *habitat area x ecosystem service* equation by a discount rate to arrive at an estimate of *discounted service hectare years* (Lipton et al., 2008). Official discount rates can be applied. For non-commercial projects, the Department of Finance recommends an annual discount rate of 5%. At a practical level, short term losses could also present problems for individuals or businesses dependent on ecosystem services.

The prospect for VEA will be assisted by the identification of sources of data and uncertainty in the first step involving the preparation of the initial assessment table described above. The assessment of complex damage could involve time consuming data collection. Therefore, the assessment process should not just identify existing data, but also the ease of data collection. In many cases, it will not be possible to quantify all ecosystem service impacts in monetary terms. Therefore a composite approach to equivalency analysis may be needed in which some indices are valued in monetary terms and others in physical units. Scoring approaches may need to be applied to the physical units in a manner akin to multi-criteria analysis.

Market Based Instruments

a) Biodiversity offsets

A potential role for market based instruments (MBIs) is introduced by complementary or compensatory remediation. Examples of MBIs include tradable permits, environmental taxes or charges, subsidies and incentives, payments for ecosystem services (PES), and compensatory exchanges or funds. The last of these includes *biodiversity offsets.*

Biodiversity offsets fall at the end of the aforementioned mitigation hierarchy which can alternatively be characterised as a sequence of avoid – minimise – reduce – offset. The presence of offsets at the end of this hierarchy implies that mitigation is either not possible or incomplete with the result that residual impacts remain.

A definition of biodiversity offsets is offered by Defra as "activities in favour of biodiversity which are carried out in compensation for expected environment impacts". Alternatively, offsets have been described as "measureable conservation outcomes resulting from activities designed to compensate for significant residual adverse biodiversity impacts" (Crowe and ten Kate, 2010).

Biodiversity offsetting has become well established. In 2010, there were 39 compensatory programmes in the world which either relied on biodiversity offsets or one-off examples. The global market is estimated to be worth \$1.8-\$2.9 billion (of which \$1.5-\$2.5 billion was in North America) (Crowe and ten Kate, 2010). There are potential direct economic benefits such as the stimulation of the environment and landscape restoration sectors which, in the US, have a turnover of \$1 billion per annum.

To ensure best practice, the Business and Biodiversity Offsets Programme (BBOP) http://bbop.forest-trends.org/ sets down the following ten best practice principles for biodiversity offsets. Offsets should involve:

1. no net loss (of species composition, habitat structure, ecosystem function, or people use/cultural value),
2. additional conservation outcomes,
3. an adherence to the mitigation hierarchy (including minimisation, on-site rehabilitation and a commitment to compensate only after appropriate avoidance of impacts).
4. acceptance that there are limitations on what is possible
5. the same landscape context (accounting for the range of biological, social and cultural values and supporting ecosystems)
6. stakeholder participation
7. equity (sharing of rights and responsibilities)
8. long-term outcomes (that are based on adaptive management and which incorporate monitoring and evaluation)
9. transparency
10. based on both science and traditional knowledge.

The challenge is to achieve a level of substitutability that does not bring biodiversity offsets into disrepute. Many offsets have been developer initiated, for instance in lieu of fees, but run the risk of potentially transferring risk, or liability, to the public or state should an offset fail to deliver environmental benefits. In principle, an offset should target the same outputs. However, substitutability is difficult to achieve. Many new habitats have been created from the likes of gravel pits, spoil or flooded land. Woodcock et al (2011) however, argue that it would take 150 years to fully restore flood meadows and even 70 years to get back the same species, aside from their former composition.

b) Conservation banking

While arguments of equivalence remain, conservation banking has emerged to allow for more flexibility and to widen the available options for compensatory conservation away from a bespoke trade in like-for-like habitats. This allows environmental liability to be transformed into a marketable asset.

In a biodiversity bank, an operator purchases a *credit* in exchange for environmental damage, i.e. the *debit*. There are also opportunities for developers (or enterprising companies or agencies) to provide their own offset. An example of conservation banking is provided by the UK's Environmental Bank (www.environmentbank.com) which has begun work of a Defra pilot with local authorities and developers in association with the Environment Agency. In this scheme, a number of Conservation Credits are allocated per hectare depending on a multi-attribute assessment of the quality of a site relative to its undisturbed state. The credits are then either matched against comparable receptor sites or are consolidated with others as part of a valued habitat. Multipliers (alternatively "ratios" or "drivers") may be applied to the credits to ensure *no net loss* and are factored to the level of uncertainty, the delay until restoration achieved or likelihood of success. The no net loss rule also allows for over-compensation where there is uncertainty. In practice, difficult decisions may need to be made, for instance where complementary habitat fulfils all main criteria, but fails to benefit one or more species found at the original site. Defra has this year launched a voluntary pilot banking scheme using this approach.

Table 1.2: The Environment Bank credit banks assigned to habitat types and credits

Tariff Band	Habitat Value	Habitat Type	Local Authority Response to development	Offset Type	Credits (per ha)
Very High	High	BAP with 'no loss' target	Strong assumption against	Within type	Bespoke calculation/ >24
High	High	Rest of BAP habitats	Assumption against	Within type	24
Medium	Medium	Semi-natural non BAP	Generally permitted	Within type of trade up	16
Low	Low	Intensive agricultural	Permitted	Trade up	8

BAP = Biodiversity Action Plan

Comparisons are often made with carbon trading as another form of compliance market. However, unlike carbon, there is no single metric for biodiversity. The question of equivalence depends on whether this applies to habitat types, ecosystem properties or ecosystem services. Following on from the UK NEA, Defra's preference is for ecosystem services to be at the forefront of habitat banking. Although The Environment Bank's system has been kept simple as it is intended as a pilot. In the United States biodiversity offsetting and banking was introduced under the Endangered Species or Clean Water Acts following a pioneering scheme by the State of California. The Williamette Partnership has developed an Ecosystem Credit Accounting System to complement its credit calculations depending on the presence of specified ecosystem functions which are deemed to contribute to an optimum habitat (www.willamettepartnership.org).

The bank must rather provide an assurance of value and of property rights. Units of trade must be backed by rules and credit registers. This process is often supported through government backed standards, accreditation or validation by a third party. Government may choose to require compensatory credits as a compliance standard, to act as a regulator, data provider, operator of the register or standards, monitoring body, broker or buyer and seller. A key reference as regards policy design is (2010) "The Use of Market-Based Instruments for Biodiversity Protection – Technical Report to EC DG Environment" (EFTEC and IEEP, 2010).

The key to an offset market is for trades to occur in units that are measurable, but which require surrogates that are simple. Credits should account for quality in addition to area and be based on a principle of representing like-for-like or better. To be acceptable to ecologists, detailed information is needed on the types, quality and status of the ecosystem and on context, e.g. landscape, corridors. The question of equivalence depends on whether this applies to habitat types, ecosystem properties or ecosystem services. The question is identical to the choice of REA or HEA introduced earlier except, that for a programme of biodiversity offsets, the stakes are arguably higher in that there is potential for a whole industry to be built around compensatory conservation. Some conservationists fear this could privatise conservation and result in a host of inferior habitats, not least because habitats require management. On the other hand, habitat banking has the potential for additionality, i.e. to deliver more valued habitats or environmental restoration than can be supplied through nature reserves or statutory conservation (McManus and Duggan, 2011). The Defra White Paper "The Natural Choice" (2011) acknowledges the recommendations drawn up in the Making Space for Nature review chaired by ecologist Professor John Lawton. It identifies biodiversity offsets as one of four natural environment priorities amongst which a principal objective is reducing habitat fragmentation.

Matched against considerations of equivalence are arguments of practicality. A degree of tailoring may be introduced to particular national needs. In Australia, for example, 'stacking' occurs where different credits are generated in one site with 'bundling permitted where credits may account for more than one environmental good or ecosystem service.

Overall, conservation banks may be more viable where like-for-like trade rules are broad. If there are too few credit types then this can reduce diversity, whereas too many runs the risk of segmentation and of outcomes falling short of biodiversity objectives (Crowe and ten Kate, 2010). To overcome the restrictions of equivalency, *trading-up* is encouraged based on 'conservation drivers' rather than matching equivalence. This allows for three types of gain, namely an improvement (better than allowed for by the guidelines), maintenance (continued control) and recognition of scarcity (achieved through new protected status). The Williamette Partnership requires buyers to purchase an additional 50% of credits to cover various environmental and financial risks.

By piloting banking through local authorities, Defra identifies potential benefits to the planning system, especially where projects are managed locally. Ideological elements may be present, but there is a potential opportunity to supplement constrained central conservation budgets. Crowe and ten Kate (2010) argue that well-managed compensatory conservation has the potential to *mainstream* biodiversity. They believe that it can be a compensation tool by which companies can manage biodiversity risk through the planning process, using licenses and financial institutions. There are benefits to business through development of goodwill or reputation, cost savings (over on-site mitigation or the risk of licenses being revoked), and through competitive advantage. Potentially, compensatory conservation can facilitate a better relationship between developers and agencies charged with protecting the environment. However, they warn that government must remain independent and avoid the risk of alignment to development interests. Offsets should not be allowed to become an alternative to public investment in conservation.

1.3 Discussion - Ecosystem service valuation and the ELD in Ireland

There is a question of how best to implement the ELD in Ireland and whether the valuation of ecosystem services for the purposes of compensatory measures can be an alternative to resource equivalence where environmental damage has occurred, in particular from interim losses before a habitat is restored. To date, the only wide-ranging estimation of ecosystem services in Ireland has been a scoping report by Optimize (2008) which estimated a selection of ecosystem service benefits worth a minimum of €2.6 billion per year. There have, however, been various environmental valuation studies, for example of forestry (Ni Dhubhain et al., 1994; Clinch, 1999; Upton et al., 2012), the agri-environment (Campbell et al., 2009), urban green space (Bullock, 2008), peatlands (Bullock et al., 2012) and water (Stithou et al., 2011b), but none of these has explicitly examined ecosystem services in any detail. A more thorough and ambitious quantification of ecosystem services is needed based on an investigation of relevant research and sectoral evidence. This is especially so given that the EC now requires that various aspects of environmental policy, for example the Water Framework Directive, be assessed using cost-benefit analysis for which a good understanding of the interface between individual preferences and ecosystem functioning is an inherent element. Valuation can enhance the efficiency with which the ELD and other EU Directives are implemented.

To further this process, the next chapter takes the example of *water*, i.e. freshwater, estuarine and inshore coastal waters While impacts to water are explicitly identified in the ELD, water is also a fundamental input to most natural habitats and to the needs of many protected species. Water and wetland habitats also provide key ecosystem services of value to human beings, including for drinking water, other water use, flood mitigation, amenity and various provisioning services. Clean water, along with the species dependent on it, are acutely vulnerable to pollution related impacts of the type most frequently addressed by the ELD.

2 Water Policy, Water Quality and Valuation Methods

2.1 Introduction

The following sections of the report build upon the preceding project chapter through the example of water. This chapter reviews relevant policy, examines the current status of water quality in Ireland and explores the contribution of previous national and international valuations. Chapter 3 describes some of the key ecosystem services relevant to freshwater environments and Chapter 4 describes the key ecosystem services found in transitional (estuarine) and inshore coastal environments.

The rationale for taking the example of water is that impacts on water quality are one of the two kinds of environmental damage identified by the ELD that Ecorisk was asked to consider. The ELD also addresses damage to the protected species and natural habitats which includes aquatic habitats and others that are dependent on good water quality to one degree or another. Moreover, some natural terrestrial habitats, such as peatlands, forest and riparian woodland are themselves Annex 1 habitats and make a positive contribution to good water quality.

2.2 Water Policy

2.2.1 Environmental damage as defined the WFD and the ELD

The WFD (2000/60/EC) is the principal policy addressing water quality. It has the objective of defending the quality of inland surface waters, transitional waters, coastal waters and ground waters. The focus of the WFD is on maintaining and improving water quality. The focus of the ELD is environmental damage which it defines as an adverse change in a natural resource or impairment of a natural resource service. The latter refers to the functions performed by natural resources (protected species and natural habitats, water and land) for the benefit of another natural resource or for human beings. For water the ELD defines damage as adverse effects significant enough to cause a change in the ecological, chemical and qualitative status and/or ecological potential as defined by the WFD. As the WFD applies to aquatic ecosystems, understanding these ecosystems and their value is central to an understanding of damage as defined by the ELD. The following sections outline the principal objectives and requirements of the WFD specifically those requiring the valuation of aquatic ecosystem services.

2.2.2 The Water Framework Directive

Objectives

The WFD was developed as an ambitious and comprehensive response to the continuous growth in demand for good quality water. It is recognised that sustainable water use requires an effective, coherent, common, consistent and integrated policy framework. The Directive builds on existing European water legislation, but provides a single comparable framework to address issues of quality and quantity of all water resources.

The principle aim of the WFD is to maintain and improve the aquatic environment across Europe based on commonly applied ecological and chemical criteria. It sets an objective for all EU water bodies to achieve “good ecological status” by 2015 (or good ecological potential in the case of artificial or modified water bodies). The focus of the WFD is therefore on the quality of the broader aquatic environment, rather than pollutants or single measure of quality. The Directive also recognises the inter-dependence between aquatic and terrestrial ecosystems.

An integrated ecosystem approach is adopted whereby the health of aquatic ecosystems is prioritised in the first instance. In this respect, the WFD is in line with other recent European Directives such as those dealing with the marine environment and soil status. Likewise, it applies a coordinated approach. Water quality had previously been addressed through standards based on human health and the direct use of water for drinking, bathing or for shellfish as exemplified by the Drinking Water Directive, Bathing Water Directive or the Shellfish Waters Directive respectively. In addition, there are Directives dealing with the discharge of pollutants, namely the Urban Waste Water Directive and the Integrated Pollution Prevention and Control Directive. Although the focus of the WFD is water quality, it recognises the need for sustainable water management to meet the economic needs of a variety of users and allows for certain derogations to water quality standards. This is in recognition of the existing regional pressures on water demand, the different water quality standard across Europe and the various responses available within technical, cost and natural constraints.

The scope of the WFD is water in its broadest sense. Water is categorised as surface or groundwater, incorporating also transitional waters (such as river mouths and estuaries), coastal waters (extending one mile from the coastline), territorial waters (12 nautical miles from the coastline), artificial water bodies (e.g. reservoirs) and heavily modified bodies.

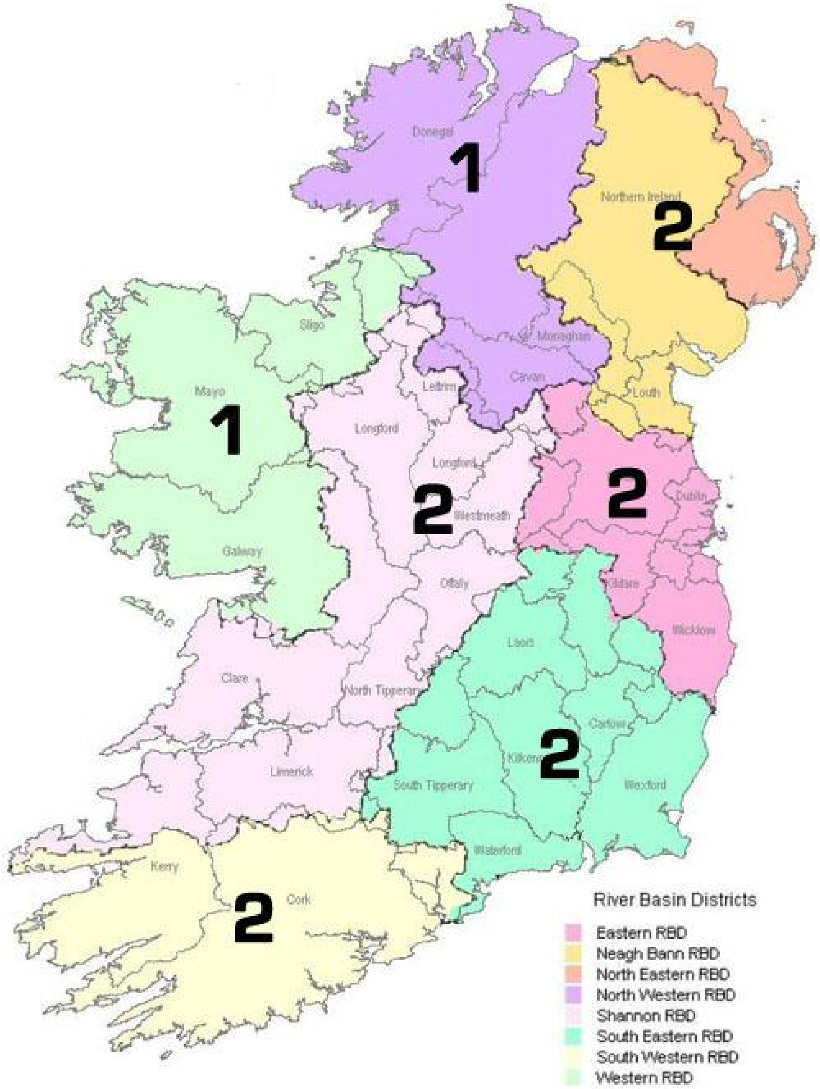
For surface waters (rivers and lakes) good status is achieved when the chemical concentration of "priority substances", e.g. organic substances, hydrocarbons and metals, is below proscribed thresholds associated with the Environmental Quality Standards set for each substance. Chemical status is either good or *fails to achieve* good status. By comparison, ecological status reflects the quality and functioning of aquatic ecosystems based on biological community, hydrological and physical-chemical factors such as temperature, oxygenation and nutrient levels. It can have a classification of high, good, moderate, poor or bad. Details on the requirements for artificial waters and ground waters are provided in Appendix 1.

In circumstances where a water body's underlying natural conditions or 'technical constraints' threaten to cause conformance with the 2015 deadline to be disproportionately expensive, the WFD allows for a *derogation* in the form of a longer time period to compliance. Any extension on grounds of technical feasibility and disproportionate cost is limited to two subsequent updated river plans (2021 and 2027).

River Basin Plans

To meet these objectives, the WFD prescribes the management structures and processes through which such standards are achieved. Management plans are prepared based on natural hydrological catchments. River Basin Districts (RBDs) combine river catchments and these in turn are divided into smaller Water Management Units (WMUs). Following an analysis of the characteristics of the river basin, the impact of human activity and an economic analysis of water use, each RBD is required to prepare a River Basin Management Plan (RBMP) detailing how good water status is to be achieved. Plans are based on the familiar DPSIR (Drivers, Pressures, State, Impact, Response) framework whereby the response is designed with respect to environmental pressures and socio-economic drivers.

Figure 2.1 River Basin Districts



Reference: SWAN Ireland

The actions proposed are detailed in a Programme of Measures (POM). Each POM includes a combination of basic and supplementary measures where the former represent those actions already required under existing European legislation. Basic measures are those already required under existing EC Directives (see Table 2.1). These include:

- recovery of costs,
- protection of drinking water,
- controls on abstraction,
- controls on point and diffuse sources of pollution,
- control of physical modifications to surface waters (e.g. registration or prior authorisation),

Supplementary measures are applied in addition to the basic measures to achieve the good water status requirement of the WFD.

- prevention of accidental hazard that could cause pollution or losses of pollutants from industrial sites.

Supplementary measures can be locally specific and relate to the status of a site, diffuse or specific activities associated with point source pollution, dangerous substances, abstraction, land use, peat extraction or water use. They could include more stringent environmental controls or the re-creation or restoration of wetland habitats (ESB International, 2008).

2.2.2. The relationship between the WFD and other Directives

Basic measures are drawn from earlier water-related European Directives. The WFD complements, and in some instances, replaces a number of these Directives. Several Directives adopted in the mid-1970s focussed on setting quality standards to protect human health and the environment, e.g. drinking water, bathing water and shellfish waters. Standards were also set for the discharge of dangerous substances by industry. Later legislation began to focus more on the sources of pollutants. For example,

- the Urban Wastewater Treatment (UWWT) Directive required significant investment in collection and treatment infrastructure;
- the Nitrates Directive controls the use of nitrogen within agriculture;
- the Integrated Pollution Prevention and Control (IPPC) Directive provided for restrictions on pollutants discharged by industry.

The WFD encompasses all existing water regulations and rationalises some of this existing legislation. The Directive identified groundwater and priority substances as requiring further legislation resulting in the subsequent Groundwater Directive (2008) and Directive on Priority Substances (2006). Achieving good status by 2015 requires low levels of chemical pollution as required under much of the existing water based legislation. The innovative element of the WFD was to combine this with the ecological status as evidence of a healthy aquatic ecosystem.

Table 2.1 Directives representing basic measures required by the WFD

Bathing Water Directive	2006/7/EC
Habitats Directive	92/43/EEC
Birds Directive	79/409/EEC
Drinking Water Directive	80/778/EEC amended by 98/83/EC
Major Accidents (Seveso) Directive	86/82/EC
Environmental Impact Assessment Directive	85/337/EEC
Sewage Sludge Directive	86/278/EEC
Urban Waste Water Treatment Directive	91/271/EEC
Plant Protection Products Directive	91/414/EEC
Nitrates Directive	91/676/EEC
Integrated Pollution Prevention Control Directive	96/61/EC

The WFD therefore specifies the role of basic measures founded on existing Directives and supplementary measures.[ii] In addition, it takes a combined approach for the control of discharges to surface waters which requires the adoption of relevant emission controls, emission limit values or best environmental practice as specified by existing Directives.[iii]

2.2.3 Protected areas

Under Article 6 of the WFD, RBDs are required (Article 6) to create a register of protected areas as specified under European legislation. These include:

- Special Protection Areas (SPAs) designated under the Birds Directive;
- Special Areas of Conservation (SACs) designated under the Habitats Directive;
- Economically significant aquatic species identified by the Freshwater Fish and Shellfish Directives;
- Recreational/bathing identified by the Bathing Water Directives;
- Nutrient sensitive areas identified by the Nitrates and Urban Wastewater Treatment Directives.

Protected areas also include locations designated for the abstraction of water intended for human consumption and nationally protected areas of natural or cultural conservation value, i.e. National Heritage Areas (NHA's). In doing so, the WFD explicitly recognises the importance of water quality to users and to other ecosystems and the biodiversity they support.

Protected areas designated under the Birds and Habitats Directives (BHD) are included in the Natura 2000 network. Any aspect of ecological status considered under the WFD must take account of the impact on the conservation status of these sites. For example, stricter phosphorus levels may be required that go beyond those normally sought to achieve good ecological status. As the objectives of the Directives are not defined in a consistent way they must be determined on a case-by-case basis. In general, good ecological status/potential of a water body will contribute to the favourable conservation status of species/habitats (EC, 2011)

While the principle aim is to protect habitats and species, Natura 2000 plays an important role in providing and maintaining a range of ecosystem services within sites and across the wider environment. A recent study (ten Brink et al., 2011) provides a very broad estimate of the overall economic benefits of the Natura 2000 network across the EU using an ecosystem services framework. Scaling-up existing benefits estimates from a number of sites (i.e. using a sites-based methodology), the benefits of the whole network were estimated at €223-€314 billion per annum. Alternatively, the report provided estimates of €189-€308 billion per annum based on existing per hectare values for habitats (habitat-based methodology). In addition, the study considered a number of existing estimates to quantify and value the Natura 2000 network's contribution to the delivery of individual ecosystem services (i.e. carbon storage and sequestration; natural hazard mitigation; climate adaptation; tourism and recreation; water provision and purification; food-related provision; health, identity and learning benefits). Based on a simplified extrapolation from these studies, the ten Brink et al report estimated the annual value of freshwater provided by the entire Natura 2000 network as falling within a wide range of between €2.8 – €3.2 billion. However, it added the caution that huge assumptions were having to be made in aggregating values from a small number of studies to the whole Natura network.

In Ireland, aquatic areas designated for protection for reasons of economic value are based on the stipulations of the Irish Shellfish Regulations or indirectly via the Bathing Waters Regulations, Nutrient Sensitive Areas or Drinking Water Directive (the EC Shellfish Directive was superseded by the WFD). There are approximately 70 shellfish production areas listed in the Irish Shellfish Regulations. Areas designated for recreation relate only to the 131 bathing areas listed in the Irish Bathing Waters Regulations. Nutrient-sensitive areas are those waters listed in the Irish Urban Waste Water Treatment Regulations, but no Nitrate Vulnerable Zones have been designated in Ireland (although some areas were considered). Areas protected under the Drinking Water Directive are restricted to the water body from which the water is abstracted. While these related Directives do not provide a single value of water quality, they highlight the importance of protecting the quality of water as a key input into key economic sectors such as fishing and recreation.

In addition, rivers and wetlands, together with the ecosystems that underpin them, have economic implications in relation to flood mitigation. The Flood Policy Review Group (OPW, 2003) included in its recommendations the inclusion of environmental issues in the cost-benefit-analysis of flood management options and formal recognition of non-structural flood management measures such as the contribution of wetland ecosystems to flood attenuation. This includes soft engineering works such as wetland creation, restoration and management. Under the Floods Directive, Member States are required to have prepared flood hazard and flood risk maps by 2013 identifying the potential impact of flooding and the number of citizens and types of economic activities that could be affected. In this respect, the Office of Public Works (OPW) have prepared a number of Flood Risk Assessment and Management (CFRAM) studies within the RBDs. In some instances, these studies have acknowledged that non-structural options represent a robust response to flood events and their likely increase in frequency and magnitude over time due to climate change. As the WFD specifically requires Programme of Measures to take account of all relevant existing EU legislation, the integration of flood management strategies into the 2007 Floods Directive is required.

2.2.4 The WFD and economic values

Rather than relying on the traditional tool of regulation and standards, a fundamental element of the WFD is its requirement for economic costs and benefits to be taken into account in catchment management plans and for the introduction of full social costing of water use. As noted above, the principle of full cost recovery is embodied in the Directive characterised by the Polluter Pays Principle and the efficient use of water based on benefit pricing or willingness to pay (Morris, 2004). The value of water as a resource is acknowledged to extend beyond conventional uses such as for industry, agriculture and fishing to include such non-market values as recreation and biodiversity.

The WFD provides for sustainable water use across the whole aquatic ecosystem through the application of standards based on the ecosystem approach. As these quality standards are there to be achieved, they do not necessarily require a monetary valuation of aquatic ecosystem services. Member States are simply required to select the most cost-effective measures to achieve good status. However, there are elements of the WFD where a detailed analysis of the benefits of achieving the standards is required. EU guidance provides a role for ecosystem services valuation which can prove useful in instances where water damage occurs, i.e. a deterioration in water quality status as defined by the WFD or where subsequent remediation is required under the ELD. The purpose of this Chapter, together with Chapter 3 is to consider in more detail those aspects that can help in our understanding and valuation of ecosystem services. While such valuations relate specifically to the benefits of good water quality, they can be a useful indicator of the value or cost of water damage under the ELD, i.e. a reduction in water status.

One of the first steps of the WFD is the preparation of a RBD Characterisation Report. This includes an analysis of the river basin's characteristics, a review of the impact of human activity and an economic analysis of water use. A National Summary Report must be submitted to the European Commission. Ireland's report, the Characterisation and Analysis of Ireland's River Basin Districts, was submitted in 2005 (EPA, 2005a).

The characterisation reports require that an economic analysis is undertaken to identify the most cost-effective combination of measures to be included in the Programme of Measures. While the Directive does not define cost-effectiveness, subsequent Commission guidance (EC, 2003b) clarifies that a cost-effectiveness analysis requires an identification of environmental objectives, an assessment of possible measures to meet these objectives and an estimate of their costs.

Irish guidance (Goodbody, 2008a) on implementing the cost effectiveness measures required by the WFD recognises that at present it is difficult to comprehensively quantify costs in relation to consumer welfare impacts or external costs because a set of robust values for these impacts is not available. Rather, the guidance recommends measurement of cost-effectiveness based on costs that are readily quantifiable. The guidance adds that significant non-quantifiable cost impacts should be "noted for consideration in the appraisal process".

A cost effectiveness analysis (CEA) requires only that costs are monetised. This is in contrast to the use of a cost-benefit analysis (CBA) in which both costs and benefits should be quantified. It therefore does not provide specifically for the valuation of the possible aquatic ecosystem service benefits arising from any alternative measures required to meet good water status. However, water pricing policy, and any extensions or derogations based on disproportionate cost, do require consideration of both costs and benefits.

Ireland's characterisation report contains a comprehensive baseline assessment of the status of Ireland's waters and so provides a basis for future river basin planning. It also includes an identification of human related pressures as required by the WFD. The report identified environmental and resource costs as those that competing water users impose on one another without making compensation, i.e. external costs. As data on these costs are not readily available, the report used estimates of the marginal cost associated with future investment in wastewater treatment services as an indicator of the environmental/resource costs. This provides an estimated partial cost of €4.3bn for meeting the WFD objectives over the period between 2004 and 2012 based on the projected wastewater treatment expenditure identified in local authorities' Water Services Investment Programme reports.

Although this approach did not provide a valuation of aquatic ecosystem services, it does provide an indicative value. It also indicated the value of benefits foregone (i.e. the environmental and resource costs arising from a loss of ecosystem services) for waters that fail to meet the good status standard. These figures can be compared with the direct costs of preserving or restoring the benefits.

The report also referred to a preliminary assessment of the benefits and costs associated with the water resource in Ireland, i.e. the 'Economic Analysis of Water Use in Ireland' (CDM, 2004). This contained a partial valuation of services such as water-based recreation using existing data such as that collected by the Economic and Social Research Institute (ESRI) on behalf of the Marine Institute (Williams and Ryan, 2003). This data mainly involved estimates of expenditure and related economic activity.

Water Pricing

The WFD requires Member States to recover the full costs of water services as a basic measure. This includes the recovery of the environmental and resource costs associated with damage or negative impacts on the aquatic environment. Water pricing can ensure that management costs and externality costs are reflected in the price of water helping also to identify the most cost-effective options for inclusion in a Programme of Measures.

Environmental and resource costs are not defined in the WFD. Subsequent commission guidelines (EC, 2003b) consider environmental costs to be the damage that water uses impose on the environment and ecosystems and those who use them. This includes use and non-use values. Brouwer et al (2009b) interpret environmental and resource costs to imply full cost recovery, i.e. of total private and social production costs of a good or service.

Disproportionate costs

As discussed, each RBD can request a derogation, i.e. an extension or exemption to the requirements of the WFD on grounds of disproportionate costs. Although the Programme of Measures is based on the most cost-effective actions and does not specifically require the estimation of benefits, Commission guidance (EC, 2009) notes that assessments of disproportionate costs will be incorrect unless an effort is made to value or assess social and environmental benefits/costs.

However, the guidance does acknowledge that it may be difficult to attribute a monetary value to many environmental or social benefits.

Related Commission guidance (EC, 2003a) explores the potential for market based methods, revealed and stated preference. It remarks that cost methods, i.e. estimates of the cost of maintaining an environmental benefit, can provide a practical option and reasonable estimates of the environmental value, albeit an underestimate. Subsequent guidance, i.e. Brouwer et al (2009b).provides more detailed information on the types of costs and benefits, on methodologies for cost-benefit analyses (CBAs), and recommendations on which costs and benefits to include and how to evaluate them

Irish guidance (Goodbody, 2008a) recognises that a request for an exemption on grounds of disproportionate cost must in some way relate to the benefits of an improvement in water quality. Given the lack of available data on benefits, the guidance recommends the use of benefit values from the UK (benefit transfer) under certain circumstances and conditions. Related guidance, i.e. Goodbody (2008b), makes specific reference to the Environment Agency (England and Wales) benefit values for both use and non-use and provides an indicative methodology for relating UK water quality measures to Irish measures. The guidance further recommended the development of a set of Irish benefit values for use in the WFD context.

The implementation of the WFD provides an opportunity to understand and examine instances of the valuation of ecosystem services. However, while European and Irish guidelines are available, practical applications within Ireland have been limited. No RBD has undertaken a full cost-benefit analysis. It is likely that future Irish River Basin Management Plans will require a more comprehensive understanding of the value of aquatic ecosystem service.

2.3 Water Quality in Ireland

For the period 2007-2009 (EPA, 2010c) the EPA reported that:

52.0% of river bodies were of high or good status
47.3% of lakes were of high or good status
84.7% of the area of groundwater aquifer was of or good status.

These measures of water quality confer with the ecological status criteria used by the WFD. The figures for surface water appear modest, but the status categories are based on the worst performing ecological component of the sample and compare relatively favourably with most EU Member States, The relative figures for river channels using the traditional Irish classification for the 2007-09 period were High 20.1%, Good 48.8%, Moderate 20.7%, Poor 10%, and Bad 0.4%. The traditional methods measured physico-chemical and biological parameters (typically organic pollution) and toxic substances. The WFD adopts a more integrated approach that includes also additional biological factors such as the presence of non-native species, specific pollutants and hydromorphology.[12]

a) Rivers

Ireland has 13,000km of river channel of which (as above) 52.0% are classified as being of good status. Rivers in the least populated Western and South-Western River Basin Districts were rated highest. Twenty seven rivers were assessed as being seriously polluted, but the contribution of municipal waste water outflows to these incidences of serious pollution has reduced to eight rivers (from 15). Agriculture accounted for three, landslides and bog bursts for a further three incidences.

[12] The previous EPA “unpolluted” category has been split into high” and “good” categories corresponding to the ecological criteria underpinning the WFD Good status..

Other factors such as historic mining, forestry, landfill and construction works were responsible for the remainder.

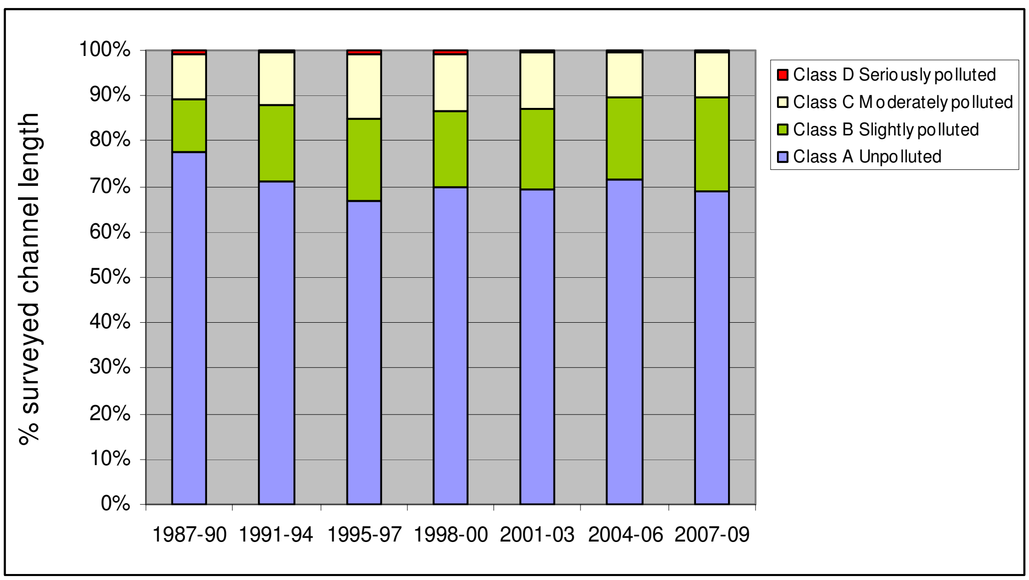
b) Lakes

Ireland's lake area amounts to 1,500km^2. Forty seven percent (47.3%) of lakes were of high or good status as indicated above (mostly good status), although almost as many (41.4%) were of moderate, poor or bad status (mostly moderate). However, unenriched oligotrophic or mesotrophic conditions accounted for 92.1% of total lake area using the traditional EPA classification. Most lakes retained a similar quality in the latest reporting period to the previous period. Indeed, there was also some reduction in cases of the most seriously enriched lakes, although the EPA suggests that some of this could be due to the filtering effect of introduced zebra mussel, the long term implications of which are still being debated. Only nine lakes are listed under the EU Bathing Water Directive, 67% of which met stringent EU guide values.

c) Groundwater

Groundwater is a source of drinking water for approximately 25% of households. It is also a contributory factor to the quality of rivers, lakes and estuaries, particularly during periods of low-flow. The extent of groundwater of good status has increased as a result of reduced phosphate and nitrate due to the combined effect of reductions in fertiliser usage, improved application and high rainfall during the survey period. However, the WFD criteria do not record microbiological condition. Faecal coliform bacteria were reported from 34.8% of samples and this represents an increase on previous figures.

Figure 2.2 Trends in river water quality



Source: EPA (2009)

Although freshwater water quality in Ireland compares favourably with many other EU States and instances of serious pollution have declined, the length of unpolluted river channel continues to show a long term decline while the extent of slightly polluted river has increased as shown in Figure 2.2. The principal threats to this water quality arise from point sources of pollution, for example, municipal sewage and industrial outflows, from non-point discharges, namely septic tanks, and from diffuse pollution from agriculture (EPA, 2010d). Eutrophication is a major consequence of this pollution due to the increased presence of limiting nutrients such as nitrogen, but especially phosphorus, in freshwater ecosystems. The result is excessive algal growth followed by oxygen depletion once the algae decay.

The contribution of waste water treatment plants to eutrophication has reduced as new municipal facilities have come on line. In addition, an increasing number of residences and small businesses that might formerly have depended on on-site disposal or treatment facilities are being connected to these networks. New legislation relating to the registering, inspection and maintenance of septic tanks should, in due course, reduce this element of pollution too. Phosphorus levels from diffuse agricultural pollution have fallen too due to reductions and management of organic fertilisers, but the reduction in nitrogen emissions has been more modest due to continued use of nitrogen fertilisers.

The quality of public drinking water supplies has continued to improve due mainly to investment in treatment infrastructure. Incidences of *E-coli* contamination were reported in 2.2.% of supplies in 2010 (EPA, 2011b). The relative level of contamination of private group water schemes was much higher at 11.6%, but here too this problem has been declining. Chemical compliance was high at 99.2%, although the EPA report notes that there is a need for improvement in reductions in organic matter and the risk of trihalomethanes as well as a need to reduce levels of aluminium and turbidity. Seventy water supplies remained on the Remedial Action List due to the high risk of Cryptosporidium contamination. The availability of water for drinking is a highly important ecosystem service. More details on drinking water quality are discussed below.

2.4 Drinking Water and the role of Aquatic Ecosystem

2.4.1 Drinking Water Supply and Quality

Amongst the principal values that we attach to the aquatic ecosystem is its capacity to provide us with drinking water. To illustrate the distinction between average and marginal values, economics text books often refer to the low average value of drinking water compared with its high marginal value in times of scarcity. It is, of course, fundamental to life. In general, though, its importance often goes unnoticed. Ireland does not suffer from a scarcity of water. Water deficits do occasionally arise in dry or freezing weather, but shortages typically arise from constraints in the supply and distribution infrastructure whose capacity is predicated on the expectation of regular rainfall. These assumptions could prove inadequate in the future under some projections of long term climate change that foresee drier summers in the future (Sweeney et al., 2009). There is an expectation that new storage or the piped transport of water from the Shannon could become necessary making water a more expensive commodity in the future.

Another reason that water is presently perceived as being an abundant commodity is that private households do not pay for its use directly. This situation is likely to change once water pricing is implemented to reflect the true value of water. To date, the cost of water provision is met through central taxation, the revenue from which is then transferred to local authorities who are responsible for supply. Largely as a consequence of the under-pricing of water, its use by Irish households is much higher, at an average of 160 litres per day, than in every other European country. A reason for this high rate of consumption, and one related to the low price of water, is our use of relatively water-inefficient domestic utilities including water hungry showers and washing machines. A more fundamental reason is our own habit of using larger amounts of water than is necessary compared with other countries where consumption has been moderated by water charging. In Denmark, for example, where all consumption is metered, average household consumption is 116 litres.

Most drinking water in Ireland is supplied through public schemes (945) managed by the local authorities. Smaller communities are supplied by public Group Water Schemes (671), i.e. schemes where the community has opted to allow responsibility to be passed to the local authority, or by private Group Water Schemes (457). In addition, there are a large number of small private sources (1284) such as individual wells or boreholes (EPA, 2011b). Prior to the progress made under the current Water Services Investment Programme (beginning 2000), the poor quality of drinking water, mainly that supplied by private wells and the private Group Water Schemes, led to the threat of EU fines for non-compliance with the Drinking Water Directive. To avoid this risk, the state has invested heavily in the water supply and treatment infrastructure. Many former private schemes have either been converted to public management or required to accept tenders for Build Design Operate (BDO) treatment plants. Bacterial coliform contamination has been a particular problem due to inadequate, or inadequately managed, household septic tanks and animal waste in the vicinity of water sources.

The EPA Report on Drinking Water Quality (2011b) reveals continuing improvement in the quality of potable water. The report presents the status of the following contaminants:

- Faecal coliforms – bacteria such as *E-Coli* associated with human and animal waste;
- Cryptosporidium – a protozoan causing potential severe gastrointestinal illness[13];
- Turbidity – suspended sediment. Often an indicator of bacteriological and cryptosporidium risk as sediment increases the potential for their survival.
- Trihalomethanes – by-products of chlorination in the presence of organic matter.
- Metals
- Organic and inorganic chemicals including by-products
- Algal blooms.

At noted in the previous section, the EPA remains concerned about the management and treatment of organic matter in drinking water sources (EPA, 2011b). However, compliance with acceptable trihalomethane levels has reached 87.9% with exceedence in the public schemes having fallen to 79 incidents (where trihalomethanes have been sampled for). Other chemical constituents, including lead, nitrates and pesticides, were generally absent in all or over 99% of cases with the exception of fluoride (97.9%).

This leaves coliform contamination as the greater problem. Although *E-coli* was detected in only 2.2% of public supplies, it was found at source in 5.1% of catchments (water supply zones). The number of incidences of contamination has fallen to just 0.01% of larger public supply samples, but remains high at 11.6% amongst private group water schemes but has fallen from 17.0% in 2009. Plants are also being required to have controls against the *Cryptosporidium* parasite following serious outbreaks in drinking water in Counties Galway and Westmeath as a result of inadequate water treatment due to problems of domestic waste pollution. Seventy supplies were classified as being at risk in 2010, but it was expected that this would fall to 35 in 2011. There were 46 instances of inadequate source protection in 56% of audits, including from animal access.

[13] Protozoa are unicellular eukaryotic organisms distinct from bacteria and more closely related to algae.

Table 2.2 Non-compliance with Drinking Water Standards for samples taken in 2009 (EPA, 2011)

	Public water supplies		**Public Group Water Schemes**		**Private Group Water Schemes**	
	No. of non-compliant samples	% of non-compliant samples	No. of non-compliant samples	% of non-compliant samples	No. of non-compliant samples	% of non-compliant samples
Microbiological parameters						
E. coli	33	0.3	5	0.4	122	6.5
Enterococci	20	0.8	1	0.5	14	4.2
Chemical parameters						
Lead	29	1.1	0	0	1	0.2
Nitrate	5	0.1	1	0.2	3	0.3
Trihalomethanes	186	12.6	24	29.6	28	10.6
Indicator parameters						
Aluminium	228	2.9	54	5.4	31	2.6
Turbidity *	115	7.8	1	3.4	7	17.9

* at treatment works

The WFD will require users to meet the marginal cost of water supply which includes the impact of water abstraction on the environment and of adequate treatment of water to remove contaminants. The installation of water metering is being rolled out along with the establishment of a new privately managed water utility, Irish Water. Although progress is being made under the Water Services Investment Programme, the cost of water provision has risen due to the substantial expenditure on improving water quality in response to the State's previous failure to comply with the Drinking Water Directive.

The WFD also requires polluters, including private households, to meet the full marginal cost of the management of waste water. There are an estimated 380,000 septic tanks in Ireland discharging at least 65 billion litres of effluent into the countryside each year, approximately one quarter of which is toilet waste.[14] Most of those built prior to new building regulations coming into force in 2001 provide for inadequate treatment while many others are inadequately maintained. According to research prepared for the National Spatial Strategy, only one third (at most) of septic tanks are emptied on a regular basis and many others have been constructed in locations with inadequate ground conditions for percolation, e.g. shallow soils (DEHLG, 2000). The inadequate disposal of this waste contributes to eutrophication of the environment and to the faecal coliform counts in drinking water. In response, government is belatedly requiring rural households to register their septic tanks for the purposes of regular inspection. This action has, however, come too late for the State to avoid the imposition of a daily penalty for non-compliance of €12,000 by the European Court of Justice.

2.5 Valuing Water Quality, Wetlands and Ecosystem Services

2.5.1 Public Good Characteristics

The public good characteristics of water resources derive from the fact that water use, be this for consumption, recreation or as a medium of waste is not costed at its full social value based on scarcity or abundance or the equalisation of marginal social benefits and costs, but rather in terms of private extraction costs (Birol et al, 2006). As such, private benefits and costs diverge from the social benefits and costs that should guide sustainable water use including the use of water for consumption or for the receipt of waste (waste sink). The WFD aims to achieve the sustainable

[14] Household discharge figures based on Gill et al (2006) and lower estimates from UK DoE figures and Environment Agency Pollution Prevention Guidelines. An average toilet flush uses between 6 and 9 litres water.

management of water resources by ensuring that the full social benefits and costs of water resources are identified for the purpose of policy and decision making. Many of these benefits and costs, such as those associated with recreation or biodiversity, are non-market and do not appear in financial balance sheets. Environmental valuation can fill this gap and has the additional benefit of promoting transparency in decision-making, identifying welfare gains to society, providing information on water resource use, and demonstrating the public's perception of the value of water (Aquamoney, 2007). However, while the WFD seeks to establish a common EU standard for water quality, a common framework for the monetisation of environmental costs does not exist (Kallis and Butler, 2001).

One challenge to a common standard for guiding the efficient use of water resources is the difficulty of accounting for the costs and benefits to different users. In principle, equi-marginal returns should apply whereby all users face the same marginal values and with these being equated to the marginal cost of supply. Opportunity costs would be taken into account including all external costs and benefits. In practice, however, the estimation of marginal values by use type is extremely difficult. At present, each use currently has its own distinct value depending on its ability to reduce or substitute water and on the contribution of the water input to the value of economic output (Moran and Dann, 2008).

Various studies have been undertaken of the value of water. The principal studies of relevance to Ecorisk are discussed below. A wider selection of studies is included in the database. Most of these studies have examined the value of water as an environmental asset which conforms with the interpretation adopted by the WFD. Some studies have been more specific to wetland habitats and some have looked at the value of water from the perspective of particular users, for example for passive recreation, angling or kayaking.

There are on-going studies supported under the EPA Strive programme that are using public surveys to arrive at a definitive guidance on the social value of water quality objectives. The objective of Ecorisk is not to produce an alternative set of survey-derived values, but rather to focus on evidence of the value of ecosystem services (or the benefits foregone in the case of a pollution or similar incident) and the choice of methods for their valuation. Social and cultural values apply to these estimates in terms of the value we place on the regulating services performed by aquatic ecosystems as well as to services that fall within the distinctly cultural ecosystem domain such as recreation.

2.5.2 Irish estimates of the value of water quality

a) Expenditure related values

A number of studies have been undertaken of the direct and indirect expenditure associated with water-related recreation. These studies do not capture the full use value (welfare value of associated with use) that is attached to environmental goods, but do illustrate a proportion of this value as well as indicating relative value compared with other non-water related alternative activities. These types of studies are popular with government agencies or departments as they indicate a stimulus to the economy.

Water-based recreation

A survey of 4,400 participants in water-based leisure was undertaken in 2003 by the Economic and Social Research Institute (ESRI) on behalf of the Marine Institute to update baseline data collected in 1996 (Williams and Ryan, 2003). The report examined participation and expenditure in various activities including angling, boating and seaside trips. It concluded with a figure for total annual expenditure of €434 million by 1.48 million people (35% of the population). The report did not provide estimates of the utility value to the individuals, although a subsequent report by Failte

Ireland (2009) did use a scoring system to identify those water bodies that were most valued for water based tourism.

Angling

Failte Ireland reports a decline in the number of visiting anglers since the 1970s, with current numbers of overseas visitors amounting to 103,000-127,000 per year (Failte Ireland, 2009; 2011). Both game (salmon, trout) and coarse angling are important economically for various rivers/lakes or for different parts of the country. A report prepared for the Central Fisheries Board (now Inland Fisheries Ireland) by Indecom (2003a) provided estimates of the economic/socio-economic value of wild salmon in Ireland. The report was part of a wider objective to provide advice on the long-term sustainable management of wild salmon. Specifically, it examined the economic value of recreational rod fishing relative to commercial drift and draft net fishing which at the time accounted for 87% of the total value of the salmon catch. The report estimated a total direct economic value for the commercial fishing sector of €4.8 million as of 2002. By comparison, the combined net economic value from angling by domestic anglers and overseas visitors was estimated at €11 million per annum. The relative values of the two activities provided evidence for the debate over the viability of game angling given the competing impact of commercial fishing of stocks. More recent data on participation and the value of angling has been provided by Inland Fisheries Ireland (IFI) and is discussed in the next chapter.

b) Estimates of generic water use and non-use values and cost estimates

DKN Report

In 2003 the DEHLG commissioned a consortium led by DKM (2004) to propose a methodology for the valuation of water supply and wastewater projects for the purposes of CBA. The DKM report set out guidelines for the valuation of the external benefits to the receiving environment due to waste water treatment funded under the EU Structural Fund. The report did not arrive at actual valuations, but rather referred extensively to the guidance provided by the Environment Agency (England and Wales). It suggested the use of benefit transfer values for non-use benefits (i.e. conservation and biodiversity) due to improvements in water quality based on studies by Georgiou et al (2000a) and Willis and Garrod (1996)

CDM Report

CDM (2004) were commissioned to provide an initial overview of the current and projected economic benefits and costs associated with water use in Ireland. This analysis was subsequently used for the National Characterisation Report presented to the EC under the requirements of the WFD (EPA 2005).

Water use benefits were one of four categories considered by the CDM report. Estimates were provided for the agricultural and industrial sectors, including key sub-sectors. In-stream uses were also addressed including angling, boating, beach visitation and other water-based leisure use. National estimates of user expenditure were used to provide a partial value relating to those who engage in these activities. Willingness-to-pay (WTP) sourced utility values were not estimated directly.

The report also considered the value of wetlands and special riparian areas corresponding to NHAs, SPAs, and SACs. In the absence of primary quantitative estimates of the values of these sites, it too proposed the use of benefit transfer values. A WTP/person/hectare/year for non-use was calculated on the basis of a number of studies of Environmentally Sensitive Areas in Scotland (MacMillan et al., 1996; Hanley et al., 1997).[15] Per person values were multiplied by the numbers of hectares of wetlands and then multiplied by the populations in the relevant District Electoral Divisions (DEDs). However, these studies were concerned with an agri-environmental scheme rather than specifically with water quality or wetlands.

The CDM report and the subsequent Irish national report (EPA, 2005b) identified the environmental/resource costs that competing water uses impose on one another, i.e., external costs. Estimates of the cost of upgrades to wastewater treatment facilities were proposed as partial estimates of the value of the marginal benefits that would be foregone if this expenditure were not undertaken. Based on the wastewater treatment expenditure projected in the Water Services Investment Programme to improve surface water quality, the partial national public environmental/resource costs for the period between 2004 and 2012 were estimated at €4.3bn. This methodology was applied to estimate environmental/resource costs for each RBD.

c) User based studies

Salmon angling - John Curtis ESRI (2002b)

The aforementioned studies present values primarily at the aggregate level by proposing estimates of expenditure related to water based recreation to estimate the welfare value of water quality. Amongst the first Irish studies to estimate the welfare value associated with use of the aquatic environment was an estimate of the demand function for salmon angling by Curtis (2002a). Data was collected by means of an on-site survey of anglers visiting Donegal. This was used to estimate a mean travel cost across all anglers of IR£68 (€86) per day. Once combined with an estimate of WTP the value of salmon angling in Donegal was estimated at IR£206/day (€261) per salmon angler. Although the study was not specifically related to water quality, it did include angling quality (good or excellent) as an explanatory variable.

Whitewater kayaking - Stephen Hynes and Nick Hanley (2006)

Hynes and Hanley (2006a) applied the Travel Cost Method (TCM) to value demand for whitewater kayaking.. The Roughty River in Co Kerry was chosen as it is considered one of the best kayaking rivers in Ireland and also because its hydro-power potential was under consideration at the time. The paper estimated a consumer surplus of €83 per trip per kayaker. Based on an estimated 2.83 trips per kayaker per year, this provided an average consumer surplus or WTP of €235 per kayaker per year. The study examined competing demands on the water resource, but did not examine water quality specifically.

In a subsequent paper (Hynes and Hanley, 2009) the authors included water quality along with other site attributes such as crowding and scenic quality. However, they found 'water quality' to be statistically insignificant in the model. Rather, it seemed that the overwhelming value held by kayakers was attached to a river's physical status, namely its 'star rating' of whitewater rapids. Much of this status could be presumed to arise from naturalness, although a minimum standard of water quality for the health of users can be presumed to be essential even if the marginal values of variations in water quality were insignificant. Non-whitewater kayakers were not surveyed, but it is possible that water quality would feature more strongly in the relative value they attach to attributes of the aquatic environment.

d) Studies focused on water quality

[15] Note that the transfer value is not like-with-like as the Scotland studies were estimating the benefits of elements of an agri-environmental policy. However, only a limited number of wetland valuation studies had been undertaken in the UK at this time.

Primary studies of water quality at catchment level

A very recent study by Stithou et al (2011a) represents the first study in Ireland to have directly examined the welfare benefits of achieving good ecological status under the WFD. The study addressed water quality at the level of a water body, in this case the River Boyne, located within the Eastern RBD and one of 40 Hydrometric Areas (HA's) defined by the WFD. The river and its tributaries are of importance to agriculture (which is also a source of diffuse pollution), for abstraction and of for natural and cultural heritage, tourism and recreation.

Water quality was represented by indicators of ecological status. Both discrete choice experiment (DCE) and the contingent valuation method (CVM) were used to estimate the economic welfare values associated with significant improvements in water quality as represented by four attributes, namely water appearance, recreational opportunities, river life and river bank. A range of increases in annual household tax payments were used as the 'price attribute' or payment vehicle for eliciting WTP for variations in levels of these physical and ecological attributes. A total of 525 households were surveyed face-to-face with additional data collected on respondent and household characteristics including distance from nearest water body. At the time of the study, approximately 19% of the river system was classified as being of 'good ecological status'. The survey provided estimates of marginal WTP for a "high impact" improvement under the WFD to "good status" for which the respective mean value was €32.70 per person per year.

Table 2.3 Attributes and level sued in the Stithou et al (2011) discrete choice experiment

Attribute	Description	Attribute levels
Fish, insects and plants	composition and abundance of biological elements (fish, plants, invertebrates, mammals and birds	1. Poor 2. moderate 3. Good
Condition of river banks	Level of erosion and presence of vegetation (shrubs, trees) and animals (mammals and bids)	1. Visible erosion 2. Natural looking banks
Water appearance	Clarity, plant growth, visible pollution, noticeable smell	1. No improvement 2. Some improvement 3. A lot of improvement
Recreational activities	Number of activities available	1. No fishing and swimming 2. No swimming 3. All available (walking, boating, fishing, swimming)
Cost	Annual household tax for 10 years	€0,5,10,20,40,80

e) Benefit transfer studies

The DKN and CDM reports both proposed use of benefit transfer in the absence of any national primary studies of the welfare value attached to water quality. As described in the preceding chapter, benefit transfer is a means by which economic values estimated for a good or service at one site can be transferred to another similar site for which a valuation is desired. At the most straightforward level estimated values can simply be adjusted to account for differences between the baseline and the study site, for instance differences in area or the size of the human (recipient) population. A value function approach is more efficient in that local data, for example on variables such as income, can be substituted into the original valuation function.

Norton et al (2012) have recently undertaken the first thorough benefit transfer exercise for water quality in Ireland for the purposes of estimating the non-market value of a surface water body achieving good ecological status. While acknowledging that very sizeable differences can arise between studies, the authors argue that benefit transfer can provide a "bedrock" of values for policy analysis. In the absence of Irish studies, the authors refer to the report by Goodbody (2008) which gave conditional support for benefit transfer given the paucity of Irish studies and the relevance of UK work given the similar geographical context.

A key consideration to ensuring comparability is to make correct allowance for distance decay (the anticipated decline in WTP values with distance). In addition, it is important to ensure the relevance of the 'economic jurisdiction', i.e. by including all those who have a significant value for a resource, an area that will vary considerably depending on whether the resource is judged to be of local or national importance (Bateman et al., 2006b; Stithou et al., 2011a)).

- Unit transfer

In the first instance, Norton et al applied a unit transfer approach to value the achievement of good environmental status across the full set of 151 Irish WMUs. The boundaries of these were overlaid with population figures for the associated Electoral Divisions (EDs). The figures omitted possible users living outside these EDs, but were supplemented by estimates of foreign tourist visits. Taking the number of EDs within 40km of water bodies in the Boyle catchment together with the adult population (>15 years), the tourist resident equivalent was estimated at 51,285, or a 3.6% addition to the resident population. Tourist WTP was adjusted on the basis of counties visited, duration of stay and expenditure.

Water quality status was standardised for each county based on a change from its average water quality status to "at least good status". This meant, though, that if a county already has an average high quality water status, then WTP would appear as zero as the WFD is concerned with achieving this quality standard for lower-performing water bodies. The authors note that while expenditure may be required to maintain good quality status, the true relevant welfare measure would be willingness to *accept* rather than WTP, a value that tends to be greater than WTP, but one that is difficult to elicit in practice.

Norton et al could identify only five studies that they thought were relevant to a unit transfer exercise in Ireland based on site similarity and the presentation of changes in water quality. These studies are those conducted by Georgiou et al (2000b), Hanley et al (2006), Bateman et al (2009), Del Saz-Salazar et al (2009) and Martin-Ortega and Berbel (2010). Together, the studies suggested the average mean values given in Table 2.4 based on large, medium and small changes in water quality as described in each study. These mean values were then weighted by the percentage of water bodies under each quality status in each WMU and adjusted by tourist numbers in each county. Aggregate values were mapped using GIS, but are evidently related to the size of the ED population rather than individually held values that may be higher amongst river users or people living nearby. Consequently, catchments with sizable populations appear in the top five ranked WMUs, whereas less populated catchments with higher existing levels of water quality appear in the bottom five.

Table 2.4. Average benefit values for changes in water quality status (household per year) (Norton et al).

Change in water status	mean	stan dev.	N	95% lower limit	95% upper limit
Large change (e.g. Poor-Good)	€66.46	€45.59	2	€3.28	€129.63
Medium change	€38.98	€20.81	5	€20.73	€57.22
Small change	€31.77	€20.71	3	€8.34	€55.22

N = the number of estimates from the five studies.

Table 2.5. Value of achieving good ecological status: 5 highest ranked and 5 lowest ranked WMUs (Norton et al. (2012)

River Basin District	WMU	Value (€)

Highest values		
Eastern RBD	Tolka	2,800,352
South Western RBD	Lower Lee/Owenboy	1,459,072
Eastern RBD	Cammock	1,416,838
Eastern RBD	Dodder	1,175,405
Eastern RBD	Shanganagh	792,670
Lowest values		
South Western RBD	Upper Bandon	3,395
North Western RBD	Eske	2,887
South Eastern RBD	Tar	2,610
South Western RBD	Glengarrif	1,296
South Western RBD	Sheen	388

- Adjusted benefit transfer

To provide a comparison based on applied data, Norton et al next applied an adjusted benefit transfer for an improvement in the Boyne catchment river quality to Moderate status. Five UK studies were selected for this benefit transfer on the basis of similarity of context. The figures were again adjusted to account for ED population size and tourist numbers, but also now for distance decay based on the estimates of Bateman et al (2006a) for a river of "regional importance". An aggregate value of €13.6 million was estimated for an improvement from Moderate to Good status. This value compares with the Stithou et al (2011a) estimate of €19.1 million, suggesting a transfer error of about one third.

- Function transfer

The final benefit transfer method applied by the Norton et al study was that of a benefit function exercise. This study used the function estimated in the Stithou et al study of the River Boyne to estimate the value of achieving good ecological status in 15 other water bodies. For each of these river bodies, experts identified the relevant attributes of river life, condition of river banks, water appearance, recreational activities and cost. For each attribute, current conditions were identified corresponding to the levels/categories used in the Stithou study relative to the levels that would correspond to good ecological status. The study identified the compensating surplus values of achieving good ecological status for each of the these river catchments, of which that for the Boyne was €57.73 per household/year. This function transfer estimate compares with a figure of €32.70 to achieve the same good status estimated by the original Stithou et al study.

Overview

Benefit transfer is appealing in that it offers an inexpensive alternative to an original primary study. However, practical difficulties arise because of the inexact match between the studies' objectives, sites or the recipient populations. The Norton et al (ibid) study concluded that benefit transfer estimates should only be used to compare the relative values across water bodies or where the demand for accuracy is relatively low. The use of benefit transfer estimates by RBDs in relation to decisions of disproportionate cost is not recommended and a primary survey is proposed wherever feasible. The Norton et al report recognises the limitations of relying on only one study to guide benefit transfer estimates and recommended additional primary studies to provide further validation checks.

2.5.3 UK and European valuation studies

Environment Agency and DEFRA

A greater number of studies of water quality values have been undertaken in the UK, elsewhere in Europe and internationally. The most useful studies for the purpose of benefit transfer to Ireland are listed in the database. The UK studies are alluded to by the DKM and CDN reports and by the Norton et al study. The Environment Agency (EA) is the competent authority for implementing the WFD in England and Wales.

Cost-effectiveness analysis is required to identify actions for inclusion in the Programme of Measures. Defra (2004) acknowledges that the identification of disproportionate costs is necessary for extensions or exemptions permitted by derogation and argues that these must be based on the principles of cost-benefit analysis. This move from CEA to CBA requires the valuation of the environmental benefits associated with improvements in the water quality.

The NERA-Accent study (2007) was commissioned to provide estimates of the benefit of water quality improvements to the wider UK environment. The study used stated preference (SP) methods to estimate the potential non-market benefit of the WFD nationally and by RBD. Potential benefits were divided into those associated with direct use, including in-stream recreation, and indirect use, from other forms of recreation, relaxation and aesthetics near-stream. The benefit estimates included near and long-term potential or option values based on the maintenance of a good environment.

Both DCE and CVM methods were selected. Following a commonly applied convention, CVM was used to estimate total benefits of the programme while DCE was applied to value marginal changes in attributes arising from policies. A baseline of no deterioration or no improvement at any site was compared with the improvements in ecological status required to meet WFD criteria. The payment vehicle adopted for both the DCE and CVM was a combination of water bills plus higher prices on everyday products. A total of 1,487 interviews were completed across 50 different urban areas from 11 river basin districts.

The survey used three methods of value elicitation that resulted in a wide range between means of £45 (€53) and £168 (€198) per household per year within which the true mean WTP was presumed to reside. The use of estimates from the lower end of the range was considered appropriate for policy-making purposes, i.e. £55 (€65) per household per year for an increment in water quality from low to moderate quality or from moderate to good. The greatest proportional benefits were estimated for a move from moderate to good status reflecting the prevalence of these quality levels

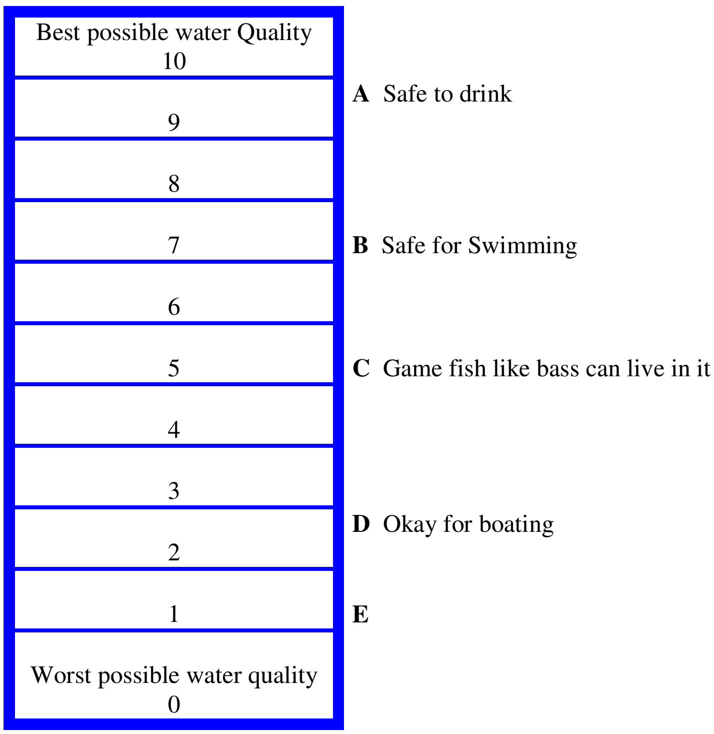
The UK River Basin Plans were published in December 2009 and plan-specific benefits information was included in the impact assessment accompanying each plan. A national impact assessment of these RBPs identified the public's WTP for the improvements in water quality status. This estimated annual benefits at £1.3 billion (€1.5bn) accounting for households' valuation of biodiversity, aesthetic value and amenity value over a 43 year period (EA, 2009).

Morris and Camino (2011) used the EA and NERA estimates to value the non-market benefits associated with improvements in water quality in both rivers and lakes. Their study identified the total aggregate benefits of improvements to good quality status of rivers and lakes across England and Wales at £1.1 billion (€1.3bn) per year (a figure similar to the EA estimates). The EA themselves drew on these results to estimate average benefits per kilometre of river per year for improvements in water quality from low to medium (£15.60), from medium to high (£18.60) and from low to high (£34.20), respectively (€18.40, €21.95, €40.36).

Benefit transfer

Defra identified benefits transfer as a source of data for valuation, but proposed further study of the method along with primary studies of use and non-use values. New guidelines produced by Defra in 2010 supply practical advice for benefit transfer. A technical report (Eftec, 2010b) provides the basis for the new guidelines and defines best practice. A further study by Eftec (2010a) provides a case study of the river Aire to demonstrate the benefits of water quality improvements following investment in waste water treatment. The study first defined the policy good and the affected population along with the change in the provision supplied by policy. A source study by Ferrini et al (2008) was selected from six candidate studies due to its focus on the 'generic' features of English rivers. The Ferrini et al study had the virtue of accounting for substitute sites and distance. It also measured quality using the ordinal 'water quality ladder' (see Figure 2.3) developed by Vaughan in 1981 on the basis of the US National Sanitation Foundation's Water Quality Index and first applied by Mitchell and Carson (1981). The resulting function allowed for the estimation of WTP for water quality improvements for households in each square kilometre. The estimated annual household WTP for each km^2 was multiplied by the number of households in that km^2, and summed across the entire affected population area to calculate the annual benefit of improved water quality. Annual benefits were estimated at are approximately £4 million (€4.7m) (EFTEC 2010a).

Figure 2.3 Water Quality Level (Mitchell and Carson, 1981; Vaughan, 1986)



2.5.4 Meta analysis

A meta-analysis compiles a valuation function based on the results of multiple studies. In principle, the range of data can provide a function that is solidly founded on data from numerous locations and which is not unduly influenced by the specific characteristics of any one site. Brouwer and Langford (1999) undertook such an analysis of 30 wetland studies which provided for 103 value observations. Their analysis explained 37% of the observed variation in value. The mean WTP for wetland preservation value was estimated at \$93 (€70) per household per year with the median value at \$51 (€38) (1995 \$ prices).[16] Woodward and Wiu (2001) also applied a meta-analysis of 39

[16] Brouwer and Langford used national currencies expressed in terms of 1990 purchasing power and then converted in International Monetary Fund's (IMF's) Special Drawing Rights (SDRs), the Fund's official monetary unit of account. Average WTP for wetland function preservation in all studies is 62 SDRs (at end of 1995, 1 SDR approximately equalled 1.5 US\$ = \$93 as above). The median was 34 SDRs (not equal \$51 as above).

studies resulting in 65 value observations which explained 58% of variation. Amenity and landscape were the factors of most influence. The study also reported diminishing returns to scale.

A more recent meta-analysis of wetlands was undertaken by Brander et al (2006a). This extended to 190 studies of which 80 were examined providing 215 observations. The meta-analysis explained 45% of variation. In this case, benefits were found not to vary with wetland size, but rather with population density (see also comments on the Stithou study). The indirect benefits of flood control and water quality were also valued highly. A subsequent larger meta-analysis by Brander et al (2008) extended to 264 observations and explained 43% of the observed variation in benefit estimates. The model was in turn applied to a UK case study by Morris and Camino (2011) which estimated an overall UK benefit of £303 (€358)/ha/year for all inland waters or £270 (€319)/ha/year and £333 (€393) for lowland and upland wetlands respectively. A meta-analysis by Ghermandi et al (2011) using a wider international database (253 observations) identified the importance of wetlands in mitigating human-induced pressures.

a) Aquamoney

Aquamoney was a multi-state EU Framework project, the objective of which was to develop and test practical guidelines for the assessment of environmental costs and benefits in the WFD. Specifically, the aim of the project was to provide guidance in relation to the cost recovery of water services and exemptions on the grounds of disproportionate costs.

Technical Guidelines were produced for use by expert practitioners and economic specialists who carry out valuation studies while policy briefs were provided for policy advisors or policy/decision-makers. The Guidelines address market and non-market goods and services within a Total Economic Value (TEV) framework with links to specific water ecosystem goods and services, e.g. drinking water, fishing, industrial process water, recreation, biodiversity, natural sink, etc. The Guidelines also identify appropriate economic valuation methods.

The technical guidance included a meta-analysis of 154 international stated preferences studies to identify values of ecosystem services related to surface water quality published between 1981 and 2006. Many of the studies identified contained a number of values for single and/or multiple ecosystem services, including use and non-use values. The review noted that most of the survey-based contingent valuation studies combine use values with some element of non-use value, but that it is difficult to assess the size of each component of total value.

The mean values identified varied considerably across different ecosystem services. For example, a mean value of $513 (2007 USD/household/year) was identified for ecosystem services relating to health; the comparable value for boating was $76, and that for non-use value was $129. The considerable variation within each category of ecosystem service was illustrated by high standard deviations for all values. For example, respective standard deviations of health, boating and non-use were $283, $66 and $155 for non-use. Differences in survey approaches, the characteristics of the water services valued, and in the water service beneficiaries, contributed to the significant variation..

The identification of a clear description of the change in water quality (water services valued) that could be translated into a standardised measure was considered to be the critical information requirement, i.e. the stated WTP for a change in water quality is partly dependent on the magnitude of the proposed change in quality. The 154 studies employed a range of different reporting formats, but those studies that valued a change in water quality from a baseline were converted into the 10-point water quality index. Of the 154 studies originally identified, this 10-point index could only be constructed for 54 studies, but with 388 separate value observations. The 10-point index was based on the above water quality ladder by Vaughan (1986) and estimated the average value of a one unit change in the index at US$89/household/year. The estimated coefficient of the water quality change variable was found to be very small and positive, but not statistically significant.

The Guidelines also noted that average values differed across different water types, for example, wetlands were valued significantly more than rivers and rivers were valued more than lakes. Contrary to many valuation studies, WTP for future welfare gains was almost twice that WTP to prevent a welfare loss. This has relevance to valuations of adverse impacts on water quality and may have been due to the perception of rights to existing water use being stronger than the effect of loss aversion associated with an endowment effect (see Chapter 1).

2.5.5 Limitations of survey based estimates of water values

In surveys such as those described above, the public have identified clean water as an environmental attribute that contributes highly to quality of life. However, the choice of payment vehicle in these surveys can be problematic. In Ireland, the public are accustomed to the free provision of drinking water and many would see this as right for which they should arguably be asked a willingness-to-accept (compensation) question rather than a WTP question as Norton et al (2012) note. By comparison, in the UK, it can be difficult for respondents to distinguish their true value for clean water from the actual water charges they face. Problems of ‘strategic bias’ can plague WTP estimates given widespread public dissatisfaction with private water companies. Even at English water prices of up to £1.50/m^3 (€1.77/m^3) there is likely to remain a sizable consumer surplus (Morris and Camino, 2011).

The scenario presented can also be problematic. It is important for a survey to ensure that respondents know what it is they are being asked to value. In the first instance they need to distinguish between their marginal value of water quality for drinking and the value of water quality in the environment. In Ireland, this exercise could be compounded by competing forces in that the quality of drinking water has improved while the quality of natural water bodies has been gradually deteriorating due to eutrophication. Perceptions are important. Valued elicited from survey methods tend to capture impacts only when these are visible or sufficient to impact on the types of uses associated with the water body. This is, indeed, consistent with the ELD that identifies an impact as being relevant only where this is sufficient to impact on the water quality status as defined by the WFD. It has been argued that the ELD sets a high bar in this respect in that “water bodies” tend to be quite large and an impact would need to be quite marked in order to change the quality status of an entire water body. [17]

2.6 Summary

Water policy, and specifically the WFD, has considerable relevance to the ELD in that it sets the standards for water quality for Member States to achieve and also the criteria for environmental protection. Chapter 2 has provided the background to the WFD and also information on the current status of water quality in Ireland. Ecosystem services perform an important regulating role in protecting water quality as is discussed in the next chapter and high water quality in turn protects the environment for the provision of other ecosystem services. The importance of water quality as a measure of environmental quality has meant that there have been numerous applied studies into the value attached to this non-market good by society. The chapter has described how environmental valuation methods have been applied to water quality, but found that rather few have addressed water quality in a manner that is of practical value to the implementation of either the WFD or the ELD and that there has only been one primary stated preference study in Ireland to date.

[17] Valerie Fogleman *pers comm*

3 Freshwater Ecosystem Services

3.1 The Ecosystem Services performed by water and related habitat

As water and protected species are both addressed by the ELD, this chapter demonstrates the role and value of freshwater ecosystem services and the implications of their loss in the event of an adverse environmental impact. Rivers, lakes and wetlands harbour many ecosystem processes (physical, chemical and biological interactions), but for ecosystem processes and functions to provide ecosystem services, depends on these functions being of value to society (Turner et al., 2000; de Groot et al., 2002).

The current value of ecosystem services is a function of demand for those services. Water often provides a classic example of the marginal value paradox in that the demands we make of this resource are increasing to the point where the underlying functions themselves are at risk of being overwhelmed despite the high marginal value that is attached to them (Soderqvist et al., 2000). A high ecosystem service value is not an inevitable consequence of a valued environment. The service that aquatic ecosystems provide as a sink for the removal of pollutants from wastewater or diffuse pollution are an example in that the particular ecosystem service associated with a clean self-regulating river is quickly overwhelmed. Current ecosystem processes may reach a threshold and the river quickly be transformed into a rather different ecosystem, but not necessarily one that society values for its capacity to support the wide range of familiar species associated with less polluted environments or for contact and non-contact amenity use. An ecosystem service perspective should seek to value these services as benefits foregone wherever possible rather than indirectly in terms of damage avoided, i.e. the cost of policies to reduce pollution or the cost of treatment infrastructure to replace the natural ecosystem service.

An understanding of the value of rivers, lakes and wetlands and of the benefits of their wise and sustainable use precedes all current EU Directives and national legislation. The importance of wetlands for biodiversity conservation, water use and livelihoods was officially recognised by the Ramsar Convention of 1971. This long established international treaty obliges signatory states, including Ireland, to maintain the ecological character of wetlands of international importance. Ireland has 45 Ramsar sites represented by fens, bogs, lakes, bays and estuaries.

Ireland's National Biodiversity Action Plan (EPA, 2010a)) lists the strategies in place to safeguard, monitor and assess biodiversity. For rivers, lakes and wetlands, the document clarifies the EPA's role in implementing the ELD and its enforcement activities in relation to seriously polluted river stretches, the preservation of high quality water status and wetlands protection as well as the management of Waste Discharge Licenses, IPPC licences, Waste Water Sludge management and audits of drinking water treatment plants. Each of these activities has relevance to water and to protected aquatic species. The EU Biodiversity Strategy to 2020 (EU, 2011) complements these obligations by requiring Member States to map and assess the state of ecosystems and their services by 2014, and to provide an assessment of economic value. By 2020 these values should be integrated into accounting and reporting systems at EU and national level. While the EU Strategy doesn't specifically define or categorise ecosystem services, it will incorporate the findings of the recent Common International Classification of Ecosystem Services (CICES) undertaken by the European Environment Agency (Haines-Young and Potschin, 2013b).

3.2 Categories of freshwater ecosystem services

It is useful at this point to reintroduce from Chapter 1 the Millennium Ecosystem Assessment (MA, 2005) categories of ecosystem services and to describe those relating to water.

a) Supporting ecosystem services

For water, the key supporting services derive from biodiversity related processes including primary and secondary production, food web dynamics and nutrient cycling. These functions are maintained through the supply of clean water to habitats such as wetlands, flood plain and riparian woodland and the species they support.

b) Regulating ecosystem services

Key regulating services include the assimilation of pollutants and the biological control of pathogens, flow/flood moderation and sediment capture. These services occur within both flowing water and in wetlands along with the maintenance of soil fertility (e.g. deposition of silt) on flood plains and carbon exchange (the balance between sequestration and emissions).

Peatlands and fens provide valuable regulating services in the form of carbon sequestration and a degree of moderation of surface run-off. Freshwater marshes have an important role in flood mitigation with the service realised through the physical presence of the wetland vegetation which complements topography by holding back water flow and excess sediment. Marshes neutralise and transform excess nutrients or pollutants. Both fens and marshes provide for water storage and groundwater recharge, services that could be valuable to all uses in drought conditions.

c) Provisioning ecosystem services

The main provisioning service is the supply of fish for consumption (or wildfowl and crustaceans). The CICES report describes the supply of water as “problematic” given that it is essentially abiotic, but accepts its inclusion as a provisioning service given convention and interactions with supporting services including habitat.

d) Cultural ecosystem services

Water has a wide public appeal. Cultural services include the value of rivers and wetlands for amenity (including appreciation of wildlife or landscape), for recreational angling and for direct contract activities such as kayaking, sailing or bathing. There are related knock-on benefits for tourism, including tourism expenditure, and for health, cultural heritage and education.

Table 3.1 Freshwater ecosystem services

Supporting services
Habitat
Genetic diversity
Regulating services
Assimilation of waste and nutrients
Biological control
Sediment capture and deposition
Flow and flood moderation
Carbon exchange
Provisioning services
Water supply (potable and other)
Fish (and other species) for human consumption
Reed
Cultural services
Recreation
Cultural / natural heritage
Health
Education

Scale is a factor particularly for regulating and provisioning services. The meta-analysis by Brander et al (2006b) reported a wide variation in values. Although the mean average wetland value was estimated at $2,800 per hectare, the database possessed a median value of just $150 per hectare. Marginal values may diminish with greater scale, although Woodward and Wui (2001) report finding constant returns to scale for larger wetlands included in their own meta-analysis. The per hectare value of all services will depend on the size of the human population especially that proportion which benefits from ecosystem services either directly through consumption or use or indirectly via regulating services.

The attached matrix (Appendix 5) lists the principal ecosystem services associated with particular types of habitat along with examples of keystone and characteristic species. The number of asterisks under each heading provides an indication of the relative importance of various habitats or key species. Appendix 1 lists wetland types found in Ireland and Appendix lists Annex 1 habitats.

3.3 Provisioning services

3.3.1 Water Supply

There are few provisioning services associated with rivers and wetlands in Ireland aside from water supply. Until recently, commercial salmon draft netting was permitted on the lower reaches of rivers and accounted for almost 20% of the €4.8 million value of commercial salmon fishing as of 2002 (Indecon, 2003b). However, this activity has been effectively prohibited since 2007 by restrictions introduced to preserve stocks particularly for higher value recreational angling. Provisioning service benefits now apply only in relation to the supply of water to trout farms.

The principal provisioning service is for drinking water and the supply of water for agriculture and industry. Water is abstracted for these purposes from groundwater, lakes and rivers with water quality requirements depending on the use to which the water is put. Pressure can be placed on lake ecosystems where abstraction risks causing an excessive fluctuation of the water level (CDM, 2009).

Although water is an abiotic resource, its supply is to a large extent determined by surface vegetation and ecosystems and their ability to collect water from the atmosphere or rainfall, to release this water to surface or subsurface channels, and to return it to the atmosphere as evaporation. While the supply of water is accepted by CICES as a provisioning ecosystem service, its quality for many purposes is maintained through the regulating services provided in-situ by the aquatic ecosystem. The value of these respective services can be difficult to distinguish and are ultimately a function of the final benefit to human beings. The use of water for the cooling of power stations requires no significant treatment whereas water used for consumption clearly has to be of a high quality. For many uses in between these extremes it is possible to use water of varying quality and even quite poor quality water can be rendered potable with sufficient purification or treatment.

Water abstraction

The WFD recognises that abstractions have an influence on the quality status of a water body along with ecological, chemical and other hydrological criteria. In many countries, abstraction has an important impact on the ecosystem as water is in short supply. An excessive loss of water (including reduced flow) will directly impact on habitat and wildlife, but also the quality of the water source as the reduced volume fails to dilute pollutants. Even in Ireland competition for water supplies is increasing and there is the prospect of reduced future supply due to climate change. The latest European Commission report on the implementation of the WFD noted that water abstractions in Ireland are in general sustainable (EC, 2012(a)), but that demand is increasing. In response, the government is preparing legislation to control abstraction and impoundments of water.

Water quality determines the suitability of water for abstraction. The designations used for special protection by RBDs include areas of abstraction for human consumption. The National Characterisation Report (EPA, 2005a) estimated that 626 million cubic metres of water were abstracted annually (approx 1.7m^3 per day) from 2,318 known surface abstraction points. The report acknowledged that a number of water abstractions were unknown and unregulated. The register was updated for the first RBD Management Plans (2009-2015).[18] While the register now contains most public and group water schemes, it is considered unlikely that all industrial and miscellaneous small private abstraction schemes (e.g. schools, hospitals, etc) were captured.[19]

In contrast to some other Member States, Ireland has an overwhelming reliance on water from surface sources (71%) compared with groundwater (29%).[20] Across Europe, the principal uses of abstracted freshwater are urban water demand (14.%) (domestic, commercial and industrial use), agriculture (30%), industry (10%) (excepting cooling water), and electricity (energy) (32%). The corresponding principle sectoral uses of water use in Ireland are urban water demand (39%), agriculture (15%), industry (21%) and energy (23%) (EEA, 1999).[21]

Much of the abstraction from surface sources in Ireland is performed by the public authorities. Groundwater is a significant source for many individual households and businesses that have their own private source of supply. Due to the limited information available on abstractions by users other than public water supplies it is difficult to identify and assign a value to these uses. The majority (94%) of known surface water abstractions are used for public water supply (WFD Ireland, 2005). Users of the public water supply are defined as domestic or non-domestic. “Domestic use” generally refers to private household who are not charged. “Non-domestic use” refers to business users, but includes a range of different users such as trades, agriculture and hotels. An estimated 35-47% of public water supply is used by non-domestic users (OECD, 1999; CDM, 2004).

Two key properties of public goods are those of rivalry and excludability. For water, consumption by one user does not necessarily diminish the amount available for others to consume. Whereas for public water supplies it is feasible to selectively exclude users by cutting off supply, it is more difficult to identify, monitor or prevent private abstractions, in particular of groundwater (Kolstad, 2009). Property rights are central to the divergence between public water supply and direct water abstractions. In effect, the Irish government is exercising a degree of ownership via extraction for public water supply. Ownership is not defined or regulated for other abstractions, although, in principle, a register of all water users could lead to a system of licensed extractions with ownership vested in the government.

To date, water pricing has been limited to non-domestic users as households do not yet pay water charges. A single water charge is imposed across a range of non-domestic users does not capture the marginal value for each user. This value depends on the ability to reduce or substitute water and the contribution of water as an input to the value of the final product. The single price paid by non-domestic users reflects the physical supply costs. Users are paying for the capital and operating costs of the water supply infrastructure, but are presented with no charge for the water itself (Hanemann, 2005).

As discussed in Chapter 2, the WFD requires Member States to introduce water pricing to encourage the more efficient use of water resources. Pricing should reflect the full cost of water services including the environmental and resource costs associated with negative impacts to the aquatic environment. The most recent European Commission assessment of Ireland implementation

[18] Data was available for just over 90% of the abstraction points.
[19] Easter River Basin District, Abstraction Pressure Assessment *Background to Water Matters Report – 22 June 2007*
[20] In 2009 a total of 517 million cubic meters (m^3) of water was abstracted from Irish surface waters, compared to 213 m^3 from groundwater. Eurostat, Water Statistics 2012.
[21] This report notes that information on abstraction and use in different countries often do not correspond due mainly to different definitions of the concepts.

of the WFD (EC, 2012(a)) notes that Ireland has rather adopted a narrow view of 'water services', i.e. current operations involving the collection, treatment, storage and distribution of water and waste water. The WFD adopts a much broader definition, including abstraction for irrigation or cooling and well-drilling for agricultural, industrial or private consumption. The Commission believes that Ireland's use of a narrow definition significantly reduces the scope of the analysis and cost recovery (EC, 2012(b)).

The current level of water charges varies between local authorities depending on the local full cost recovery without profit, including capital, operation and maintenance costs. Indeed, local authorities are at different stages in the implementation of full cost recovery and current charges may not reflect the full costs. Secondly, within each local authority area, the same price is applied to a variety of users, but the *value* of this water will vary by and within different sectors depending on the use and value of the final output, i.e. water does not appear as an input cost which impacts on the final price paid by the consumer. Thirdly, while commonly referred to as water charges, non-domestic water charges generally incorporate a charge for both water and waste water services. The principle of water-in water-out applies, i.e. the charge is based on water supplied, with the assumption that similar volumes of wastewater are discharged. In principle, though, it would be possible to separate the price paid for these services.

The current Irish definition offers little opportunity to use water charges to reflect the true value of water. Despite this, in the absence of other available information water service charges can be used to provide an indicative, albeit lower bound, estimate of the value of water supply.

Value of Water Supplies

Water use can provide direct benefits, for instance where it is "extractive" or consumed (e.g. as drinking water), or indirect use benefits, i.e. recreation or aesthetic values. Recreational activities such as kayaking or angling, as well as navigation, could be thought of as making direct use of water while being non-extractive or non-consumptive. For present purposes a key distinction is between the value of water used for abstraction (e.g. for human consumption) and the value of water in-situ (e.g. environmental quality).

In 2004, CDM conducted an economic analysis of water use in Ireland as part of the Characterisation Report required by WFD. The report was based on existing available information and provides an overview of economic benefits and costs associated with the use of water resources. The economic impacts of key water-using sub-sectors were based on five common economic impact parameters: establishment costs, gross output values, gross value added estimates, employment and wages and salaries. Information on all parameters was not available for each sector and data from different years were employed.

The study identified key water-using or water-dependent sectors for which water is a critical resource input based on volume of water used or the absence of suitable substitutes. This dual approach captured sectors where volume was low, but where dependence on water is high, i.e. high marginal value.

The CDM report identified three categories of water use values:

- Abstraction was based on a per-unit basis. For example, cattle and sheep water use values were based on per unit use estimates multiplied by animal counts and the standard charge. Domestic water use values were based on a per capita consumption rate.
- In-stream water use relating to angling, boating, beach visitation and other water-based leisure. Values were based on the national Marine Institute/ESRI (Williams and Ryan, 2003) study of estimated expenditure by those engaged in such activities.

- Other water use valuations considered were those relating to wetlands and special riparian areas (incorporating NHAs, SPAs and SACs. Non-use values were based on estimates from England and Wales, Scotland and Austria.

Additional water uses are recognised in the report, but valuations are not provided due to the absence of data. These include inshore commercial fishing, aquaculture, hydroelectric, water transport, forestry and logging.

a) Households

When examined separately from water *quality*, the maintenance of water *supply* has been shown to be important to private households. In Ireland, the domestic sector has not so far been charged for water supply. In the UK, the water regulatory agency is quoted by Moran and Dann ((2008b) as having estimated the public's willingness-to-pay for reduced supply interruptions at between £29 and £33 per year (approx €34 & €39). The figure is much less than the actual charge for clean water supply.[22] Drawing on estimates by Moran and Dann, the Scottish Environmental Protection Agency estimates marginal values for household treated water of £0.50/m^3 to £1.38/m^3 (€059-€1.62) compared with raw water supply at £0.23/m^3 to £1.38/m^3 (€0.27-€1.62).

b) Agriculture

Rain provides Ireland's grass-based dairy sector with a significant cost advantage, but a explicit value is not assigned (DAFF, 2010).[23] Due to Ireland's high annual rainfall there is little need for irrigation and only an insignificant percentage of total agricultural land is irrigated (Baldock et al., 2000). The principal sectors of Irish agriculture, i.e. milk, beef and lamb production, are primarily rain-fed.[24] For example, less than 2% of total water consumption in the beef and dairy sectors is provided by abstraction from rivers, groundwater or from mains water supplies. Of the water that is abstracted for use on Irish livestock farms, approximately 90% comes from groundwater and 10% from mains water supply (Hess et al., 2012). Where irrigation does take place, over 80% is abstracted from surface waters and used primarily used for early potatoes, vegetables and soft fruit (Baldock, ibid).

In principle, a value for direct water abstraction for use in agriculture can be captured where information is available on input costs and the output value. Using this information, the maximum amount the water user is willing to pay for the water input can be identified as the difference between the revenue from production and the other input costs, i.e.

water value = (price * output) – non-water input costs.

This netback approach was employed by Moran and Dann (2008a) to calculate the value of water to the potato crop in Scotland. Their study was undertaken on behalf of the Scottish Environmental Protection Agency (SEPA) and was based on two Scottish river basin catchment areas (West Pfeffer and Tyne), the only locations for which the necessary data was available. The study identified a mean value of £5,128/ha (€6,050/ha) for water used for irrigation of potatoes. As this value included both water for irrigation and naturally available water, Moran and Dann compared yields with/without irrigation to identify the value of water used for irrigation. This provided an estimated

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[22] although OFWAT (Jacobs[22]) have estimated willingness to pay at £10 per day for uninterrupted supplies once account if taken of the environmental state of the source water body.

[23] Due to our abundant rainfall water availability may not be considered an issue. The water exploitation index (WEI) is used to identify pressure or stress on freshwater resources, i.e. amount of water abstracted each year as a proportion of total long-term freshwater resources. A WEI above 20% implies water stress. The latest WEI data indicates a WEI for Ireland of 1.5%. EEA/Eurostat latest data

[24] Department of Agriculture, Food and the Marine Fact Sheet on Irish Agriculture (April 2012); DAFM Website

average value for water used for irrigation of potatoes in the UK of between £0.23 (€0.27) and £1.38 (€1.63) per m^3 (2003 UK£).[25]

c) Industry

While some information is available on water use by domestic and agricultural sectors, there is a lack of information on water use and value for industrial sectors.. In order to understand the value of water to individual industries, a detailed understanding of the production processes and water use for each sector/user would be necessary. One such approach, involving the construction of a cost function for individual sectors, has been applied by Renzetti and Dupont (2003b) to Canadian manufacturing.[26] The restricted cost function provided a valuation of water input by sector through the change in short-run costs associated with an incremental change in the quantity of intake equivalent to a shadow value of the firm's marginal WTP. The study supplied values for water intake for 14 Canadian industrial sectors and arrived at a mean value of $0.046/$m^3$ (€0.03) (based on 1991 CAN$).

A subsequent study of Canadian industry by Dachraoui and Harhaoui (2004) examined water and non-water inputs to estimate the marginal value of water for 36 industries/sectors. This study also estimated a shadow price across all industries averaging $0.73/$m^3$ (€0.53) (1991 CAN$) per cubic metre. Dachraoui and Harhaouri explain the difference from Renzetti and Dupont as being due to "irreconcilable" differences in coverage, consistency in the data sources, methodological differences and estimation techniques.. When the recirculation of water in the production process is considered this reduces to $0.55/$m^3$ (€0.40) (1991 CAN$).

An alternative marginal productivity approach was adopted by Wang and Lall (2002) to estimate the value of water to Chinese industry. Using data from around two thousand Chinese industrial firms, a marginal productivity function for water was derived and used to provide an estimate of the marginal value of water for industrial use (2.45 Yuan/ m^3 (€0.30)). The study notes a large range of values across sectors and between regions. A more recent study (Ku and Yoo, 2012) employed the same approach to estimate the average value of water to Korean industry (USD 1.05/m^3 (€0.79) (2010) per cubic metre). Both studies identified water values that are considerably greater than the actual price paid by users. In theory, both cost and production function approaches should yield the same results as the marginal cost should be equal to marginal value of production under the assumption of profit maximisation.

Sufficient data on water use and non-water costs for Scottish industry was not available to Moran and Dann (2008) to replicate the netback approach used for agriculture to industry. Instead, their estimates of the value of water to industry relied on the findings from Renzetti and Dupont (2003a) above. Accounting for inflation and exchange rates, the value of water to thirteen UK industrial sectors (UK 2004£/m^3) were presented. Values ranged from £0.003 (textiles) to £0.157 (refined petroleum and coal products) to (€0.004-€0.19). Moran and Dann note some limitations to the application of the Canadian data to the UK. While they assume that water use is similar between Canada and UK industries, the implicit assumption had to be made that there has been no change in the efficiency of water use over time. This is unlikely to hold in practice and so the values are at best indicative only of the relative value of water use in different industrial sectors.

3.2.2 Water treatment

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[25] Moran and Dann note that the applicability of these estimates across different agricultural uses and different locations is limited. Price and output will vary considerably over time, with different locations facing different natural conditions.

[26] In Canada, almost all water intake is from private abstraction and requires a permit that cannot be easily transferred or altered. This regulation ensures that water has a quasi-fixed input cost.

Artificial water treatment for consumption

Nutrient cycling and biological control are two of the regulating ecosystem processes that contribute to water being clean and safe for consumption and for direct contact activities. These services tend usually to be supplemented by artificial treatment or purification. However, if the source quality is good, the level of treatment required to bring water to a standard suitable for human consumption is reduced along with the associated costs.[27] In a pristine natural environment, primary treatment using filtering may be all that is required.

The cost of treatment is included in the price of water, but as discussed, only municipal supplies to non-domestic consumers are priced and these water charges are fixed by local authorities at the beginning of each year. They reflect a combination of average annual treatment and distribution costs and not the varying quality of the source water supply and the amount of pre-treatment required.

Chapter 2 described how water quality for human consumption has shown continuing improvements due to better enforcement of legislation and investment in water services. Pollutants of various kinds impact on water quality. Dirt and organics (e.g. from decayed plant material) can be picked up by run-off. Bacteria and pathogens can be collected in surface or groundwater from human or animal waste. Agriculture can contribute phosphates or nitrates, both of which can be harmful to human health in high dosages, especially nitrates. Toxic pesticide residue can also find its way into water supplies. Mining and industrial activities can introduce other elements including toxic chemicals or heavy metals.

The effect of impurities in the natural environment is often rendered harmless by physical and ecosystem processes. Much suspended matter settles to the bottom of rivers or lakes while organic impurities are oxidised in the upper layers of the water column. Harmful bacteria are killed through changes in environmental conditions, competition, predation and exposure to sunlight. Oxygen maintains these essential ecosystem functions and is introduced into rivers where they flow over weirs or rapids. By comparison, wetlands retain water in one place allowing much of the physical degradation of pollutants to be completed. Aquatic and wetlands ecosystems contain plant and animal species that break down, consume or absorb pollutants with particular roles being played by invertebrates and bacteria.

Water treatment, operating and capital costs

Once abstracted for human consumption water is typically treated to ensure the removal of harmful bacteria and or other substances. The quality if the output is defined by drinking water regulations and standards. Where the source water quality is of good quality, this process can be quite straightforward, but additional stages have to be applied where more contaminants are present. There additional stages may be necessary where pressures (i.e. point and diffuse pollution) exceed the assimilative capacity of the source ecosystem.

Successive stages of water treatment:

1) Screening
Water is passed through mesh screens to remove debris and particulate matter.

2) Aeration

[27] The cost of artificially treating water is not high by volume, but the construction of the associated infrastructure of treatment plant requires varying levels of investment.

The water is brought into contact with air sometimes through use of a cascading structure or by being passed through a forced flow of air. This removes some unpleasant dissolved gases and helps to oxidise some metal salts such as iron and manganese.

3) Coagulation

Coagulants, for example, alum or ferrous salts, are added to combine with suspended matter to accelerate settlement. If the pH is too high, acidic substances may need to be added to encourage coagulation, although the pH of the water leaving the plant must subsequently be neutralised to minimise any corrosion of pipes. Flocculation is the process by which the water is mixed with the coagulant.

4) Clarification

Clarification is achieved through a process of floc settlement in upward flow sedimentation tanks. A sludge settles towards the bottom of the tank Dissolved air flotation is sometimes used as an alternative to sedimentation whereby tiny air bubbles are used to carry floc substances to the surface.

5) Filtration

Filtration is achieved through the downward passage of water through sand filled settlement tanks. As particles are removed the filter becomes clogged and must be cleaned by the physical scraping off of material. Slow sand filtration has typically been used with flow rates of 0.1-0.25 metres per hour.

Rapid sand filtration is an alternative where volumes require faster flow-through or where coagulation or sedimentation has already occurred. Filtration rates of 5 to 7.5 metres per hour are common. Sediments may be removed by periodic backwashing or upward flushing of the tank. Pressure or gravity systems are used. However, rapid systems can be less effective than slow systems at removing bacteria. Activated carbon may be added to speed up the process or to help remove particular problematic substances. Activated carbon may be added to speed up the process or to help remove trace organic compounds or toxic compounds or metals. These substances are absorbed by the carbon until it becomes completely saturated at which stage it must be replaced.

6) Disinfection

Chlorine is normally used in disinfection to kill off remaining micro-organisms. If these have been removed by earlier processes or if the source water is clean, then this stage can be minimal. However, some chlorine is added to ensure that not re-infection of the water supply occurs in the distribution network. Indeed, the length or poor state of many distribution networks is often the primary consideration in the amount of chlorine to be added.

If too high a level of chlorine is added, for instance to deal with higher levels of contamination, this can combine with organic substances to form the by-products trihalomethanes. These substances have been identified as potentially carcinogenic. Consequently, a balance needs to be struck between levels of chlorination sufficient to kill bacteria and levels that could give rise to secondary contamination. Ultra violet radiation or ozone gas in combination with activated carbon are occasionally used as an alternative, but a level of chlorine will still be needed to maintain water quality through the piped network.

Source: Hoboken (2012)

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A lower-bound indication of the value of the regulating ecosystem service can be provided by an avoided cost approach. A regularly cited example is the protection of the forested Catskill Mountain watershed in upstate New York. This is the source of New York's water supply and its conservation has allowed the city to avoid the cost of constructing a $6 billion water filtration plant (hitherto at least) (de Groot et al., 2002). Similar approaches are being adopted in north-west England where a sustainable water catchment management plan (ScAMP) has involved expenditure of £10.6m (€12.5m) (2005-2010) on conservation activities to secure a wide range of environmental benefits, including water quality. A particular focus has been on reducing discolouration arising from organic matter than could result in excess level of trihalmethanes with potential health implications and liability costs.

It is possible that SCaMP's impact on water colour could lead to delayed costs, or perhaps even avoided costs, for the upgrading of some treatment works. Current data show some signs of stabilisation in water colour in restored plots compared with continued deterioration elsewhere. However there is very high uncertainty whether any water quality improvement will be detectable on the catchment scale and so it is not yet possible to accurately determine the expected level of cost savings. Savings on plant upgrades could potentially be significant, but are impossible to quantify at this time due to the interconnected nature of the water supply infrastructure. However, the avoided water treatment cost for one of the SCaMP peat restoration projects in the Peak District has been estimated at £1m to £2m per year (Cornell, 2011). Significant benefits of carbon sequestration may also be realised (Worrall et al., 2009)

The catchment approach highlights the potential value of ecosystem services, but also the difficulty of isolating the value of individual services within the wider catchment. Treatment costs provide only a partial reflection of the value of aquatic ecosystem services in that terrestrial services are also involved in the delivery of good water quality. In principle, a minimum estimate of the replacement value of all the natural ecosystem services is supplied by the varying cost of the treatment process as it relates to source supplies of varying quality. However, as discussed earlier, these cost based approaches do not indicate the true value of water quality or society's maximum WTP for the resource. Water treatment is of critical importance, but the variable costs by volume of water treated and in relation to varying water quality are low. For example, the Staleen treatment plant in County Lough (capacity 32,000m^3/day) treats water at a total operating cost of 8.69c per m^3 of which chemicals contribute 3.37c and sludge removal costs 1.54c.[28] Data on water samples and chemical application for the plant are patchy, but do show evidence of a relationship between periods with high turbidity and high suspended solids for late autumn, a period influenced by high run off from degraded peatlands upstream and the relative seasonal inactivity in the aquatic ecosystem.

Capital investment in treatment plant is a more significant cost as was recognised by the CDM study. EU regulations have required significant new investment in water and waste water treatment. The Water Services Investment Programme (WSIP) commenced in 2000 and provides for major investments (individually over €1 million) in water supply and wastewater infrastructure. The WSIP (2010-2012) expenditure plan included contracts and projects in progress (about €1 billion), contracts to be progressed to construction (€1.8 billion) in 2010-12 and schemes and projects at the planning stage. The €1.8 billion was divided between wastewater projects (56%) and water supply projects (44%). This compares with an allocation of two-thirds of the budget to wastewater infrastructure in previous investment plans.[29]

An illustration of a typical scheme is presented by the treatment plant element of the Arklow Water Supply Scheme costing €12.5 million to service a population of around 13,000 (capacity 18,000). At a discount rate of 6.67%[30] over 30 years, this equates to a net present value of €900 per resident or €32 per resident per year. However, it cannot be assumed that this level of investment will be typical for future years as much of the investment must fill the environmental infrastructure deficit arising from inadequate investment in previous years.

Furthermore, water treatment plant is not designed to substitute for ecosystem services. For example, nitrogen can be either retained or removed by natural ecosystem processes, but drinking water standards require a level of removal relating to public health standards. Data on water treatment costs is rarely available to demonstrate the cost of removing any particular substance and are also specific to a particular plant, its age and technology. Information on UK water investment is presented in Table 3.3 and provides an indication of treatment costs related to particular aspects of water treatment. In this instance, the data relates to a drinking water quality programme which

[28] Figures courtesy of Louth County Council.
[29] The Water Services Investment Programme 2010-2012 is being rolled forward to incorporate 2013, and referred to as the Water Services Investment Programme 2010-2013. (DECLG Circular L 4 /12)
[30] As set for DBO medium to longer term projects by the National Development Finance Agency.

includes investment in treatment works to achieve new standards as well as improvement of existing works where there has been a deterioration in the quality of the water abstracted.

Table 3.3 **UK Water quality programme 2005-10: Water Treatment**

	Capital expenditure 2005-10 (£m) (% of total capital expenditure)	**Additional operating expenditure 2009 -2010 (£m/yr)** (% of total additional operating expenditure)
Nitrate reduction	£288 (42%)	£2.0 (34%)
Pesticide treatment	£73 (11%)	£6.0 (11%)
Total Water Treatment Expenditure	£689	£174

Source: Ofwat, Future water and sewerage charges 2005-10 Final determinations

A UK study (Pretty et al., 2002) identified an annual cost of £19 million (capital plus operational costs) to prevent the proliferation of algae.[31] This study is interesting as it explicitly identifies the cost arising from the increase in the rate of production and accumulation of nutrients in excess of what the ecosystem can normally processes (Díaz et al., 2012). The study was based on water utility costs (1992-98) reported to the UK water regulator. It assumed that 10% of operating costs and 5% of capital costs were sensitive to algae (although no justification for these assumptions was provided). The estimates are particularly relevant to Ireland given that eutrophication is considered the main threat to Irish surface water (EPA, 2008). However, these estimates do demonstrate the limitations of using treatment costs to value the ecosystem services provided by clean rivers. The relevant cost here is really that of treating wastewater to meet the standards set by the WFD for natural ecosystems. The waste water standards and the costs of treatment are determined by the need to preserve the quality, including the assimilative capacity, of the receiving water body.

3.2.3 Avoided ill-health

Another means to measure the value of the ecosystem is assimilating waste is the avoided cost of ill-health. In most cases health problems arise from bacterial coliforms or cryptosporidium. Costs associated with the former are difficult to estimate because coliform contamination typically tends to result in low level illness, although this can involve recurrent intestinal problems and diarrhoea. Furthermore, much of this will be due to unsatisfactory preparation of food as much as to water. Nevertheless, the costs are substantial. Gastroenteritis from all sources, including food poisoning, has been estimated to cost the Irish economy at least €135 million per year (Rodrigues et al., 2007) before consideration of the personal welfare impact. Although often an inconvenience for healthy adults, symptoms can be severe in vulnerable population subsets such as young children or elderly people. While food preparation does play a part, the Food Safety Authority of Ireland has found that there is a serious potential risk to health from poor water quality. It provides the example of 18 cases of VTEC (*E-coli* 0157) contamination in West Limerick in an outbreak in 2005 during which two children required hospitalisation.

No cost estimates are available for trihalomethanes (THM), although the substances are of concern given experimental evidence of the link between various THM such as chloroform and bladder or colon cancer. Serious illness can also arise from *Cryptosporidium* of which there were over 400 cases following contamination of water supplies in Galway in 2007 (Garvey and McKeown, 2004). In the most serious *Cryptosporidium* outbreak to date in Wisconsin in 1993, nearly half a million people were affected of whom 100 died. Per person costs have been estimated by Corso et al (2003)

[31] A similar study in the US (Dodd et al, 2008) estimated that $813 million is spent on treating drinking water due to eutrophication. Due to lack of comparable data on Individual treatment plant costs this is simply based on annual bottled water purchases attributed to taste and odour problems.

at \$116 (€87), \$475 (€356) and \$7,808 (€5,856) for mild, moderate and severe symptoms respectively based on productivity losses, medical care and hospitalisation. Corso et al defined "severe" cases as ones requiring hospitalisation, although actual costs could vary considerably depending on the length of stay. On average in a *Cryptosporidium* outbreak one third of cases require hospitalisation (Garvey and McKeown, 2007). Indeed, in the Galway incident this occurred for 29% of cases (180) identified by the Emergency Department (Collins et al., 2013). On the basis of Corso et al definition this could imply a cost of at least €400,000, although the treatment cost component of this figure would have been lower in that the particular *Cryptosporidium* parasite in Wisconsin was a more virulent (C. hominis) type than they which occurred in Galway (C. parvum).

However, estimating the avoided cost of ill-health as an indication of the value of ecosystem services has the limitation that these estimates rather demonstrate the benefits of supplementary water treatment and catchment management at least as much as baseline environmental quality. Contamination by bacteria or *Cryptosporidium* is typically introduced by human or livestock waste and its removal in the natural environment may be due as much to physical processes (e.g. light) as to the biota.

3.3 Regulating Ecosystem Services

Water treatment costs at source provide for only a weak indication of the implications of the failure of the ecosystem to sustain good water quality. A superior expression of the value of aquatic ecosystem services is provided by our willingness to maintain it as a source of water supply, as habitat, for purposes of recreation and amenity, and also out of a sense of moral responsibility. This in turn is partially reflected in the sums spent, i.e. our WTP, to protect environmental water quality through waste water treatment and environmental management. In practice, the guidelines and criteria are set by legislation, but ultimately these have to be democratically acceptable. This acceptability will be tested by applications for derogation due to disproportionate costs.

A good quality environment sustains regulating ecosystem services which benefit water quality. Much of the breakdown in contaminants occurs through physical exposure to air and sunlight, but the assimilative capacity of the aquatic ecosystem has an important role in removing these. This assimilative capacity has its limits as discussed above. The freshwater pearl mussel (*Margaritifera*) may be valued as a rare species deserving of protection (there are no economic estimates), but the benefits of its filtering effect on water quality are largely negated by its association with low nutrient status rivers. This illustrates the limits of an ecosystem services approach in that many species, including species that are now more scarce, have evolved in a low nutrient environment where their regulating service value may not be critical from a human perspective (i.e. in terms of removing waste). Once this high quality status is lost, the ecosystem is not eliminated, but much of the biodiversity associated this high quality is lost too. Many of these species are those which are most valued by society, for example salmon, otters and many types of birds.

Many of the regulating services relevant to clean water are associated with the terrestrial ecosystem. Surface vegetation, including river margins and riparian woodland, captures many of the excess nutrients before they can reach rivers and lakes. Consequently, catchment management extends to terrestrial ecosystems and land use.

3.3.1 Water quality and waste assimilation

Rivers

Rivers provide an ecosystem service in their capacity to accept and assimilate waste. The principal pressures arise from organic matter (plant, animal and microbial organic waste), sediment (mainly fine inorganic material from soil erosion) and excess nutrients, namely phosphorus and nitrogen.

- Organic matter depletes oxygen as it decays, a process measured by Biological Oxygen Demand (BOD). The principal sources of organic matter are municipal and domestic wastewater, industrial waste, decayed plant material and slurry. There have been sharp reductions in levels of organic matter in wastewater across Europe in recent decades due to improved treatment (EEA, 2012).

- High levels of fine sediment tend to be associated organic run-off from agriculture. Silt can smoother gravels on the river bed that are home for many invertebrates or are used by fish for spawning. Organic matter tends also to combine with this sediment.

- Phosphorus and nitrogen are nutrients whose limited availability restricts biological activity in the natural environment. The quantities typically released from waste water, domestic waste facilities and agriculture are often well in excess of those found under natural conditions. Their increased availability is the principal cause of eutrophication as excess algal and plant growth is stimulated which then depletes oxygen due to respiration or decay. The level of nutrient inflow from municipal waste has declined in recent times as new treatment facilities have been installed. However, the widespread use of slurry spreading in Ireland remains a persistent source of nutrients. Donohue et al (2006) argue that catchments with more than 69% agriculture are unlikely to meet the requirements of the WFD without improved pollution controls.

Despite these pressures, rivers often display a remarkable ability to assimilate waste, providing a sink for anthropogenic activity. Indeed, waste water treatment plants rely on this ecosystem service without which permissible effluent standards would need to be much tighter. Where the receiving waters are vulnerable to eutrophication, treatment is required to reduce phosphorus to 2 mg/litre and nitrogen to 10-15 mg/litre. In the presence of freely available dissolved oxygen, aerobic heterotrophic micro-organisms provide rivers with a self-cleaning ability. Oxidation of organic matter is greater in cool temperatures, but the oxygen rapidly diffuses from water. Consequently, the process is most efficient where oxygen levels can be sustained, for instance in the well-aerated conditions permitted by cascades typical of the upper stretches of rivers. On the other hand, slower moving water downstream allows more time for oxidisation especially where macrophytes are present to provide photosynthesis or act as a habitat for micro-organisms. The reduction in oxygen follows a characteristic 'sag curve' with oxygen levels falling due to biological demand before rising once again. De-oxygenation can therefore occur some distance from the source of pollution, but these poorer quality stretches of water can then be replaced by improved conditions downstream. Weirs in these locations can assist in raising oxygen levels (Gray, 2004).

Following pollution from a point source, a river will pass through stages of degradation, decomposition and recovery before returning to clear water (Gray, 2004). Much of the breakdown of organic matter at source is performed by bacteria with phosphate and ammonia as by-products. Their activity is then replaced by algae which provide food for grazers and crustaceans before the algae dominance is replaced by macrophytes. Much of the heavy lifting is performed by benthic micro-organisms attached to the river bed or in the substrata below the river bed or behind the river banks (Lewandowski et al., 2011). One of the reasons that there is so little information on the self-cleaning capacity of rivers is that these microbial functions and species are so little understood.

In principle, a linear relationship exists between the EPA's Q-value, which provides a biological classification of water quality, and levels of phosphate and nitrate. Where the Q value is very high (i.e. 4 or 5) there tends to be abundant macroinvertebrates and a low supply of nutrients. Where the Q-value is very low (i.e. 1 or 2) levels of the nutrients are high. In the latter circumstances water is dominated by algal growth, most deleteriously the filamentous algae *Cladophora* which tends to smoother the river bed (Donohue et al., 2006). Recent work undertaken for the IMPACT project led by UCC has revealed that, between the extremes, the relationship between phosphate, nitrate and eutrophication is much less predictable. A principal reason appears to be grazing by invertebrates, in particular the *Baetis* mayfly. However, the abundance of these invertebrates is decided by contradictory factors, namely habitat suitability and food supply. Conditions suitable for the invertebrate population include overhead shade, but the population performs well where there is no shade corresponding to high level of algal growth (as measured by levels of chlorophyll A). As the algae can smoother the gravel bed required by the invertebrates, the local balance between invertebrate grazing and algal growth is determined by a variety of factors (Sturt et al., 2013). Some of these factors are unknown, but it is known that fish will eat invertebrates and that slight to moderate eutrophic conditions will favour coarse fish species and trout rather than salmon. Another factor that displays a noticeable relationship with reduced algal dominance is periodic flushing due to spate conditions.[32] Fortunately (at least in this respect) the Irish climate is characterised by frequent downpours

The invertebrate population therefore provides an important ecosystem service in reducing eutrophication to levels below those that might be expected based on the phosphate and nitrate content alone. *Baetis* are quite robust, but the mayfly population, along with other grazers such as larger caddis larvae, snails, crayfish and fish, is vulnerable to environmental impacts such as toxic substances and low oxygen levels arising from excess organic material, especially if an event occurs at the wrong time in the mayfly life-cycle that allows the *Cladophora* algae to gain a critical mass (Sturt et al., 2013).

Another potentially significant impact on the ecosystem and its ability to "self-clean" is the dredging of channels for agricultural drainage or flood relief. This leads to the rapid smothering of the gravel bed required by invertebrates and salmonids and its replacement by a cover of fine silt suitable for algal growth. The Mulkear LIFE (www.mulkearlife.ie), an EU funded LIFI Nature project on the Lower Shannon SAC has been working to restore the ecosystem of much of the Mulkear catchment and its salmon population following damage due to drainage dating back to the mid-nineteenth century. The work has involved the construction of over 25 rubble mats on the river bed. These have been used to create artificial riffles where the natural combination of turbulence and pool areas had been replaced by continuous uniform glides which were unable to sustain a large juvenile salmon stock even if food supplies were available. The rubble mats reduce the cross-sectional area of the river thereby increasing flow velocities at low summer flows. The faster flowing area on top of the rubble mat is quickly colonised by a considerable variety of invertebrates that favour such conditions. In the absence of riffles, a channel reach cannot sustain In the space of two years, the average salmon parr density has tripled along with improvements to the overall biodiversity of the river, including also increased numbers of sea lamprey, The five year programme of work, has invested hundreds of thousands of euro in habitat restoration work.[33]

The Mulkear river provides a suitable test bed for restoration. The water quality for the most part is high and the salmon population had survived the drainage work due to the undamaged downstream environment. It is less clear whether the same restoration would be effective for a river with inferior water quality. In these circumstances, catchment management, wide riparian buffers and changes in agricultural practice are likely to be more suitable. The Lough Derg and Lough Ree Catchment Monitoring and Management Project provides a pilot in this respect (Environ, 2008).

Lakes

By comparison with rivers, lakes represent a more stable environment. Consequently, they are more vulnerable to a rise in nutrients and not suitable for a high waste sink ecosystem service. The stable environment means that microorganisms tend to be found in layers within the water column rather than on the bed or shore. These microbes are responsible for removing up to a third of nitrate through dentrification (Kuznetsov, 1959). In contrast to rivers, much of the grazing is performed by zooplankton. In the absence of a significant nutrient inflow, a balance exists between algae, zooplankton and the fish population for which it is prey (in particular roach). The introduced zebra

[32] Simon. Harrison, UCC (*pers comm.*)
[33] Ruairi O'Conchúir, Mulkear LIFE (*pers comm.*)

mussel population has also imposed its own imprint on lakes, raising water quality at the expense of algae but also zooplankton and the diversity of fish species. The ultimate consequence of their invasion remains uncertain. As lakes do not incur the period flushing experienced by rivers they have greater vulnerability to organic and nutrient impacts. Scheffer (1998).has described how a change in environmental conditions can shift the lake ecosystem to another locally stable equilibrium or alternative stable state In these circumstances, it could be difficult to restore good water status.

Nutrients are not necessary bad. Rather their impact depends on the baseline environment. Amongst the richest environments are naturally eutrophic lakes which support abundant macrophytes. The macrophytes in turn provide a habitat and protection for zooplankton whose grazing keeps the water clear despite the presence of the nutrients (the balance of this relationship therefore resembles that for invertebrates in rivers). Even where the water is less than transparent, the biomass of cyprinid fish species such as minnows can exceed that of salmonids which prefer clearer waters (Mason, 2002). Problems arise for naturally eutrophic lakes when the nutrient pollution tips the balance in favour of algal growth. Macrophyte diversity can increase with greater phosphate availability whereas invertebrate populations seem to respond to nitrate due to the grazing opportunity that plant and algal growth provides (Harrison, 2013). If conditions veer towards those that preferentially support algae, the rapid depletion of oxygen that results leads to anoxic conditions and a loss of biodiversity.

Wetlands

Wetlands, are also a relatively stable environment in comparison with rivers, but can perform a useful ecosystem service through waste assimilation. The slower flow of water allows more time for the nutrient uptake. The processes are somewhat different from rivers with an additional role for standing plants. Phosphates and nitrates are absorbed by microorganisms living on plant surfaces. Microbial populations also break down certain heavy metals, albeit at the expense of changes in the microbial population and oxygenation.[34] Some plants also appear to be good at accumulating heavy metals, including water parsley (dropwort), duckweed, sedges and bulrushes. Many of these plants along with their microbial populations also provide rapid and almost complete reductions in *E-coli*, *Enterococci* and *Salmonella* for which rates of removal have been estimated by Kulzer (1990) and Karim et al (2008) amongst others. Adams (undated), however, argues that microbes living in the surface water layers can remove ten times more nitrates than plants. The nitrates are assimilated into ammonia (NH_3 or NH_4) and then released to the atmosphere.

Wetlands with different species and different water inflows and regimes vary in their capacity to remove nutrients and pollutants. Typical removal rates have been estimated at 75%-93% for suspended solids, 30%-50% for phosphates and 75%-95% for nitrates (Denison and Tilton, 1998). The quality and volume of the source water is the most important factor. Other important factors are outflow ratio, temperature, pH and contact or retention time. Of the last of these, it has been estimated that one week is sufficient for particulates to settle with two weeks or more being needed for the removal of phosphates (Adams, undated). On Adam's rough approximation, a wetland area of 1%-2.5% of that of the watershed would be needed with a larger wetland clearly being necessary if the area is fed by a stream draining an area of more intensive farming. Ideally, Adams claims, 50% of the wetland should be 2.5-15cm deep, 25% 15-30cm deep and with the remainder up to 90cm deep. The various depths permit slight differences in temperature and provide different habitats for plants and microbes. The analysis was undertaken on North America. Quite different rates of assimilation could occur in Ireland.

Breaux et al (1995) note that wetlands allow water to be treated to tertiary level in comparison with the secondary level applied by most artificial systems. They also argue that effluent treatment

[34] Nicolas Crispin (*pers comm.*)

through wetlands can be less expensive than standard sewage works. On the basis of this argument, the equilibrium between costs and benefits is shifted to a lower cost, higher quality environmental outcome. For these reasons, constructed wetlands are becoming popular for storm and wastewater treatment (Babatunde et al., 2008). The removal of pollutants is more assured than for natural wetlands because of the deliberate planning for physical, chemical and biological processes and the maintenance of oxygen supply. Constructed wetlands can also provide a dividend for biodiversity as demonstrated by the wetland created in Tolka Valley Park in Dublin (OPENFIELD, 2008).

Constructed wetlands permit the faster removal of nitrates and phosphates. Coliform removal of between 95% and 99.9% is possible (Ottoval et al., 1997). However, rates of removal vary considerably by location depending on scale and the level of nutrient inflow relative to flow and temperature. In Tribodaux, Louisiana, Breaux et al (1995) estimate reductions of 72%-85% in nitrates and 31%-76% in phosphates in the 1600m length of wetland and alluvial woodland below the town's sewage outfall. The savings are estimated at $785-$885 (€588-€664) per acre. To avoid health and ecological risks the state of Louisiana still requires UV treatment without which Breaux et al estimate the net value would be $1400-$1500 (€1050-€1125) per acre. Even in the Southern US, the efficiency of systems varies considerably. In two other examples, Breaux et al estimate savings of up to $9,635 and even $34,700 per acre. Depending on the level of nutrients in the inflow, the reductions and values could be quite different in the cooler Irish climate. There are also concerns over the functionality of such wetlands in Irish winter conditions given that the nutrient supply inflow remains undiminished in contrast to many natural wetlands.[35]

If degraded, natural wetlands can become exporters of nutrients as plants decay. Just as a healthy stable environment can provide regulating ecosystem services, so the degradation of ecosystems can leave natural systems more vulnerable. For example, in their natural state, peatlands are saturated such that water flows quickly over the surface. However, once degraded by peat extraction or forestry, organic matter is transported in runoff. The dissolved organic carbon (DOC) is often evident in the discoloration of water supplies, but the main problem arises from the combination of this organic matter with chlorinated water. The by-product of trihalomethanes has been identified as a serious health hazard. In Ireland, raw water quality data for the Staleen treatment plant in County Louth reveals a distinct peak for organic carbon and turbidity each January (O'Callaghan *pers comm.*). Frequently, this organic matter coincides with high coliform levels too. Contamination is first treated by chlorination, but the process can be imperfect given that the bacteria is somewhat protected from the treatment by the presence of the suspended material. The higher doses of chlorine then required to finish off the bacteria can lead to the appearance of trihalomethanes. The risk is being taken seriously by water companies in the UK causing them to invest significant sums in peatland restoration and catchment management. The reintroduction of saturated conditions helps to restore sphagnum mosses and other vegetation that can then perform an ecosystem service by recolonising worked bogs or other areas at risk of erosion.

Summary – waste assimilation

The capacity of aquatic ecosystems to assimilate waste diminishes as bacteria or algae deplete the level of dissolved oxygen. From the prospective of social values the proliferation of algae or the depletion of oxygen level is a bad thing. Many of the ecosystem services we value, i.e. angling, swimming and passive amenity, are lost in such an environment. Increases in nutrients cause rivers and lakes to often become choked with vegetation affecting both their appearance and navigation. The cyprinid fish biomass may be higher, but they are not valued as highly as salmonids they replace. Where nutrient levels are high, filamentous algae and sewage fungus (a slime consisting of bacteria, fungi, protozoa and some algae) smoothers the river bed preventing successful fish spawning. Once this growth becomes dominant, the macrophyte population diminishes, water clarity is much reduced and the water appears green and unsightly. The filters in water treatment plants are clogged more frequently adding to the costs and the algae blooms that can follow are

[35] John Lucey, EPA (*pers comm.*.).

toxic for fish, animals and human beings. Impacts are most serious is the stable ecosystem of lakes, but can be hard to reverse in many cases especially once alternative ecological states become established.

3.3.2 Natural hazard regulation

For rivers and wetlands, the term "natural hazard regulation" is often used to describe the ecosystem service of flow moderation by which aquatic ecosystems reduce the incidence of flash floods or aerial flooding. The wetland vegetation interferes with surface flow and, in the process, also provides for water storage, groundwater recharge and silt deposition. Wetland creation is now being considered as a "soft engineering" contribution to flood mitigation at least for smaller catchments.[36] A double-dividend is claimed due to the additional benefit of habitat creation for which reason wetlands have been a popular use of funds in North American offsetting programmes (Brouwer et al., 2009a). On the other hand, the saturated ground and hydrology of wetlands limit their capacity to eliminate the risk of more extreme flood events. Indeed, wetlands located in headwater locations could actually increase run-off (Burt, 1995). The hazard regulation service may be more effective downstream in the form of the natural topography of low lying flood plains irrespective of the presence of wetland habitat (Williams et al., 2012). Restoration of flood plains for this purpose is being actively adopted in the UK's Making Space for Water strategy (Defra, 2004). There is, though, some evidence that allowing agricultural areas to flood more regularly, particularly in summer, could lead to problems of nutrient mobilisation (Banach, et al 2009).

Peatlands are not an effective buffer. In their pristine state they behave like a sponge in absorbing water. The ecosystem service is limited by the fact that most are saturated already. Ironically, more buffering may occur with worked cutaway peatlands. However, this depends on the balance between the drier bog surface and the extent of the remaining surface *acrotelm* of sphagnum moss as well as the rate of run-off via drainage ditches (David and Ledger, 1988; Holden et al., 2008; Grayson et al., 2010). Consequently, the regulating ecosystem service of peatlands for flooding is indeterminate and probably location specific.

Wetland plants could be impacted by pollution, but for this report it has not been possible to identify publications on the particular role of plant species in reducing run-off. However, native riparian woodland is vulnerable to clearance and does provide a regulating ecosystem service by moderating run-off such that the more constant base flow can represent about 60% of total flow compared with 20% in non-forested catchments (Kilfeather, 2000; Neary et al., 2008). Riparian woodland also provides for physical interference with river flow by increasing the *roughness* of channels holding back river flow by 15%-70% while the duration of peak flow is extended and moderated by 20-140 minutes (Thomas and Nisbet, 2007). This characteristic could potentially cause problems upstream, but moderates the response downstream where larger settlements are typically found. Along with other riverside vegetation there are also benefits from reduced bankside erosion including of productive agricultural land.

The value of the ecosystem services is therefore location and context specific. For moderation of run-off, the benefit can be measured in terms of damage avoided and is greater where urban areas are at risk rather than agricultural areas (OPW ref). As these tend to be downstream and impacted by high intensity events, the damage is greater than for higher frequency "hydrological" floods (Williams et al., 2012). For built up areas in the UK it has been estimated that a property with a 1% annual risk of flooding has an annual equivalent damage risk of £84 (€100) (Penning-Rowsell et al., 2010). However, most alluvial flooding affects agricultural land. In this case, the economic impact is less, although it is worth noting that the value of more productive grazing land in Ireland exceeds €11,000 per acre. In vulnerable areas, damage to grazing land has been estimated at between €100 and €750 per hectare per flood event (Posthumus et al., 2009). Much of the damage is realised in terms of erosion, but as flooding is typically finite in duration, the impact is more significant for

[36] Mark Adamson OPW (*pers comm.*.)

summer flooding (which may be becoming more common with climate change) when fields are in production or being cut for silage. Reductions in erosion would not only preserve agricultural land, but also prevent the suffocation of bethnic flora and fauna with sediment. Hence, by reducing erosion, riverside vegetation also performs a service in preserving fish spawning grounds and rare species such as the pearl mussel.

Estimates of damage *costs avoided* represent a minimum measure of the benefit of flood avoidance as there can also be considerable disruption to peoples lives, especially to more vulnerable population groups. There have been a few studies of people's willingness-to-pay to avoid flood damage as an estimate of *defensive expenditure*, but these have revealed a tendency amongst respondents to underestimate the potential costs (Turnstall et al., 1994; Shabman and Stephenson, 1996).[37]

Alternatively, it is possible to estimate the *replacement cost* of a "hard engineered" flood mitigation measure to substitute for natural ecosystem service performed by wetlands or flood plains. However, this method may underestimate the total value of the natural ecosystem service or be unable to exactly replicate the benefit (Brouwer et al., 2009a).

Impacts on the moderating effect of natural ecosystems on run-off and flood moderation would follow principally from the removal of natural vegetation including for drainage and canalisation works. These works tend also to shift the flood risks downstream and release silt into the environment.

3.3.3 Groundwater recharge

Wetlands contribute to groundwater recharge especially where evaporation is low as in Ireland. However, this only occurs where the underlying geology is permeable. Most groundwater recharge in Ireland occurs from limestone bedrock and there are only small (2%) shallow areas of sand or gravel and very small areas of permanent wetland. Most of the relationship with wetlands occurs through the supply of water to wetlands from groundwater rather than the other way around, for example turloughs and fens such as at Pollardstown on the edge of the Curragh gravel bed, although this latter performs an ecosystem service in supplying the Grand Canal. Supplies for industry and agriculture would not necessarily be significantly reduced in the event of rivers drying out in that there is very low abstraction, There are, however, predictions of groundwater deficit in parts of Ireland in the event of projected climate change (Moe et al., 2007; Hunter Williams and Lee, 2008).

3.3.4 Climate regulation

Peatlands provide a regulating service in terms of carbon storage and sequestration. They are also home to some specialist plant species and important habitat for several breeding and wintering bird species. However, the sequestration function applies to less than 8% of raised bog and the 21% of blanket bog which remains intact (Foss et al., 2001). Even here it occurs at a modest level and fluctuates depending on rainfall and the level of the water table (Alm et al., 1999). Rather, peat cutting has rendered Irish bogs as a net source of emissions.

Instead, peatlands are more significant as a carbon store. The storage value is very difficult to quantify in current economic terms, although it would be possible to arrive at a rough present value estimate if assumptions are made of the future costs of climate change. The value of the sequestration service is discussed in Wilson (2010) and Bullock and Collier (2012) and is entirely related to the value of current mitigation of climate change. Typically, the value of sequestration has been related to the price of carbon allowances on the European Emissions Trading Scheme which, in principle, should reflect the future cost of climate change through the number of emissions allowances permitted by policy.

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[37] UCD (GPEP) are preparing a paper on the same for Bray and report similar results.

Other kinds of wetlands can be either a source of net emissions or sequestration. Cole et al (2007) have argued that the overall total amount of emissions from wetlands is twice that of the dissolved carbon captured by the water which then continues downriver to the sea. Under pristine conditions wetlands can act as a net receiver of carbon. For example, Natural England (2010) has estimated carbon sequestration by restored fen at 1.14 t CO_2-eq ha^{-1} (i.e. carbon dioxide equivalent). However, once disturbed, there is a strong shift to net emissions of between 1.57 to 2.85 t CO_2-eq ha^{-1} for lowland fen. In assessing the implications of climate change for the UK's terrestrial environments wetlands, Natural England (Alonso et al., 2012) provide little information on lowland wetlands that are neither raised bog or fen, referring only to a study by Dawson and Smith (2007) which reported improvements in sequestration of between 0.1 and 1 t C ha^{-1} once lowland wetlands are restored. In the interim, damage to wetlands would have an adverse impact on economic along with an associated economic cost in addition to the restoration cost incurred.

At the level of the wetland vegetation, the contribution of macrophytes and their share of surface area appears to have a significant influence on the balance between carbon sequestration and emissions. Along with the level of the watertable, the balance of emissions varies significantly depending on the corresponding vegetation mix (Wilson et al., 2009) and so is likely to be location specific. Net emissions are more likely where conditions are not anaerobic and there is an opportunity for vegetation to decay. In reviewing various studies, Wilson et al (2009) and Foss et al (Foss et al., 2001; 2009) report that lowland marshes are typically sources of GHG emissions in their natural state averaging 5.85 t CO_2-eq ha^{-1} of CO_2 and 1.05 t CO_2-eq $ha^{-1}$of CH_4 Methane (CH_4), in particular, is released in the littoral zone where vegetation contributes to emissions from microbes living in the saturated, but periodically exposed mud (Strack et al., 2004). Within a 100 year time relevant to climate change policy, methane has a significantly higher global warming potential (GWP) than CO_2 by a factor of 23:1 In the context of anthropogenic climate change these emissions therefore have a significant negative social value (Wilson, 2008; Bullock et al., 2012). Any disturbance that increases methane emissions in particular, e.g. impacts on hydrology that cause large fluctuations in the watertable, will have the more significant economic cost.

In principle, emissions trading provides an indication of the ecosystem service value of carbon sequestration (and the benefits of climate change avoidance) as reflected in climate change policy. In practice, though, prices depend on the artificial supply of allowances and market sentiment as to the effectiveness of future climate negotiations. In 2008, prices had been almost €30 per tonne, but have since fallen back to just €2.81 per tonne largely due to an oversupply of allowances in the context of the international recession rather than the long-term discounted cost of climate change. These prices should recover in time and an average figure of €20 has typically been used in calculations. However, a more accurate valuation of the economic and social benefit would be in terms of the direct benefit of avoided climate change damage in the future. In the absence of any such estimate the UK (DECC, 2009) has recommended that carbon be valued in terms of the national abatement cost of emissions reduction. At present, this implies a cost of £50 (€50) per tonne of CO_2 equivalent (CO_2-eq), a figure that is expected to rise to £70 (€82) per tonne by the 2030s as emission reduction commitments get tighter.

3.4 Supporting ecosystem services

By retaining water wetlands provide a supporting service for soil formation by capturing silt that contributes to the productivity of these lands for grazing (or cultivation in the case of flood plains). This could also be considered to be a regulating service in that it is associated with the moderation of run-off noted above. Unsurprisingly, the fertility of alluvial floodplains is significantly greater than non-alluvial flooded areas (Wheeler et al., 2009). A gradual depletion of nutrients occurs where wetlands are rain-fed and colonised by species-poor vegetation.

Rivers and wetlands are also important habitats for a variety of species including specialist mammals, birds, amphibians and invertebrates. Fish populations are sustained by a mix of open and shaded habitat and by the food supply. When wetland is present in a mosaic with riparian woodland and other habitats it enhances valuable salmonid habitat and has been shown to increase fry survival (Broadmeadow and Nisbet, 2004; Malcolm et al., 2008).[38] The value of ecotourism and angling is discussed below under the heading of cultural ecosystem services.

Rivers and wetlands are evidently a habitat for numerous species and a migratory destination for others. In principle, people value this service as a non-use good, but valuing species in economic terms is problematic. In a CVM exercise in which people were asked for their willingness-to-pay for the protection of endangered wetland species, Splash (2000) found that a large proportion of respondents held a 'rights based' position (i.e. believed in a species' right to protection). This resulted either in protest zero bids (i.e. respondents protesting at the exercise and bidding zero) or arbitrary positive bids that were difficult Furthermore, the public's perception of the link between species and associated habitat could well be tenuous and difficult to distinguish from the influence of familiar parameters of preference or from income. Higher stated preference values are often attracted to so-called charismatic species. It might be possible to interpret these values as being representative of the entire habitat, but these are often species, i.e. birds of prey, that reside at the top of the food chain rather than those involved in more critical primary or secondary production.[39]

Consequently, environment economists shy away from asking people to value species in favour of valuing habitats, typically measures to protect such habitats. The context with which survey recipients are presented often refers to use values (cultural ecosystem services), indirect use values (mostly regulating services) and/or option values (that environmental goods should be protected for future possible direct or indirect use). Supporting ecosystem services are more likely to be presented in relation to survey recipients' non-use values, including existence values. However, very few surveys attempt to distinguish the share of willingness-to-pay that is represented by each of these values. This process tends to be problematic given the overlaps that exist between each in preference. The only wetland valuation undertaken in Ireland to date does approach a distinction between use, option and existence values, but only indirectly in relation to future scenarios and their relationship with these different components of Total Economic Value. This survey was applied to peatlands within the EPA Strive BOGLAND project (Bullock and Collier, 2011). A sample was taken from a belt of central Ireland across to Dublin, but was most concentrated in the Midland peat cutting area. Different subsets of respondents were asked to value a) an overall national policy of peatland protection, and to b) management scenarios relevant to a prospective peatlands national park in the Midlands. The scenarios included:

- a peatland restoration scenario (mainly a supporting ecosystem service, but with harvesting opportunities possibly assumed by some respondents),
- a wetland scenario of a mosaic of open water, reedbed, bog woodland and peatland (a supporting service backed by possible cultural services related to bridwatching, etc),,
- a wetland mosaic scenario with more emphasis on recreation (supporting and cultural),
- a wetland/open water/woodland scenario that placed more emphasis on tourism (mainly cultural ecosystem services).

The wetland scenario was ranked second of four by respondents after peatland restoration followed by the recreation scenario.

[38] See Broadmeadow and Nisbet (2004) and Malcolm et al (2008).
[39] The corncrake could be a wetland exception (although its main association is with traditional pasture).

Figure 3.1 BOGLAND Wetland scenarios. 1) wild wetland scenario and 2) tourism wetland scenario

Source: Bullock & Collier, 2001 (pictures by Brian Gallagher www.bdartcom)

In the event, the scenario of peatland restoration was most popular especially amongst local people. The tourism scenario was the second most popular choice, but the natural (wetland mosaic) scenario was the overall preferred option taking into account respondent's second and third preferences. An option of "do-nothing" was still preferred by 20.7% of respondents. Over 69% of respondents were willing to pay in principle for a peatlands park with their average willingness-to-pay (non-parametric approach) being €79 per person per year. A follow-up choice experiment identified a continued potential for household turf cutting as a major incentive for peatland restoration amongst the local population, but also amongst many non-locals too. This result occurred even though peat cutting of all kinds had been described (in brief) as incompatible with sustainable peatland management.

Other surveys that have set out to estimate habitat values include Bateman and Langford (1997) for the Norfolk Broads, Farber (1988) in Louisiana and Van den Berg et al (2001) in the Netherlands. The difficulty of asking respondents to value habitat directly, as a supporting ecosystem service, means that these studies have chosen instead to present management scenarios, for instance protection of the habitat from seawater inundation, as the context to the valuation.

Elsewhere, Brouwer et al (2009a) reviewed numerous wetland papers in their meta-analysis of water valuation surveys, finding that wetlands were typically valued more than rivers or lakes. Brander et al (2006b) specifically focused on wetlands in their meta-analysis of 190 international studies. Of the total number of wetland goods valued (390) in the Brander et al sample, the largest proportion of studies (19%) described wetlands as habitat and nursery areas while only a small minority (5%) sought to value wetland biodiversity specifically.

3.5 Cultural services

Angling and other forms of contact and non-contact water-based recreation (e.g. bathing, kayaking and birdwatching/nature viewing) are the main categories that fall under the heading of cultural ecosystem services. The CICES classification acknowledges that these activities can be more accurately described as environmental benefits arising from supporting and regulating ecosystem services such as clean water, habitat and biodiversity.

3.5.1 Kayaking, Sailing and Bird watching

An estimate of the value of whitewater kayaking has been provided by Hynes and Hanley (2006b) referenced above. Kayak touring, windsurfing and sailing also occurs to a modest degree on Irish rivers and lakes. It can be difficult to survey these interest groups given that not all participants are club members and therefore potentially contactable. High water quality is evidently important to all these activities, especially from a health perspective given the contact nature of the sports. However, it is not necessarily fundamental to any of these activities as Hynes and Hanley discovered. In this case, a significant willingness-to-pay did not extend beyond health considerations.

Wetlands, rivers and lakes such are also visited by birdwatchers. For this interest group, water quality is a much important consideration as it sustains the bird populations adapted to this habitat. The Wexford Slobs and Lough Neagh would be among the principal destinations for this activity in Ireland, although the former (though freshwater) attracts birds commonly associated with coastal environments. In this respect, there is a direct and obvious link between utility values and biodiversity or habitat quality. Birdwatch Ireland has 14,000 members while the Royal Society for the Protection of Birds has 11,000 members in the North of Ireland. However, membership numbers provide only partial evidence of the value of this activity given that bird and wildlife sitings enhance passive use value of the countryside generally.

3.5.2 Angling

There are 265 lake, river and stream habitats identified as being suitable for migratory salmon, trout and eel across the island if Ireland, These systems account for 132,275ha of lake habitat and 21,606ha of fluvial habitat of which 2,826ha is believed to be of high quality status (DCENR, 2008). The national salmon population has been in a long-term decline for many years. The independent scientific report by the Standing Scientific Committee for Salmon (IFI, 2011a) found that of national stocks in 2010, 60 had an identifiable surplus but 80 failed to meet their Conservation Limit and so were allocated no allowable catch. There is no firm evidence of the reasons for the decline which could relate to exogenous factors such as climate change, marine catches and predation or to local factors such as barriers (including hydroelectric dams), poor habitat or water quality. Of the last of these, the composition of a number of important salmon/trout lakes has been changing in response to the spread of the alien zebra mussel and the impact this species is having by increasing water clarity. Where lakes previously suffered from elevated nutrients, this could possibly benefit salmon as their productivity is impaired in waters of moderate to poor quality (IFI, 2011a). However, clarity and quality are not synonymous. Any benefits are interim and there is a serious risk that the zebra mussel could reduce the productivity of aquatic ecosystem.

Fish, and the habitat that supports them, provide a cultural ecosystem, service in the form of angling. Anglers were responsible for 63% of the estimated salmon catch of 32,279 in 2010 (IFI, 2011a) with 17% of the total having been fished from the River Moy and 10% from the Blackwater. Evidence of the value of angling participation and expenditure was available from the survey undertaken for the Marine Institute (Williams and Ryan, 2003) described earlier. Data just published by Inland Fisheries Ireland (2013) arrives at an estimate of participation of 252,000 for freshwater and sea angling combined. Nine per cent of the sample was represented by sea anglers fishing for bass, although the IFI acknowledges that the actual number of sea anglers could be greater once those fishing for other species are included. (the Marine Institute (Williams and Ryan, 2003) had estimated participation at 71,000 as of 2003). Of these anglers, salmon anglers who purchased licenses have been estimated to amount to approximately 6,458 (IFI, 2011b), of whom 2,395 caught and released fish with an average catch per angler over the year was 5.3.

Recent Failte Ireland statistics for 2011 report a decline in the proportion of angling holidays to 5% from 8% in 2007. The same decline is evident for anglers from overseas whose numbers in 2012 were estimated by Failte Ireland at 127,000 (from 113,000 in 1999) with about the same number coming from Northern Ireland (IFI, 2013). However, anglers from overseas tend to spend much more than domestic anglers with expenditure of €89 million estimated for 2010 (Failte Ireland, 2011) equating, according to the latest IFI survey (2013) to average per trip expenditure of up to €2,116 (over 9-11.5 days). By comparison, average expenditure per trip (1.4 days) by domestic anglers was estimated at €196. Direct expenditure (on tackle, permits, salmon licenses, gillies and boat hire, etc.) represented 49% of this average. Total aggregate expenditure on both freshwater and sea angling was estimated at between €371 and €497 million.

However, expenditure is only a partial representation of value. Welfare values seek to capture the full consumer surplus. In this respect, the survey of salmon anglers by Curtis (2002b) estimated travel costs of IR£68 (€86) and a sizeable total willingness-to-pay of IR£206 (€261) per day. The IFI survey (2013) also included a contingent valuation survey, in this case an average willingness-to-pay for the preservation of national fish stocks at €66.52 per year for anglers and €15.97 per year for the general population. Although pollution has a greater impact on salmonids, the Marine Institute (ibid) survey had noted that 27% of coarse species anglers were concerned with pollution compared with only 11% of game anglers. This difference probably derives from the relative distribution of the two activities with lower value coarse fishing being more widespread and salmon angling now confined to protected good quality rivers. Habitual coarse fishing destinations such as the Border Lakelands have suffered from poor water quality and nutrient pollution in recent years (EPA, 2010c).

Game fishing has been experiencing a recovery in some prime areas. According to the Indecon report (2003b), salmon angling was worth €11.3 million as of 2003 (compare this with the latest IFI estimates[40]). However, each rod-caught salmon was estimated to be worth €1,000 in direct tourism expenditure and more in terms of the wider economy. Each salmon caught on a premier angling river such as the River Moy in County Mayo was estimated to contribute €2,000-€8,000 to the capital value of riverside fishing rights, equivalent to as much as €500,000 per kilometre for the best sections of the river.[41] Although there are only a handful of salmon rivers that can claim capital values of this order, there are numerous rivers across Ireland with this potential.

3.5.3 Boating

Boating and cruising is important on a number of rivers such as the Shannon, Boyle, Erne and Barrow. Around 42,800 people are involved according to the Marine Institute (Williams and Ryan, 2003) report of whom 8,900 reported making at least one overnight trip or 59,500 overnights in total. Expenditure was estimated at €12.7 million plus an additional €4.2 million on equipment. Cruising holidays by foreign tourists appear to have declined in recent years. The number of tourists using hire boats has averaged 20,000 per year between 2000 and 2006 (Fáilte Ireland 2009). On the other hand, domestic cruising had risen in popularity prior to the current economic recession. More recent figures for 2006 compiled by Waterways Ireland reveal that Ireland's boat owners make around 15 trips per year (21-30 days) spending an average of £135 (€160) each time. The Shannon and Erne systems are by far the most popular destinations accounting for around 85% of trips (DIT, 2006). In terms of ecosystem services, good water and landscape quality, along with opportunities for wildlife viewing or angling, clearly contribute to the utility value of boating trips. However,

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[40] Note that this figure applies the survey expenditure data to the 7% of respondents in Milward Brown Lansdowne's monthly omnibus survey who have undertaken *any* kind of angling in the previous year. This approach is vulnerable to over-estimation given that many local angling trips involve zero expenditure.
[41] Figures are a mixture of those produced by Indecon (2003) and through personal communication with Dr. Declan Cooke of Inland Fisheries Ireland.

cruising and marinas can have an adverse impact on water quality due to oil seepage and on riverside habitat due to vessel wash.

Table 3.3 Domestic participation and value placed on angling and boating

	game angling	coarse angling	boating
participation	80,500 *	66,500	42,800
expenditure	≥ €12.5m	€4.8m	€16.9m
utility	≤ €287m	n/a	

Source: Marine Institute (2003) & Indecom ((2003). * IFI puts active number of salmon anglers at 6,458.

3.5.4 Bathing and swimming

High water quality is amongst the key criteria that public beaches must attain to be awarded international Blue Flag status (in Ireland this status is awarded by An Taisce). High water quality is also essential for the health of bathers. Most sampling of bathing water quality has been for coastal locations. Only nine inland locations were surveyed by the EPA (2012). All of these inland locations met 'sufficient' water quality standards and 67% met the 'good' water quality standard. However, bathing occurs at many more destinations than have been registered by local authorities for Blue Flag Programme. In addition, triathlon swimming has had a dramatic increase in popularity in recent years.

3.5.5 Passive amenity

River and lakeside passive recreation are of more importance than any one specialist activity. In addition, many people own their main home or a holiday property beside or near rivers and lakes. There are no figures for participation in river/lakeside amenity, but numbers are certain to be substantial. Many river and lake locations are inherent features of highly valued landscapes, for example the Killarney Lakes or the River Nore. This in turn attracts recreation, especially light or passive recreation, i.e. walking, family visits and picnicking. These activities provide very significant economic welfare benefits and sometimes significant economic spinoffs for local communities. Based on the data from the EPA BOGLAND final report (Renou-Wilson et al., 2011), the average number of visits per person per year to rivers and lakes is five with some individuals visiting as often as 30 times or more per year. Based on the survey results the breakdown by activity was as follows:

Table 3.5 Activities when visiting rivers or lakes

walk	picnic	birdwatch	water sports/swim	angling	other
52.6%	32.3%	11.9%	9.7%	17.4%	7.6%

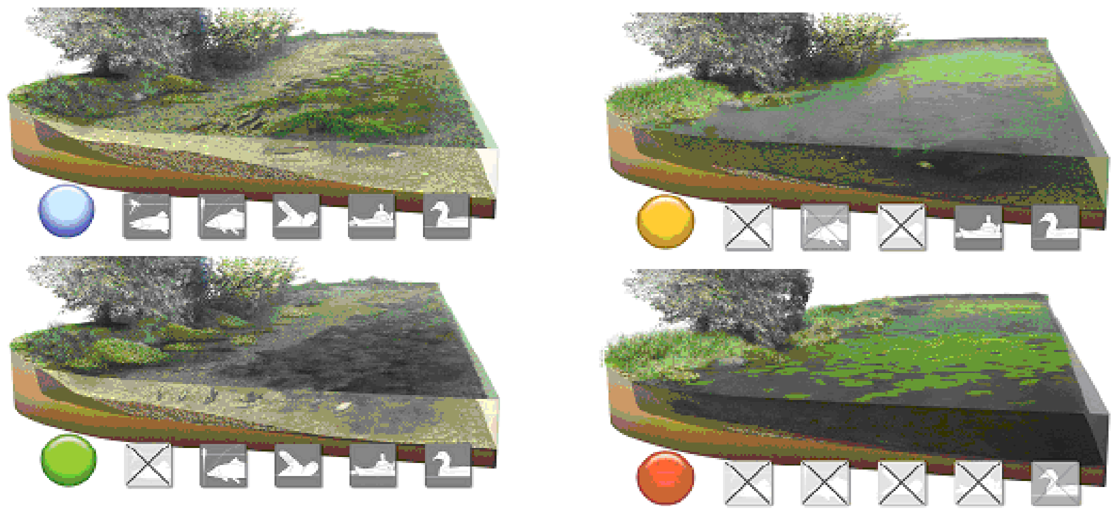
Measures of the amenity-related value of water, most specifically water quality, are evident from numerous studies. These include both revealed and stated preference studies of water related active and passive recreation. Brouwer et al (2009a) warn that the density of lakes in any location and the presence of substitution effects with water activities elsewhere or recreation opportunities of any kind should be taken into account. For Ireland, welfare value estimates are currently being compiled within another EPA Strive project, but evidence to date of passive values is limited to the Stithou et al (2011a) survey in relation to the Boyne and to the Norton et al (2012) benefit transfer exercise discussed in Chapter 2. The quality of the environment, including water quality, along with wildlife, are essential elements that enhance this experience and are often promoted along with landscape as an attraction for visitors.

Household stated preference surveys always have the challenge of adequately communicating environmental quality to a public with mixed awareness, experience or recollection of the good in question. To communicate the effect of water quality on environmental quality, Resources for the Future (Vaughan, 1986) developed the aforementioned water quality ladder which lists the amenities that can be undertaken at various levels of quality, for example swimming or boating. Although it has been widely used, particularly in North America, the use categories did not relate strongly with objective measures of water quality. In addition, subsequent studies have shown that, in practice, bathers have a higher tolerance of poor water quality than the ladder suggests. Brouwer et el (2009a) recommend that researchers ask bathers explicitly about the clarity of water in which they would prefer to bathe. To allow "the ladder" to conform more closely with the good status sought by the WFD, Hime et al (2009) have prepared a new ladder that relates use values more closely to ecological change for the purpose of linking public surveys and scientific definition of ecological quality. The revised ladder also indicates some of the ecosystem services at work. The ladder (Figure 3.2) has been prepared for a generic river with increasing levels of eutrophication. Quality categories are listed from Blue (highest quality), through Green to Yellow and Red (lowest quality) as defined by appearance (clarity), river vegetation, riverbank vegetation, fish species and the substrate likely to be found in each category based on guidance from Hatton-Ellis (2003), Holmes (1999) and the Environment Agency. Hime et al demonstrate how the revised water quality ladder can be used to provide more conformity in European valuations of water quality for the purpose of benefit transfer and of mapping spatial variation in WTP. Table 3.6 provides a summary (only) of these definitions for some key species at successive levels.

Table 3.6 Revised water quality ladder (Hime et al., 2009)

Blue (50% plant surface cover)	**Green (60%)**	**Yellow (70%)**	**Red (85%)**
No algae. Good clarity	Slight increase in turbidity	Green hue.	Brown hue. Algal mat covering substrate
Rhynchostegium riparoides. 60% Raunuculus penicillatus ssp (25%) Callitriche platycarpa and stagnalis (15%)	Apium nodiflorum L (50%) Rhynchostegium (50% riparoides (35%) Callitriche hamulata and stagnalis (15%)	Apium nodiflorum L (50%) Algae Cladopora (40%)	Algae Cladophora (40%)
Reed, water cress, sweet-grass. willow	Reed, water cress, sweet–grass, willow	Reed, willow	willow
Mostly game fish, some coarse (trout, minnow, chub)	Virtually no game fish, more coarse fish Bream, carp, perch, rudd	Less coarse fish Bream, carp, roach, rudd	No fish
BOD limit <4mgl Cat A & B Ammonia<0.6mgNl	BOD limit < 6mgl Cat C Ammonia < 1.3 mgNl	BOD limit < 8mgl Cat D Ammonia <2.5 mgNl	BOD > 8mgl Cat E & F Ammonia > 2.5 mgNl

Figure 3.2 Water quality ladder based on table above for public survey use
(Mick Posen © c/o Hime et al (2009) **P**ermission required before publication).



A limitation of the survey-based approach to ecosystem services valuation is that incidents of pollution will most often only be perceived by the wider public once they reach significant levels (i.e. the yellow or red categories). Protection against lower level pollution is only likely to attract a significant willingness-to-pay if the impact on aquatic wildlife is explained within the information accompanying a survey. Consequently, information provision has to be dealt with very carefully to avoid leading recipients into expressing high values that do not necessarily reflect the value levels that might typically attach to environmental quality.

3.6 Habitat offsets and banking for freshwater habitats

Chapter 1 described how habitat offsetting and banking is being used in countries such as the United States to extend conservation objectives or to provide no-net-loss or replacement environments with higher than equivalent levels of ecosystem services in the event of impacts to natural habitats. The Willamette Partnership provides details of one such initiative for wetlands, namely the Oregon Rapid Wetland Assessment Protocol (ORWAP) which provides acreage credits for new or replacement habitat based on an assessment of ecosystem functions (Adamus et al., 2009). The protocol is not monetary or economics based and currently provides no means to convert ecosystem functions into ecosystem service scores. However, it does include "value scores" that are the basis for credits that account for ecosystem services.

A total of sixteen functions are included in the assessment. These are grouped into five primary services:

Table 3.6 Services and ecosystem functions used for habitat offsets in ORWAP

Primary services	Ecosystem functions
Hydrologic	water storage or retention
Water quality	sediment retention and stabilisation, phosphorus retention, nitrate removal, thermo-regulation
Fish support	migratory fish, non-migratory fish
Amphibian, invertebrate and	aquatic habitat, waterbird feeding, waterbird nesting, organic matter

water bird habitat	export
Plants, pollinators, songbirds, raptors and mammals	terrestrial songbird, raptor and mammal support, native plant diversity and pollinator habitat
Public use and recognition	
Provisioning services	``
Wetland Ecological Condition	
Stressors	
Sensitivity	

The five functional groups are scored through the use of an Excel spreadsheet populated with macros defining relationships and interdependencies based on the results of ecological models, narrative criteria and best professional judgement. The group scores are based on two scores which are allocated to the 16 constituent ecosystem functions' effectiveness and value as measured by 140 indicators. The Partnership argues that the mathematics behind the indicators have been carefully researched, but that completion of the spreadsheet by users is fairly straightforward The group scores are equal to the maximum value of any of the functions and fall between 0 and 10 points. In addition, scoring is also allocated to information on 'ecosystem condition' (defined primarily by vegetation composition including functional integrity and native species diversity), and to provisioning services, public use and recognition (i.e. designation), sensitivity and stressors.

The spreadsheet requires the insertion of data from various sources including:

- Spatial data on the wetland and other land cover drawn from aerial photographs or maps = 49 indicators including enclosure by roads and distance to roads (e.g. affecting animal movement), forest landscape extent, type and proximity of natural land cover, other water features and locations, tidal proximity, wetland's uniqueness in the catchment, hydrologic connectivity, designation, fish, waterbird, plant and other species concern, upstream storage, downstream flood vulnerability, known water quality issues downstream, phosphorus and nitrate loadings, salinity, etc
- Wetland characteristics = 81 indicators including type, hydrology/seasonality and saturation, depth, water sources, vegetated zone width, submerged and floating vegetation, non-native plant species, bankside vegetation, surrounding vegetation, fish access, etc.
- Water regime = 9 indicators including factors affecting water regime, factors contributing nutrients or contaminants, factors affecting site's soils, vegetation removal from site including grazing, etc.

While "values", as defined by the ORWAP model, include ecosystem services, they also emphasise benefits to protected species (although this protection has been socially determined). The values relating to clean water supply and other regulating and cultural services are not explicitly defined except for some reference to flood mitigation, carbon sequestration and fisheries. It is noted that while values are dependent on ecosystem functions, the two can perform independently of one another. For example, a wetland that can be good for water retention may have a rather modest ecosystem value if it is located in a catchment without large numbers of properties that are vulnerable to flooding. The ORWAP Manual makes the point that functional opportunities are often located upstream while the maximum ecosystem service values, for example from flood mitigation, may be downstream. The system is claimed to perform reasonably consistently for different users, although it is acknowledged that 'ecosystem condition' and stressors can be poor predictors of ecosystem functions and values. It is also acknowledged that the indicator macros contain a good degree of standardisation and must be informed and verified by the user with the support of local fieldwork.

Having filled in the spreadsheet, the next stage is to determine whether two areas are eligible for a "trade" and requires the ORWAP Effectiveness scores of the debit and crediting areas are then multiplied by their wetland acreage. Trading ratios are applied to both locations to account for risk, time and other factors.

Naturally, ORWAP describes scenarios for the types of wetland and species found in the Willamette Valley and north-west United States. The method, but not the scores may be applicable to Ireland. Chapter 1 noted that Defra is piloting a simple scheme in the UK largely to extend conservation beyond designated nature reserves. There is good potential to use offsetting and banking for the same ends in Ireland, including in relation to potential impacts addressed by the ELD. There is no monetary factor included here, but a recognition of ecosystem services values that could possibly be extended from those included in the ORWAP example to include more regulatory and cultural services.

3.7 Summary - Freshwater Ecosystems

Water resources are valued for direct consumption, other use values, indirect use and non-use values. The quality of water as well as supply is valued significantly, but to a varying degree depending on use. Ecosystem services service valuation essentially redefines these values more closely to the ecosystem functions that supply them. Most economic valuation studies to date have examined the value that people attach to high quality water resources through the use of surveys using revealed or stated preference. These methods remain valid for cultural ecosystem services and, where adequately understood by respondents, for many regulating services too. For other regulating and provisioning services, alternative, but conventional methodologies, such as production function methods, can provide an appropriate means to estimate value. The principle distinction is to relate these values as closely as possible to ecosystem functions and, where data is available or forthcoming in the future, to define the marginal value of the ecosystem service in relation to varying levels of environmental quality.

4. Transitional and Inshore Coastal Ecosystem Services

4.1 Policy context

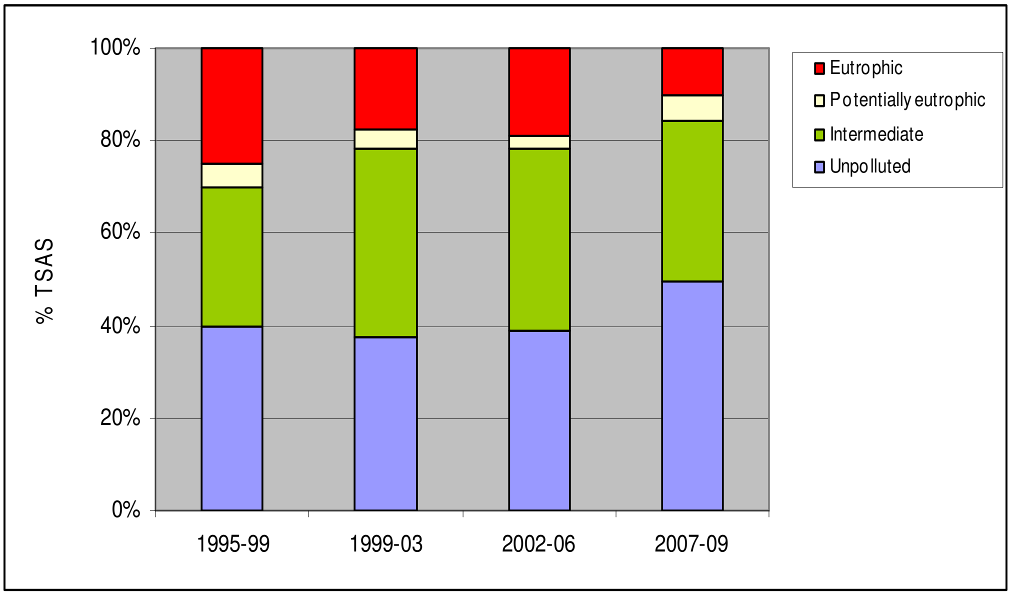
In addition to freshwater environments, the WFD addresses the Good Ecological Status of transitional and coastal waters up to one nautical mile from the coast and Good Chemical Status up to 12 nautical miles. Coastal management in Ireland is also subject to the Marine Strategy Framework Directive (2008/55/EC) which was transposed into national law in 2011 (SI No 249) and which aims that European seas achieve Good Environmental Status by 2020. Under the Directive, Member States are obliged to define indicators of marine health and to have monitoring programmes in place by 2014 together with appropriate measures by 2015/16. As a signatory to the OSPAR Convention, Ireland is also committed to developing an Ecosystem Approach to Management (EAM) for the marine environment of the North-East Atlantic. A pilot project in the North Sea has identified nine Ecological Quality issues referring to broad functional components or key descriptors of the ecosystem. Each issue is accompanied by a management objective based on Ecological Quality Objectives. Indicators were evaluated in the Irish context by Shephard et al (2013).

4.2 Status of transitional and coastal waters

For the period 2007-2009, the EPA (2010c) reported that 45.5% of transitional (i.e. estuarine) and coastal waters were of high or good status. Ireland has one of the highest percentages of high quality status coastal waters in the EU. Indeed, the quality of sampled transitional waters has been improving with fewer sites listed as eutrophic in the latest sample. Using the traditional Irish criteria the EPA report that 10.1% were eutrophic, 5.6% were potentially eutrophic, 34.8% were intermediate and 49.5% were unpolluted. While just under half the sampled surface area corresponds to WFD targets, the EPA reports an improvement in quality with the vast majority (99.5%) of sites having with good phytoplankton growth and satisfactory oxygen levels capable of supporting nearly all forms of aquatic life. However, the agency notes that 31 (35%) sites had excessive levels of winter Dissolved Inorganic Nitrate (DIN).with instances of nutrient enrichment especially evident in some estuaries of the south and south-east. Releases of nitrogen and phosphorus from aquaculture have diminished, although the report does acknowledge that the proportion of shellfish graded A has fallen from 50% in 1991-94 to 24% in 2006. Furthermore, while 93% of the 122 designated seawater bathing areas complied with EU minimum mandatory values, the level of compliance has continued on a downward trend. High rainfall during the sampling period was a factor in this respect, the effect of which is in contrast to the dilution effect of rainfall in freshwater bodies.

The Trophic Status Assessment Scheme (TSAS) has been developed for estuarine and coastal waters (see Toner et al., 2005) as a means to measure compliance with good environmental quality. The index measures DIN and ortho-phosphate (MRP) along with data on chlorophyll and oxygen conditions. In tidal fresh waters and intermediate waters the threshold levels for dissolved oxygen are within 70-130% saturation. Saturation levels below 60% are indicative of oxygen depletion. For example, saturation in the Avoca estuary in 2006 reached just 13%, a level likely to adversely affect most aquatic organisms. Nutrient enrichment increases the frequency and duration of phytoplankton blooms (including of toxin-emitting species), depletion of dissolved oxygen and changes in the structure and functioning of marine food webs. Shallow harbours and estuaries are most at risk, for example Wexford Harbour as phytoplankton tend to be flushed out of deeper channels removing the competition for macroalgae. Excessive growth of macroalgae is encouraged by nutrients. This growth typically has an adverse visual impact and emits strong odours as it degrades. Inadequate wastewater treatment, such as that from Letterkenny in 2002-06, can significantly contribute to eutrophic status (EPA, 2007). Wastewater treatment has been improving. Eighty-two per cent of municipal wastewater received at least secondary treatment as of 2004-05 compared with 21% in 2000-01. However, there is no treatment or only basic treatment at 112 locations with the majority of these discharging to estuarine or coastal waters (EPA, 2010d).

Figure 4.1 Estuarine and Coastal Waters Trophic Status Assessment



The eutrophication status of Irish waters were amongst the MSFD indicators evaluated by Shephard et al (2013). Eutrophication applies to the estuarine and near-shore area. In the OSPAR area it is monitored through a common procedure comparable to TSAS. Under the convention, thresholds have been set for winter nutrient concentrations of DIN and DIP, median phytoplankton chlorophyll, phytoplankton indicator species, oxygen saturation and kills of zoobenthos species (of which there should be evidence of none). Other indicators assessed by the team that are relevant to estuarine or near-shore waters included seal and seabird population trends, contaminant concentrations in seabird eggs, and the proximity of commercial fish stocks to safe biological limits.

Table 4.1 TSAS criteria and thresholds (check)

TSAS Categories	Threshold levels
Category A (Nutrient Enrichment)	
Dissolved Inorganic Nitrogen mg/l	
Tidal Fresh Waters	>2.6 Median Winter or Summer
Intermediate Waters1	>1.4 Median Winter or Summer
Full salinity Water2	>0.25 Median Winter or Summer
Ortho-phosphate (MRP) µg/l	
Tidal Fresh Waters	>60 Median Winter or Summer
Intermediate Waters1	>60 Median Winter or Summer
Full salinity Water2	>40 Median Winter or Summer

Category B (Accelerated growth)	
Chlorophyll (g/l)	
Tidal Fresh Waters	>15 or >30 Median and 90%ile Summer
Intermediate Waters1	>15 or >30 Median and 90%ile Summer
Full salinity Water2	>10 or >20 Median and 90%ile Summer
Category C (Undesirable disturbance)	
Dissolved Oxygen (per cent Sat)	
Tidal Fresh Waters	<70 or >130 5%ile and 95%ile Summer
Intermediate Waters1	<70 or >130 5%ile and 95%ile Summer
Full salinity Water2	<80 or >120 5%ile and 95%ile Summer

Climate change may be having an impact on estuarine and coastal habitats with evidence of changes in species composition, including expansion of warm water species and reductions in cold water species (Mieszkowska et al., 2006; Hawkins el al, 2008). In particular, future sea level rise, extreme weather events, changes in currents and warming of shallow waters, could lead to dramatic changes in composition and greater exposure to storms as natural barriers are submerged. Populations of coastal seabirds such as kittiwakes and auks already appear to be falling due to northward migration of sand eels.

4.3 The Character of Estuarine and Coastal Biodiversity

Estuaries and coastal ecosystems illustrate the divergence that can exist between levels of species biodiversity and functional diversity or resilience. According to Constanza et al (1995) coastal ecosystems, and estuaries in particular, are characterised by low species diversity, but high functional diversity. The former is low as a consequence of the inherent low predictability of the environment to which relatively few species are adapted. Tropical coral systems are an exception due to the more stable environment.

The low predictability of the environment arises from the high variability in environmental conditions. This is a consequence of rapid water movement and associated changes in salinity which in turn place severe osmotic stress on organisms' ability to maintain an internal balance of salts. These coincide with physiological needs to adapt to high amplitude stresses from the hydrological processes of waves and tides interjected with occasional storms. These conditions do though allow for easy dispersal and consequently species have adopted high levels of mobility in response. There are rather few species which can be described as specialist or keystone species on which various ecosystem processes depend. There are also few standing ecological structures with the exception of sea grasses, oyster reefs or rooted algae such as kelp. Rather, most species are generalists which are not restricted to performing a single ecological function. They are also opportunists, able to respond to changing conditions or to re-colonise the environment following a major disruptive event, including any niches left vacant by other species.

Although taxonomic biodiversity is low, functional diversity and biological productivity are high. In part this arises from high nutrient inflows from rivers in particular. There are high levels of carbon fixation, high levels of nitrogen assimilation and high levels of nutrient cycling. The functional diversity is facilitated by high levels of adaptation which in turn allow for high levels of ecological resilience. Constanza et al give the example of Chesapeake Bay where the ecology had fully recovered within two years from a one in two hundred year tropical storm in 1972 that dumped huge volumes of sediment in the bay and replaced virtually all salt water with a deluge of freshwater. Nevertheless, they note that there are some key ecosystem processes that are vulnerable to changing conditions. Amongst these is the rate of nitrification-denitrification which is directly proportional to nutrient loading. Gradual increases in the nutrient flow into the Chesapeake Bay due to human activity have contributed to excessive algae growth and losses of oxygen supplies to benthic species. Over-harvesting of oysters has exacerbated the situation by reducing the removal of algae while the reduced clarity of the water has shaded submerged vegetation such as sea grasses which are important as habitat and for sediment capture. Overall biological activity has remained the same, but the combination of ecosystems has changed.

For humans, the problem is not fragility within the ecosystem or loss of productivity, but rather the imbalance within a potentially productive ecosystem. Under ideal conditions, the high levels of primary productivity support equally high level of secondary productivity, for example of bivalve and fish populations. However, the relationships at work are complex and shellfish populations that did well at one location for several years may suddenly experience low recruitment because of the mobility within their life cycle and their dependence on conditions at some other location. The resilience of the ecosystem does not exclude the prospect of toxic substances accumulating in shellfish or of desirable provisioning species being replaced by other less desirable species following, for example, a pollution incident.

Constanza et al characterise the coastal ecological systems as comprising the phases of

- exploitation (e.g. colonisation)
- conservation (consolidation),
- release (disruption due to storms, etc.) and
- reorganisation.

They argue that estuarine systems are constantly being reset to the exploitation phase. In this environment, cause and effect are difficult to define and anthropogenic management of the ecosystem to maintain high levels of desirable ecosystem services is challenging. As regards incidents that could be applicable to the ELD, it is reassuring that the estuarine and coastal zones are regarded as being relatively resilient. This is as well given the enormous pressures being placed on these environments through settlement, tourism, pollution and over-exploitation of resources. However, while resilience may be retained in overall ecosystem functioning, individual habitats that may be valued by human beings remain vulnerable, including shellfish populations. While new species might quickly emerge to fill vacant niches, there is no guarantee that these will provide the same ecosystem service. The complexity and dynamics of the environment greatly complicates sustainable management, for instance in relation to the total allowable catch (TAC) of shellfish.

4.4 Categories of ecosystem services

Estuaries and coastal waters provide an important mix of supporting ecosystem services in the form of primary and secondary production and habitat provision, regulatory services such as water quality, nutrient cycling, detoxification of wastes, physical transport of sediment and storm protection, provisioning services of fish, shellfish, crustaceans and seaweed; and cultural services including beach tourism, angling and birdwatching, etc. The UK Valuing Nature Network (VNN) has been preparing a list of ecosystem services based on habitats and species matrices for estuarine and coastal environments originally prepared by Fletcher et al (2012) for the English Marine Conservation Zone. The list is comprised of broad scale habitats, more specific habitats and species against which are listed the various ecosystem services with which they are associated along with the respective level of scientific evidence of their value. This list provides the basis for our own attached matrix of freshwater habitats, species and ecosystem services (Appendix 5) and for the matrix representing the same set of ecosystem services associated with estuarine and coastal environments (Table 4.2). Perhaps to a greater degree than any other environment, the estuaries and coasts display more interdependence between ecosystem services. While this environment is resilient in the face of many natural and human pressures, there is still the capacity for impacts to have multiple effects at various levels through linkages that science still poorly understands.

4.5 Supporting ecosystem services

All the broad habitat types are valuable for supporting ecosystem services, but those which feature strongly include intertidal outcrops, largely submerged (infralittoral) rock outcrops, intertidal mud and sediment, subtidal sediment, reefs and salt marsh. These habitats are important for species diversity, but also for primary and secondary production, larval supply and nutrient cycling. Primary production within estuaries has been shown to contribute to secondary production up to 4km from the estuary mouth, although with impacts that vary from site to site depending on currents and local conditions. This 'resource subsidy' is an important feature of estuaries and can account for 60% of primary production in the environment as a whole (Savage et al., 2010). As these services are intermediate, it is important to avoid double counting the benefits with provisioning services.

Salt marshes contribute to filtration of pollutants entering estuaries, for instance from adjacent agricultural land (Metsch and Grosselink, 2008). King and Lester (1995) note that salt marshes can either be net accumulators or exporters of sediment or organic matter depending on whether they are growing or not. They refer to Boorman and Wells (1993) who estimate primary productivity as between 975–1031 dry wt g/m/yr in Essex, but note also Adam's (1990) warning that figures are likely to vary significantly from site to site depending on the status of the saltmarsh.

Seagrass, seaweed and tidal swept algal communities provide a range of vital supporting services including living space for other organisms including scallops and juvenile cod, halibut, flounder and plaice (Austin et al., 2012). Kelp is especially important for juvenile cod as is salt marsh for shrimp, bass and herring (Green et al., 2009). The most common types of seaweed found in Ireland are kelps (*Laminaria*) and wracks (*Fucus* and its relatives including *Ascophyllum nodosum*), the latter found in the intertidal environment, as well as the detached coralline algae maerl. Both common maerl and coral maerl are important for supporting services, especially as refuges and sources of food for juvenile fish and shellfish including pollack and scallop.

Native oyster beds are valuable food for natural predators such as oystercatcher and wintering eider duck as well as providing important regulating services for water quality and a provisioning service for human beings. Goss-Custard et al (2004) found that shellfish populations eight times above the levels previously thought satisfactory are needed to ensure that bird survival is not put at risk, especially where opportunities for feeding at high tide are limited. Estuaries, lagoons, coastal fields and in-shore waters are also extremely important as resting and feeding grounds for migratory and wintering birds, not least in Ireland which harbours significant proportions of the world population of some species.

For salt marsh, Luiseti et al (2011) undertook a choice experiment to determine the public's valuation of the benefits of the habitat for birdlife as well as amenity. They found that these values are very dependent on the size of the area (being proportionately higher for smaller areas) and proximity of a nearby large population, but subject to sharp distance decay. However, use values (cultural services) tended to account for three-quarters of the values expressed. Total values for the Blackwater Estuary in Essex varied between £360 (€426) and £771 (€910) for an 832 hectare area at an average distance of 32 and 8 miles respectively from the relevant population. Values were only slightly higher (£482 and £1005) for much larger 2400 hectare areas at the same distances. The Luiseti et al study therefore demonstrates the familiar characteristic of survey-based methods whereby respondents express diminishing returns by area of salt marsh. By comparison, the benefit of supporting services would be expected to rise at least proportionately with size of area.

4.6 Regulating services

4.6.1 Waste assimilation and nutrient cycling

Inter-tidal sites are characterised by sediment deposition including organic matter. The disturbance of sediment and burrowing of nutrients by benthic invertebrates living in mudflats and salt marsh is the first stage in bioturbation and is essential for secondary production. Organic material and nutrients then pass to microfauna and bacteria where they are further broken down (Fitch and Crowe, 2011). By some accounts, bacteria could be responsible for a 40% reduction in nitrogen, the principal limiting nutrient in the coastal environment and second to phosphorus in estuaries.[42] Key species are listed in the attached matrix (Appendix 5) as being important for food web dynamics and diversity. The matrix does contain a large proportion of empty blocks as the relationships are poorly understood given that most studies have been carried out in laboratory environments based on a few species. What is known is that under normal conditions, cool temperature waters tend to be well oxygenated to the benefit of nitrifying communities. Huge uncertainties apply to interrelations between species and so there are a large number of individual species missing from the list.

In fine mud, oxygen diffuses only in the first few centimetres limiting primary production to eelgrass (*Zostera Spp*) and diatoms (algae). Secondary production is performed by bacteria, invertebrates (worms and protozoa that feed on the bacteria) and larger worms, crustaceans and molluscs. Mudflats are often the recipients of significant amounts of organic matter so there is plenty for these organisms to feed on. The microbial communities play an important role in the production and remineralisation of organic matter. Opportunistic green algae often colonises the surface in these situations. Oxygenation occurs as the organic matter is degraded with bacteria responsible for making this available to larger organisms in carbon fluxes. The bacterial activity has the effect of smoothing seasonal variations in primary production, freeing up food supply and allowing the re-absorption of dissolved nutrients. However, beyond 1cm, the poor oxygenation in fine silts and clays means that decomposition is anaerobic and so proceeds more slowly. The carbon-to-nitrogen ratio rises rapidly below the surface (McLusky, 1989).

By comparison, sands are more unstable and difficult for plants to colonise, but easy for organisms to burrow into where waterlogged. In contrast with mudflats, sands tend to have high levels of oxygenation and consequently low levels of organic matter unless mixed with finer materials. Intertidal sands are oxygenated by the movement of water and shifting of sediment.

In each of these environments, conditions vary considerably at local level. In common with other studies, Cook et al (2004) find higher levels of primary production in more exposed upper estuary locations. By comparison, they report low and constant levels of primary production along the profile of lower estuary mudflats subject to periodic submergence. In Lyme Bay in Dorset, England, Rees et al (2012) found a degree of spatial segregation in intertidal areas dominated by mud, with mixtures of mud and sand being least favourable for energy transfer, but with soft mud characterised by active bioturbation processes. Light (including water clarity), exposure and temperature (including seasonal considerations) all appear to be positively correlated with primary production (Cook et al., 2004). Stresses on these processes arise from acidification or higher temperatures. Excessive organic material, for instance from upstream erosion or pollution, will raise the demand for of oxygen and present a risk of eutrophication.

Biological traits analysis is used to examine ecosystem functions in relation to biological assemblages. Ecosystem processes relevant to nutrient cycling, climate regulation and bioturbation include energy fixation, transfer and burial/enhancement for microbial processes. Ecosystem cascade theory (Haines-Young & Potschin 2007) has been used to present a linear framework between biodiversity, ecosystem functioning and human well-being as a prompt to the complexities of the relationship.

4.6.2 Natural hazard regulation

Amongst the most significant of ecosystem services provided by coastal habitats are the contribution of salt marsh, other tidal swept communities, seaweed and dunes to defence against

[42] Pers comm with Shane O'Boyle of EPA.

storms and storm surges and the benefits this provides in terms of cost avoided for communities and infrastructure. The protective value of the salt marsh depends largely on its slope and width and the depth of water offshore. The former two determine how effective the marsh will be in dissipating energy and the height of any seawall that might be required. It has been estimated that an 80 metre strip of salt marsh can allow a reduction in a seawall height from 12 to 3 metres with cost savings of £2,600-£4,600 (€3,068-€5,428) per metre (National Rivers Authority referenced by King and Lester, 1995). Salt marshes also supply additional shelter to marinas. There is no evidence that mooring costs take this contribution into account, but that mooring bills can be significant as can the cost of storm damage to boats.

Table 4.3 Savings on seawall construction and maintenance costs due to protection from salt marsh

width of marsh	wall height (m)	cost of new wall (£ m)	maintenance costs	savings on construction costs (£ m)	Savings on maintenance costs (£ m)
80	3	400	1	2600-4500	49
60	4	500	5	2500-4500	45
30	5	800	15	2200-4200	35
6	6	1500	25-30	1500-3500	20-25
0	12	3000-5000	50	0	0

Source National Rivers Authority (1992) and King & Lester (1995)

However, subsequent studies have shown that the relationship is non-linear and varies from site to site. If submerged for only short periods, salt marsh becomes established with the vegetation binding the sediment and strengthening the shoreline against erosion. Water depth plays a part with Moller et al (2006) arguing that salt marshes in the UK have a dissipating role at depths under 3.7m. At low inundation depths, they estimate that wave attenuation has is 87%, falling to a still respectable 72% at greater depths. More recently, Angus et al (2012) estimate that salt marsh and shingle dissipates 50% of wave energy within the first 10-20 metres. At very shallow depths waves are likely to break before they reach the salt marsh except at times when storms are combined with high tides. However, salt marsh has a more minimal role at the greater depths that might be expected during a storm surge. At these depths the dissipating effect may be replaced by erosion of the marsh. This could imply that salt marsh is effective is protecting agricultural land and structures (including walls), but less effective in protecting against the more serious losses that would follow from severe storms.

The consolidation of sediment, typically fine silts and clays, in intertidal environments, also has a role in strengthening coastlines. The consolidation forces out porosity providing flats with significant strength to resist erosion. By contrast, sand flats lack the same cohesiveness. Where it occurs, seagrass is very effective at consolidating sandy sediments, but it is vulnerable parasitic epiphytes or reduced water clarity where nutrient levels are high or to physical disturbance including local changes in currents brought about by coastal structures.[43]

Rocky habitats are of evident value in regulating storm hazards and consequently erosion. So too, to a slightly lesser extent, are oyster beds, for example those in the Oosterschelde Estuary in the Netherlands (Troost 2010). Mussel beds also appear to stabilise sediments (Reise, 2002) while oyster beds provide additional regulating services by protecting juveniles from predation and supporting services by stimulating water turbulence that increases food availability. By agglomerating into large reef structures that bolster survival, Pacific oysters could also be filling a niche left empty by diseased native oysters while possibly also enhancing biodiversity by providing greater shelter and sediment clear conditions for other species living on their reefs (Markert et al., 2009). Consequently, any destruction of shellfish beds will not only impact on harvesting, but also regulating and supporting services too. On the other hand, there is potential for an extension of this

[43] Padraig Whelan. Pers. Comm..

set of ecosystem services following managed realignment of coastlines in response to climate change. The defensive advantages have already been demonstrated with the deliberate breaching of defences at Alkborough Flats on Humberside (Angus et al, 2012).

There are economic welfare benefits too as well as damage avoided. Various surveys have been conducted in the US of the value that people place on dunes and beaches for coastal protection. For example, a hedonic study based on property values in Georgia, USA (Landry and Hindsley, 2011) estimated this contribution at between $71 and $195 (€54-€141) per metre for 100m and 300m of high tide beach width or at between $52 and $132 (€37-€95) per metre for between 100m and 200m of dunes width. Ocean frontage was still valued, but contributed $39,000-$75,000 to property values compared with $121,000- $128,000 for more protected inlet frontages.

4.6.3 Climate regulation

Intertidal habitats are valuable for carbon sequestration. The carbon flux is determined by the combined effect of vegetation, macroalgae and microbes in the context of a dynamic environment. This ecosystem service has been measured by Adams (2008) and Fonseca (2009) and found to favour sequestration in contrast to the emissions produced by many freshwater wetlands. Luisetti et al (2011) report estimates of carbon sequestration for sedimentation rates of 1.5mm and 6mm per year the Humber and Blackwater Estuaries (Essex). The rate varies by location even within short distance depending on the duration of submergence. At a temperate location in Australia, Cook et al (2004) found that levels of gaseous CO_2 exchange were 3-4 lower than that taking place once the area was submerged.

Table 4.4. Carbon sequestration per ha per year (Luisetti et al, 2012)

Sediment *spartina* marsh	**C burial**	**CH4**	**CH4 (CO2-eq)**	**N2O**	**N2O (CO2-eq)**	**Net C burial**
1.5mm	1.027	0.0012	-0.025	0.00237	-0.735	0.266
6mm	4.108	0.0012	-0.025	0.00237	-0.735	3.347

Seagrass had been thought to make a modest contribution to carbon sequestration, but recent evidence suggests that this could be more significant (Duarte et al., 2010). Seagrass occurs at various locations around Ireland and some large areas are covered by submarine species. Sequestration of carbon by sea grasses has been reported at between 0.2 and 2.0 t C ha^{-1} (Romero et al., 1994). Indeed, recent data provided by Wium-Anderson and Borum (1984) suggests rates of 8.84 t/C $ha^{-1}yr^{-1}$ are possible. As these estimates were derived from Denmark it can be expected that estimates for Ireland will depend on the density of growth (optimality of conditions) and be subject to the differing environmental conditions and rates of productivity. However, current rates of sequestration of carbon, rather than long-term storage, is of most relevance for seagrass given the small volume of the biomass and because most carbon is recycled into the ecosystem over time. The objective is therefore to sustain this environment and activity. In this respect, seagrass is vulnerable to pollution and disturbance as well as to invasive species such as *Spartina anglica*. Re-establishment of degraded beds has proven to be difficult (Parker et al., 2004).

Macroalgae, i.e. sea weed and especially kelp (*Laminaria hyperborean*), also has an important carbon sequestration role. Growing on rocky surfaces, kelp does not result in long-term burial or storage of carbon either, but the volume of biomass is important for storage with sequestration ranging from 1.2 to 7.2 t/C $ha^{-1}yr^{-1}$ (Reed and Brezenzski, 2009) to as much as 12.9 t/C $ha^{-1}yr^{-1}$ (Abdullah and Fredriksen, 2004). In the UK, sequestration of 4 t/C ha^{-1} has been recorded (Gavaert et al, 2008).

Other transitional environments also have a regulating value for greenhouse gases. Salt marsh has been estimated to sequester between 0.64-2.19 t C $ha^{-1}yr^{-1}$ (Angus et al, 2012)(Cannell et al., 1999).

Figures exist for carbon sequestration by plankton too, but there are no estimates for macroalgae in intertidal locations (Austin et al., 2012).

Methods of carbon valuation were discussed in Chapter 2. Estimates range from £20-30 per tonne (Pearce et al.; 1996, Li et al, 2004; Tol, 2007) to £350 (€412) per tonne (Stern, 2007).

4.6.4 Removal of heavy metals

Some coastal or marine macrophytes have the capacity to absorb heavy metals such as chromium, cadmium and zinc through bioaccumulation. Absorption occurs gradually rather than in response to a sudden pollution incident, but this does permit seaweed to perform a useful indicator function of local environmental pollution.[44] In Ireland, in a laboratory environment, Murphy (2007) has found that the seaweeds *Fucus versiculosus* and *Polysiphonia lanosa* were most effective at removing cations and anions in high solutions with *Palmaria palmate* performing well in low solutions. This absorption can have implications for human health where wild seaweed is consumed, but seaweed's capacity to absorb metal also indicates some potential for deliberate management in the vicinity of industrial waste outflows.

4.7 Provisioning ecosystem services

4.7.1 Inter- relations between supporting and regulating services

Landings of crustaceans and bivalves in Ireland over the period of 2004-10 have varied annually from 29,533 tonnes in 2004 to 14,000 in 2008 (Marine Institute, 2011b). This activity supports 1,959 vessels. In 2010, the value of the harvest was €43 million with the most valuable species being scallop, lobster and edible crab. Total fish landings into Irish ports in 2010 were 245,956 tonnes with a total value of €207 million. Landings of demersal fish, which have most relevance to the coastal environment, amounted to 40,867 tonnes with a value of €79 million. Landings by Irish vessels accounted for 60% of this tonnage (SFPA, 2010).

Although tidal swept beds are important for shellfish collection and commercial fishing, it appears that relatively little is understood about the relevant ecosystem processes. A verdict on the environmental impact of introduced Pacific Gigas oysters (*Crassostrea gigas*) is disputed, but they do provide a more productive alternative to the provisioning service once provided by the native oyster. As they have naturalised, Pacific oysters are no longer confined to aquaculture, but are being harvested by shellfish boats.

Many estuarine and coastal environments support fish and shellfish at various stages in the life-cycle, most notably areas of infralittoral rocks. Amongst the more mobile species, salmon are important for provisioning services and use estuaries in the process of migrating from marine to freshwater environments. The value of most fish species, including demersal species, is relevant to a fuller assessment of the marine environment, but it is evident that catches of most species are below maximum sustainable yield and that fish populations, and their economic value could be greatly increased by conservation efforts. The Atlas of Commercial Fisheries around Ireland (Marine Institute, 2009) notes that stocks of cod, whiting and various flatfish are all depleted and below historic levels.

Transitional environments are important as nursery or overwintering grounds for a number of commercial fish species. The factors that influence the productivity of nursery grounds for different fish species are varied and not well understood, but estuaries do provide a feeding ground, a lower salinity environment (possibly reducing osmotic stresses in juveniles) and reduced risk of predation of eggs or juveniles due to the turbid waters. Temperatures tend to be warmer and the shelter and reduced predation attract overwintering fish. High seasonal prey densities, for example of shrimp,

[44] For example, seaweed in Cork Harbour was able to demonstrate high levels of zinc (P. Whelan pers. comm.)

are evident in the Fifth of Forth in Scotland as are variations in predation from other fish or wintering ducks (Elliot et al., 1990). The supply of benthic invertebrates is clearly a key influence (Howell et al, 1999). Kosteckia et al (2010) find that flow variability and sediment load have a significant influence on sole recruitment. Low salinity appears to be important during the juvenile stage of the species and so reductions in flow would be expected to have an adverse effect (Costa and Bruxelas, 1989).

Although there is insufficient knowledge of the spatial population dynamics of many fish species, some broad economic projections are possible. Elliot et al (1990) note that the Forth Estuary is believed to support around 0.5% of (similar sized) stocks of plaice and cod in the North Sea. If this observation can be extended to harvestable amounts then the ecosystem service provided by the estuary would be valued at €95,000. However, this figure rests on landings in 2009 of 22,000 tonnes valued at £19 million (OECD, 2009).[45] For most of the 1970s landings were well in excess of this total at over 300,000 tonnes (ICES 2009).

Cordier et al (2011) pulled together fisheries experts to construct an input-output model which demonstrated that restoration of 74km^2 of the Seine Estuary could restore the sole population to 44% above that of a business-as-usual scenario with consequent economic benefits. The choice of an input-output model reflects the interdependencies in the ecosystem. Luisetti et al (2011) focus only on marketable fish, primarily bass, in assessing the nursery value of the Humber and Blackwater estuaries and supply the following table of upper and lower estimates based on data from Forseca (2009). The table is only a guide as actual survival and growth will be dependent on a range of factors.

Table 4.5. **Value (€ per ha) and weight of bass contributing to inshore fishery** (Luisetti et al, 2011)

Survival parameter estimates	**upper**	**mean**	**Lower**
Value per ha at average wholesale price (£7/kg)	55.99	13.63	2.28
Value per ha at lowest wholesale price (£4.50/kg)	35.99	8.77	1.46
Weight (kg) of juveniles per ha surviving to 36cm after 4-5 years	6.78	1.65	0.28

4.7.2 Productive output

The prospects for several shellfish species are negative in part due to over-exploitation and an absence of stock management such as input or output controls or TAC. The Marine Institute provides a discussion of the prospects for several species in its 2011 Review (2011a). The section on the cockle provides an illustration of recent recruitment failures compounded by the uncertainies given the inherent complexity of the ecosystem and local over-exploitation. While management plans have been agreed in the Dundalk Bay and Waterford Harbour Natura sites, the cockle fishery was closed in Tramore due to the failure to reach a local consensus.

Recruitment of surf clams also varies considerably annually and management is compounded by the high mortality of small clams returned after fishing. For native oysters (*C. edule*) the main remaining locations are Inner Tralee Bay, Lough Swilly and Galway Bay. However, the performance of native oysters has continued to be poor with recruitment having failed in recent years. The population is still reeling in some locations from the accidental introduction of the parasite *Bonamia ostreae* across north-west Europe in the 1960s. Although protected under the Habitats Directive, no management plans have been agreed despite the relatively small number of operators. Local TACs are described as being arbitrary and not based on scientific evidence.

The Marine Institute describes the presence of the Pacific oyster *Crassostrea gigas* as a potential threat. These more productive oysters were introduced to fish farms in the 1970s, but naturalised in

[45] Data for North Sea and Skaggerak combined. Note may be just Danish vessels but think not

the cool Irish waters despite assurances to the contrary. As noted above, opinion on their impact varies. According to Troost (2010) the Pacific oyster does not appear to present a significant competitive impact on the native oyster in the Netherlands or on food sources for other species.

Oysters are vulnerable to pollution or coliform contamination. Dioxins are especially toxic and, as with other shellfish, can accumulate in tissue. Dioxins refer to 205 substances (including PCDDs and PCDFs of which 17 are of toxicological concern) and are highly persistent in the environment. Fortunately, fish samples from around Ireland have indicated that concentrations are well below EU legal limits, although no maximum acceptable daily consumption limits have been set. Shellfish are monitored for *E-coli* and heavy metals such as cadmium, lead, mercury and chromium. Monitoring of shellfish toxins is undertaken by the Marine Institute.

Table 4.6: SFPA Regulation (EC) No. 854-2004 Microbiological Treatment Required

Status	E-Coli contamination	Treatment
A1	<230 *E. coli* per 100g flesh and intra-valvular liquid.	May go direct for human consumption
B	<4,600 *E. coli* per 100g flesh and intra-valvular liquid.	Must be depurated or relayed to meet class A
C	<46,000 *E. coli* per 100g flesh and intra-valvular liquid.	Relaying for a period of at least 2 months prior to sale

nb Shellfish going directly for consumption must also be free from *Salmonella* spp.

The contribution of estuaries and inshore coastal environments for fin fisheries is dealt with in the River Suir and Waterford Harbour case example to follow.

Seaweed is collected commercially for fertiliser in the west of Ireland and for use in the pharmaceuticals sector. The market for kelp (*Laminaria digitaria*) in Ireland is worth $18 million annually (BIM, 2011). Seaweed has a wide variety of important commercial uses in the cosmetics, food and chemical sectors and as a fertiliser. *A. nodosum* is collected for alginate (gum) production as are small amounts of *Laminaria digitata and L. hyperborea*. Potentially, 130,000 wet tonnes of *A. Nodosum* are available for collection of which current collection is 34,000 tonnes according to the Irish Seaweed Centre (www.seaweed.ie) annually. There is currently much interest in so-called Blue Biotech pharmaceuticals and its value to the biotechnology sector is currently estimated at €18m annually.

A more traditional activity is the digging up of ragworms, lug worms and peeler crabs for bait. In the UK it is estimated that 500-700 tonnes are dug for personal use with a further 300-500 tonnes being harvested for a market worth £25-£30 (€30-€35) million (Fowler, 1999).

4.8 Cultural ecosystem services

4.8.1 Activity participation and expenditure

The coastal environment attracts a considerable level of tourism and amenity. Specific activities do, of course, vary by environment with sailing occurring offshore, birdwatching most often along areas of coastal wetlands, mudflats or rocky areas and much general amenity being attracted to beaches or miscellaneous stretches of coast. The Marine Institute/ESRI survey (Williams and Ryan, 2003) estimated levels of participation and the annual economic expenditure that was associated with coastal tourism at the time (Table 4.7). Total expenditure was estimated at €377 million. Curtis (2003) reports a level of social exclusion for boating activities, but high consumer surplus estimates overall even after allowing for zero reported trips. The highest estimates were for swimming at £72 (€91) or £184 million (€233m) in aggregate (2003 figures). More recent estimates published by the

Irish Marine Federation (O'Driscoll et al., 2007) confirmed an upward trend in all forms of coastal and marine tourism which it valued at €566 million per annum.

Table 4.7. Participation and expenditure value of coastal tourism

	participants	expenditure
Trips to beach or swimming	1,488,000	€278 million
Sea angling	71,000	€26 million
Sailing	100,000	€25 million
Other water-sports	53,400	€35 million
Nature-related tourism	65,500	€7 million
Visits to islands	33,200	€6 million

Table 4.6 provides details of estimated levels of participation in these coastal activities indicating predictably high levels of participation in general seaside related activities and in swimming.

Table 4.8 Participation and average expenditure

	.000s	*average expend per trip (€)*
Angling		
sea angling from shore	74,100	4.22
sea angling from boat	53,000	17.13
Boating		
sailing	58,800	5.11
kayaking/boating	32,100	1.61
power boats, etc	24,600	6.62
Watersports		
Water/jet skiing	19,200	28.11
surfing/sail boarding	17,800	8.34
scuba diving	9,100	8.52
other sea sports	7,300	19.31
Seaside trips		
swimming	353,500	5.58
Whale/dolphin watch	9,600	11.50
Birdwatching	12,400	8.10
Nature reserves	43,500	13.88
Other trips	1,136,600	3.10
Trips to islands	33,200	26.68

Table 4.8 shows the estimated level of expenditure on equipment and day trips per person associated with these activities. Possibly some of the figures for day trip expenditure can be questioned. However, the high figures for sailing and for scuba diving are quite plausible. Overall, the estimated annual coastal equipment spend of €61 million and day trip spend of €108 million confirms the high level of total expenditure related to the most popular activity of seaside trips and swimming. Certainly, it can be expected that relative trends in activity will have changed again since 2004. The report notes a reduction in the number of day trips compared with the preceding survey in 1996 and it is likely that there has been increased participation in surfing and scuba since the 2003 survey.

Table 4.8 Expenditure on day trips

	equipment (€ per person)	**day trips** (€ per person)	**total equipment aggregate** (€ per person)	*total day trips aggregate* ***(€m)***
Angling				
sea angling from shore	97.9	61.3	**7.2**	**4.5**
sea angling from boat	86.2	109.5	**4.6**	**5.8**
Boating				
sailing	159.1	52.2	**9.4**	**3.1**
kayaking/boating	19.1	19.7	**0.6**	**0.6**
power boats, etc	27.1	55.6	**0.7**	**1.4**
Watersports				
Water/jet skiing	685.2	234.7	**13.2**	**4.5**
surfing/sail boarding	63.5	82.6	**1.1**	**1.5**
scuba diving	532.7	113.0	**4.9**	**1.0**
other sea sports	36.9	61.9	**0.3**	**0.5**
Seaside trips				
swimming	14.4	43.9	**5.1**	**15.5**
Whale/dolphin watch	58.3	23.0	**0.6**	**0.2**
Birdwatching	0.7	4.5	**0.0**	**0.1**
Nature reserves	16.0	38.5	**0.7**	**1.7**
Other trips	10.1	58.1	**11.5**	**66.0**
Trips to islands	27.2	44.9	**0.9**	**1.5**

For overnight trips, individual expenditure varied between €50 per year for water skiing/jet skiing to €628 on surfing/sail boarding. The figures in Table 4.8 should not necessarily be accepted at face value in that the report notes that the figures on expenditure on accommodation may reflect package trips. The high accommodation expenditure reported for surfers and kayakers looks rather questionable. Less open to dispute are the high travel and additional costs reported by these groups. Given the relative number of participants, the column for total expenditure does demonstrate the high value anticipated for general seaside trips, swimming in the sea and sailing. Total overnight related coastal and marine expenditure amounted to €215 million.

Table 4.9 Expenditure on overnight trips

	accomm. (€ per person)	**travel** (€ per person)	**'other'** (€ per person)	**total** (€ per person)	*total aggregate* ***(€m)***
Angling					
sea angling from shore	80	14	24	118	**2.2**
sea angling from boat	40	10	13	62	**1.3**
Boating					
sailing	128	10	11	148	**12.3**
kayaking/boating	244	28	33	304	**2.8**
power boats, etc	62	12	0	74	**2.0**
Watersports					
Water/jet skiing	20	13	17	50	**0.1**
surfing/sail boarding	400	91	137	628	**4.6**
scuba diving	107	19	11	137	**3.5**
other sea sports	20	7	17	43	**0.1**
Seaside trips					
swimming	247	29	37	312	**62.9**
Whale/dolphin watch	111	19	24	155	**0.8**
Birdwatching					**0.8**
Nature reserves	143	21	16	180	**1.7**
Other trips	163	25	34	159	**116.2**

Trips to islands	100	25	34	159	**3.4**

4.8.2 Coastal water quality and cultural values

Water quality can be expected to have a strong influence on people's willingness to participate in coastal recreation, although its significance is not a great for some activities as might first be expected. Ekkert and Olsson (2009) valued water quality in coastal Sweden using a choice experiment in which the attributes included bathing water quality, fish numbers and biodiversity. The study reports heterogeneity of preferences, but a high willingness-to-pay (WTP) to avoid biodiversity losses of SEK 1,400 (€168) (from a median level) and a value of SEK 600 (€72) for improved biodiversity (from the same level) coincident with prospect theory. Improved bathing water quality (worst to best quality) attracted a WTP of SEK 600 (€72) too. The authors acknowledge that vague definitions of biodiversity could have caused some respondents to be sceptical and more inclined to select choice sets with the less ambiguous rise in biodiversity (in this case, fish stocks).

In the UK, Hanley et al (2002a) used a contingent behaviour model to examine the value that beach users place on water quality and whether this is sufficient to support the investments in water treatment required for beaches in Central Scotland to meet satisfactory standards under the EU Bathing Water Directive. In the event they found that improvement would result in a very modest 1.3% increase in visits equivalent to a WTP £0.48 per trip or £5.81 (€6.86) per household per year. Figures for comparable scenarios in Scotland have been estimated by Hanley and Kristrom (2002b) for Ayr and Irvine of £9.22-£12.13 (€10.88-€14.31) and 5.29-£7.62 (€6.24-€9,00) per household per year respectively. In England, Georgiou et el (1998) arrived at higher estimates of £20.17-£37.41 (€23.80-€44.14) for beaches in Great Yarmouth and Lowestoft, the first of which was below satisfactory standards. The higher estimates for the Georgiou et al study may arise from the relatively higher importance of seaside tourism in southern England or the influence of relative incomes, but also due to the inclusion of health as a factor in the information given to respondents.

A study by Mourato et al (2003) provides some explanation for the low value placed on water quality by beach users relative to other attributes. They undertook a choice experiment survey of beach users' behaviour and attitudes towards bathing water quality, finding that 57% of respondents reported never going into the sea and 78% said they never engaged in water sports. Although these proportions are surprisingly large they broadly conferred with the output from preceding focus groups. In terms of importance, water quality was citied only sixth out of thirteen beach attributes, with respondents being most concerned with beach cleanliness and water that looked clear of foam and litter. The survey reported an average of 1.6 swims or dips per year. Thirty four respondents believed that they incurred a stomach illness from contact with the water, although this was equivalent to just 0.042 cases per person (a figure reported to be comparable to previous studies). On the basis of the best-fitting nested logit model, respondents were willing-to-pay between £0.90 and £1.10 (€1.06-€1.30) per household per year for one day less of poor water quality and between £1.10 and £2.20 (€1.06-€2.60) for a one in 100 reduction in the risk of getting a stomach upset. On the basis that the actual risk per swim in 2001 was 4.2%, the authors concluded that conformance with EU guidelines would result in a reduction of in this risk of 2.3% (Failte Ireland, 2012).

Sea angling

Sea angling from beach, shore and from boats has grown in popularity. Many fish species are dependent on shallow waters as nursery areas or for migration, but bass is currently the only species managed in Ireland for sea angling.

The number of sea angling operators has increased in recent years with government support for diversification from commercial fishing. Key centres include the coastal counties of Mayo, Galway, Kerry, Cork and Louth. In addition, to high participation in shore and beach angling, boats are usually rented by groups. Direct economic returns are complemented by local spending in areas that are often remote from mainstream economic activity. Based on household survey data, sea angling is the activity chosen by 28% of overseas angling visitors (compared with 12% coarse angling, 16% pike and 42% for game angling (IFI, 2013).[46]

Scuba

Diving is growing in popularity with participants engaging in either wreck dives or exploring underwater biodiversity, for example in Strangford Lough. It can be an expensive pastime. Based on a survey of five companies, Morrissey et al (2011) estimated that participants spend on average €330 per trip but realise a consumer surplus of €250 each time.[47] There are no cumulative economic figures, but the Irish Underwater Council (CFT) has 2,600 members accounting for approximately two-thirds of all divers. Non-equipment related expenditure for all-Ireland by domestic divers and overseas visitors is estimated at around £1 million (Ozdemiroglu et al., 2012) of which perhaps one third is not related to wreck diving.

Birds and wildlife

The supporting ecosystem service value of estuaries links in with the highly important cultural service associated with bird watching, especially of migratory and wintering bids. Examples of directly elicited estimates of utility values include Birol and Cox (2007) for the Severn Estuary. They grouped their sample into four profiles depending on income and family characteristics, but estimated average willingness-to-pay for a one fifth increase in wetland area at £13.80 (€16.28). The payment vehicle was an increase in water rates per respondent. However, the influence of so-called "charismatic" species on preference was evident from the relatively high value placed on the creation of otter holts at £31.60 (€37.29) per respondent.

For ecotourism, the ESRI study reported expenditure of €11.50 per trips for whale and dolphin watching equivalent to €1 million per year. However, in Scotland this industry has been estimated to be worth £10 million (€11.2m) per year (Parsons et al., 2003), a figure that may have risen in recent years to as much as £10 million for dolphins in the Moray Firth alone (Davies et al., 2010).[48] Also in the UK, Bossetti and Pearce (2003) estimated non-use values for grey seal protection at £5.26 (€6.20) (in terms of donations from seal sanctuary visitors) and use values for viewing trips of between £5.08 and £7.74 (€6.00-€9.13).

Another form of wildlife tourism (albeit one somewhat at odds with the above activities) is wildfowling. Although now undertaken only for recreation, in parts of Britain wildfowling rights can be quite sought after. King and Lester (1995) quote figures of £150-£493 (€177-€582) per ha per year for wildfowling rights in Essex and sale prices of between £444 and £4000 (€524-€4,720) per acre. However, wildfowling is a minority activity in Ireland and also fairly localised, being perhaps more prevalent in Northern Ireland.

Although ecotourism has the potential to raise the value of ecosystem services and to act as a sponsor in the event of climate change, it also places significant pressures on the coastal environment particularly within the no-man's land between low and high water. Problems arise from disturbance, erosion, litter, ill-sited development and sewage pollution.

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[46] Combined figures for day and overnights with sea angling defined to include bass and other sea fish. Anglers fishing for pike are also likely to fish for salmon and trout.
[47] The authors used a travel cost method.
[48] These studies may have included a wider definition of direct and indirect expenditure than the ESRI.

4.9 Summary - Estuarine and Coastal Ecosystems

In contrast to freshwater ecosystems, estuarine and coastal environments have received less attention from either revealed or started preference valuation. This provides an opportunity for future ecosystem service valuation given our evident dependence on these ecosystems to supply us with fish and shellfish, to act as a sink for our waste, to protect us from natural hazards and to provide us with the quality of environment we demand for coastal recreation and ecotourism. The challenge is that so little is understood about the dynamics, species and spatial interdependencies of marine ecosystems that it can be difficult or impossible to apply marginal economic values as ecosystem functions are not specific to any one ecosystem service. Weinstein (2008) argues that a better understanding of ecosystem services would allow us to apply location specific management to different estuaries based on their capacity to support urban infrastructure, fisheries and supporting services, or conservation interests. On the other hand, the interdependencies between supporting and other ecosystem services are such that only a holistic approach based on integrated coastal zone management (ICZM) is needed supported by the evidence of an economic assessment of valuable species or input-output modelling using a collection of best-guess coefficients.

Rees et al (2012) describe the difficulty of identifying functional diversity and of understanding the implications of external impacts for indirect ecosystem services such as the regulating services performed by the benthos. They comment that this arises from issues of redundancy or substitution, the multifunctional role of many species and from the inter-relations that exist between species which determine the effect of an impact in any one location (Solan et al., 2004). Furthermore, as well as the uncertainty attached to the functions of macrofauna, we know almost nothing of the microbial community. Given the difficulty, Rees et al agree with Haines-Young and Potschin (2007) that a better solution would be to ensure the sustainability of habitats at a broad-scale supported perhaps with evidence of locations or species that might be especially sensitive to damage.

5 Ecosystem Service values for the River Suir and Waterford Harbour

5.1 Introduction

This case application examines the potential for applying economic valuation methods for the purposes of remediation to the freshwater ecosystem services of a particular river, namely the Suir, and to the ecosystem services of its transitional waters, namely Waterford Harbour. Freshwater flow into Water Harbour is shared with the rivers Nore and Barrow and there is a tidal influence on the River Suir as far as Carrick-on-Suir. In addition, inshore coastal ecosystem services are examined in the immediate vicinity of Waterford Harbour.

The River Suir is 184 km long and drains an area of 3,546 km^2, or around 4% of the land area of Ireland. However, the total length of river channel in the catchment including the main river and the significant tributaries is about 530 km.[49] Four local authority are located in the catchment of the Suir. The river's source is in the Devils Bit Mountains near Moneygall, County Offaly, and its tributaries flow mainly through counties Tipperary, Kilkenny and Waterford. Waterford City is the largest urban centre in the South East River Basin District (SERBD).

The Suir is not an atypical Irish river. Its catchment contains areas of upland and forestry, but it mainly drains average to good agricultural land used for livestock and some tillage. It is an important salmonid river and is valued highly for its angling, while not attracting the premium angling of rivers such as the Moy or Blackwater. It has some areas of adjoining wetland and alluvial woodland, but in common with most Irish rivers these are not substantial in size and do not play a very significant part in the ecological functions of the river. As with most of Ireland's rivers, the Suir is used for water abstraction, but mainly by the public authorities with uptake of water by agriculture or private industry being slight. Waterford Harbour is of modest value for commercial fishing and is of moderate, but not outstanding value, to wildlife. Tourism activity is slight within the confines of the estuary, but somewhat more significant along the adjoining coast at Dunmore East and Tramore.

The Suir is presented here, not as a case study, but rather to demonstrate how the values contained in the database, along with information from other sources, may be used in ecosystem services valuation, specifically to determine the consequences of any impact on water quality relevant to the Environmental Liability Directive. Each section is provided with the same subheadings, namely:

1) sources of information,
2) nature of the ecosystem services
3) values for remediation

Many ecosystem services tend to be location specific, i.e. dependent on specific underlying processes at a particular location and the societal choices and values of the local population (Haines-Young and Potschin, 2009). Species and ecosystem processes may be different at another location despite similar characteristics. Therefore, while societal values can be transferred from another similar location, they could still vary due to differences in respective preferences, not all of which will be identifiable.

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The chapter provides an introduction to the practical issues in identifying the relevant or key ecosystem services and an example of the procedures for undertaking a valuation. It does not presume to identify all ecosystem services in the study area, nor does it provide a complete valuation of these ecosystem services. While much has now been written on ecosystem services and ecosystem service values, this research still has to be operationalised into policy. This will be a challenging transition.

5.2 Ecosystem services

5.2.1 Supporting and regulating services

Introduction

The habitats along the River Suir and its tributaries have an important supporting ecosystem service role in sustaining a healthy freshwater aquatic ecosystem and in supplying water to wetlands and connected habitats downstream. The aquatic ecosystem in turn regulates water quality through the river's capacity to 'self clean' so that, below particular thresholds, the river is able to provide a pollution sink service by adjusting to inputs of pollutants, nutrients and sediments. The same functions are performed by wetlands from which water outflows into the catchment.

Nature of the ecosystem service

As with many Irish rivers, nutrient levels and eutrophication are a cause of concern along sections of the river. However, it is the elevated nutrient status of the Cabragh Marshes near Thurles that attracts wildfowl. These nutrient levels are a consequence of past human activity in that they were raised at the time when the former beet factory discharged into the tributary. Human impacts are also evident at Kilsheelan Lake which is one of only two sites in Ireland known to support carp which generally requires warm temperatures. Although an introduced fish, the NPWS ecological survey notes that interesting invertebrate fauna could be associated with the species.

The Lower Suir SAC (site code 002137) is designated for a suite of Annex I habitats (alluvial and yew woodland, floating vegetation, old oak woodlands, eutrophic tall herbs, Atlantic salt meadows and Mediterranean salt meadows) and Annex II species including Sea Lamprey, River Lamprey and Brook Lamprey, Freshwater Pearl Mussel, Crayfish Twaite Shad, Atlantic Salmon, Otter.. Alluvial woodland is declining in Ireland and the site contains best examples of this habitat at Fiddown Island south of Carrick-on-Suir. Floating river vegetation can be found along sections of the Suir, notably along the unmodified Aherlow River and the Mulleen River.

From an ecosystem services perspective most of these wetlands are of supporting ecosystem service value for protected rare, unusual or remnant flora and fauna or as breeding, wintering and migratory locations for wildlife generally. Potentially, the wetlands have an option value as a refuge or migratory staging post for species in the event of climate change. A given density of wetlands could permit species to move in the event of a deterioration in habitat elsewhere, particularly given the linkage provided by the river. Cabragh Wetlands and Coofin Marshes would be amongst the best for wildfowl with the latter supporting a population of wintering Greylag Geese. Cabragh would be regularly visited by small numbers of people for birdwatching.

Sources of information

The **SERBD** is a primary source of information on water quality, habitats and water uses. SERBD contains 672 river water bodies and, after the River Shannon, two of Ireland's longest rivers (the Barrow and the Suir). There is no central database of wetlands in Ireland, although a Map of Irish Wetlands is currently being compiled as a single portal at **Wetland Surveys Ireland** (http://www.wetlandsurveysireland.com). The website is intended to show the location, conservation status and protection status of all the nation's wetland areas. The database currently contains more advanced inventories for the counties of Kildare, Lough, Monaghan and Clare. Specific information on nature conservation sites and some other habitat and species datasets are available from the **NPWS website**.

From Thurles, along the Suir, the WSI website lists wetlands at Cabragh, Gortroe Fen, Marlfield Lake, Kilsheelan Lake, Tibberaghny Marshes (reedbed and alluvial woodland) on the Suir floodplain, Fiddown Island in the River Suir (alluvial woodland) and Coolfin Marshes (wet fields). No wetland SACs are listed on the NPWS website, although the Devils Bit and Galty Mountains form an SAC and would have an influence on water quality as part of the Suir catchment. Philipston Marsh SAC is listed on the NPWS website for which a vegetation report is available (Lockhart, 1992). However, this marsh and fen drains into the Mulkear River rather than the Suir discussed in Chapter 3. Slievenamon Bog is a NHA comprised of high quality upland blanket bog, albeit with some disturbance from conifer plantations and windfarm road development (NPWS site synopsis).

Values for remediation

This aquatic vegetation along with much of the fauna is most at risk from river-borne pollution, especially elevated levels of nutrients and associated eutrophication. The impact of this pollution could be quantified in relation to cultural ecosystem service value attached to the river as a visual feature of the landscape, as an amenity and as a habitat for wildlife as these attributes are valued by human beings. None of the key wildlife species has a provisioning value and even most salmon caught by anglers are typically returned to the river (the seasonal bag limit is just five fish). Some familiar species such as otter or salmon can be expected to have an existence value related to cultural ecosystem services values as so-called charismatic species.

The limitation of the cultural valuation approach is that incidents of pollution will most often only be perceived by the wider public once they reach significant level that is visible, that reduces familiar wildlife species or place human health at risk. Protection against lower level pollution is only likely to attract a significant public good value if the impact on aquatic wildlife is explained within the information accompanying a stated preference survey. Information provision has to be dealt with very carefully to avoid leading survey respondents into expressing high values that do not necessarily reflect the value levels that they might typically attach to environmental quality. For instance, the pearl mussel may be valued as a rare species deserving of protection, but the anthropocentric benefits of its filtering effect on water quality are negated by its association with low nutrient status rivers. Ironically, introduced zebra mussels have been more effective in contributing to improvements in water quality, for example for angling in Lough Derravaragh, County Westmeath, although their ultimate net environmental impact in degraded locations is as yet uncertain. As noted previously, this is an illustration of the limits of an ecosystem services approach in that many species, including species that are now more scarce, have evolved in low nutrient environments where their regulating value is less valuable from a human perspective except in terms of helping to maintain a low nutrient status for other valued species.

There are possible regulating ecosystem services associated with waste assimilation by the wetlands noted above where these are adjacent to or fed by the river. However, much of the inflow and flooding occurs during the winter when microbial activity is low. Potentially, the alluvial woodland has a regulating value in terms of the filtering effect on water quality as well as benefits in terms of moderation of run-off and floods.[50] Nevertheless, none of these sites are large enough for these benefits to be realised. No significant monetary value can be assigned to any of these ecosystem services at this stage. Rather, the wildlife, its habitat and the associated landscape has a cultural value that could only be ascertained through a primary research, namely public surveys, or though targeted benefit transfer.

[50] A preliminary study of the Suir for the South East CFRAM is not yet available.

5.2.2 Provisioning services

Introduction

The provisioning service value of water can be identified by the use of the cost-based production function or netback method to estimate the value of water used in the production of final goods and services, including as drinking water and for agriculture and industry. In the first instance, it is necessary to identify water abstractions, specifically surface water abstractions. Each RBD has compiled its own register of abstractions as part of their finalised 2009 Management Plan. However, there is limited information available on abstractions in Ireland other than for public water supplies. While this restricts the valuation of water across all users, the majority of known surface water abstraction is for public water supply (WFD Ireland, 2005). Users of public water supply are defined as domestic or non-domestic users. Domestic use generally refers to household use and is not charged. Non-domestic generally refers to business use, but includes a range of different users such as trades, agriculture and hotels. An estimated 35-47% of public water supply is used by non-domestic users (OECD, 1999; CDM, 2004).

Nature of the ecosystem service

The provisioning value of water derives from both its availability for various uses and its quality. The two may be difficult to separate as the value of the latter depends on the use to which the water is put, with evidently high quality needed for human consumption, for food processing or certain industrial activities. This source water quality is managed by the ecosystem through regulating services. Where source water quality is good this reduces the cost of water treatment, an issue that was discussed in Chapter 3. The terrestrial ecosystem also has a role in supplying water, i.e. by capturing water to recharge aquifers or by channelling it into streams and rivers. This ecosystem service could be at risk from incidents that are covered by the ELD such as land clearance. Excessive abstraction of ground water could also reduce the availability of surface water for the aquatic ecosystem. In this respect, the following section illustrates the importance of water as a resource noting also that availability and quality will not always be entirely separable.

Sources of information

Abstraction registers are maintained by the three county councils (Tipperary, Kilkenny and Waterford) and by Waterford City Council (due to be amalgamated into the County Council) that border the River Suir. The information in these registers is limited to abstractions for public water supply. However, for some of these local authorities, *Ecorisk* found that information was not available, provided for different reporting years or had been compiled using different methods, e.g. different categories of non-domestic users. A range of abstraction sources are included in these registers, but for the purposes of this case study the focus is on surface water. There are no direct public abstractions from the River Suir in any of the four public water supply abstraction registers so the information presented includes a range of surface waters across each local authority area.

Abstractions are used for public water supply which is used to supply both domestic and non-domestic users. Only non-domestic users pay water charges. In principle, the price paid by such users provides an indicative value of water to these users. While domestic users do not pay directly for water, central government provides a large annual subvention (funded from general taxation) to local authorities to fund water services. This too could be used to provide an indicative value of water (via public water supply) to the domestic sector.

The abstraction registers do not provide details of abstractions other than those for public water supplies, but a number of industrial users have been identified. Under the Local Government (Water Pollution) Regulations 1978, local authorities are required to maintain a register of private water abstractions greater than 25 cubic metres per day. However, this register does not seem to have been actively compiled by any of the four local authorities.

In 2007 the national register of abstractions identified 46 surface water abstractions for public water supply in the SERBD, with 115,440 m^3 of water abstracted per day. The SERBD reported the lowest total surface water abstraction and highest groundwater abstraction across the seven river basin districts (RBD).[51] Of the four local authorities, Waterford City receives its water from the County Council while Kilkenny is located alongside a tidal section of the river (although water is taken from one of the river's tributaries).The percent of abstractions from surface waters for public water supply range is 71% in South Tipperary.

Table 5.1a. Registered water abstraction from the Suir: Volumes and source

Council	Total Registered Water Abstraction* (m^3/day)	Surface water (m^3/day) (% of total)	Direct Abstractions from River Suir (m^3/day) (% of total
Waterford County	NA	NA	NA
Kilkenny	NA	NA	NA
South Tipperary	45,441	32,469 (71%)	0

* Average daily abstraction rate 2013

Table 5.1b. Public Water supply: Domestic and non-domestic

Council	Total Public Water Supply (m^3/year)	Domestic volume (m^3/year) (% of total supply)	Non-domestic volume (m^3/year) (% of total supply)
Waterford City*	19,885	16,035 (80%)	3,850 (20%)
Waterford County	NA	NA	NA
Kilkenny	NA	NA	NA
South Tipperary	NA	NA	NA

* 2011

Table 5.1c. Public Water supply: 2011 Charge

Council	Water + Wastewater (m^3)	Water (m^3)	Wastewater (m^3)
Waterford City	€2.35	€1.15	€1.20
Waterford County	€2.66	€1.06	€1.60
Kilkenny	€2.89	€0.96	€1.93
South Tipperary	€2.00	€1.05	€0.95

In the absence of a complete national register of abstractions, the local authorities were asked to identify significant operations that were known to them and this information was supplemented with data from the EPA on IPPC licensed facilities located close to the Suir. The majority of IPPC licensed facilities source water from groundwater and/or public water supply rather than surface water which is the focus of this case application. The volumes for such groundwater abstractions are included in Table 5.2. They are not directly relevant to Ecorisk given our focus on surface water bodies, but do indicate the scale of unregistered abstractions. For example, in 2011, Bulmers in Co. Tipperary abstracted 443,508 m^3 of groundwater, equating to approximately 3% of the estimated total water abstracted for public water supply in South Tipperary in 2013.

[51] Eastern River Basin District, Abstraction Pressure Assessment *Background to Water Matters Report – 22 June 2007*

Table 5.2. Significant non-registered abstractions

County	Sector/activity/use/company, estimated volume)
Waterford	SmartPly Europe Ltd., Belview, Slieverue, via Waterford Water is sourced from on-site groundwater. Water usage data was not presented. (Annual Environmental Report 2011)
Waterford	ABP Waterford, Ferrybank, Co. Waterford Water is sourced from off-site groundwater. In 2011 total abstraction was 212,795m^3. (Annual Environmental Report 2011)
Waterford	Waterford Brewery, Mary St., Waterford Water is sourced from on-site groundwater (73%) and municipal supplies (275). In 2009 total water consumption was 210,463 m^3. (Annual Environmental Report 2010)
Kilkenny	Dawn Meats, Grannagh, Co. Kilkenny, Kilkenny Water is sourced from on-site groundwater and municipal supplies. In 2011 total water consumption was 144,274m^3. (Annual Environmental Report 2011)
Tipperary	Bulmers, Clonmel, Co. Tipperary Water is sourced from groundwater (99%) and municipal supplies (1%). In 2011 total water consumption was 449,866m^3. (Annual Environmental Report 2011)
Tipperary	ABP, Cahir, Co. Tipperary Water is sourced from on-site groundwater. In 2011 water consumption was 190,246m^3. (Annual Environmental Report 2011)
Tipperary	Merck Sharp and Dohme, Clonmel, Co. Tipperary Water is sourced from the river Suir. In 2011 water consumption was 734,745 m^3. (Source: Direct contact)

The only significant surface water abstraction is by Merck Sharp and Dohme (MSD). Water is abstracted from the River Suir and treated on-site. After treatment to potable grade standard the water is used in various applications before discharge back to the river after treatment in the company's waste water plant. These volumes are presented in Table 5.3, but should be put in context. For instance, the average daily abstraction rate from surface waters for public water supplies by Tipperary South County Council in 2013 was 32,469 m^3, or approximately 16,586,000 m^3 p.a. The flow rate of the receiving waters (WWTP discharge) is 7.25m^{3}per second (Dry Weather Flow) and 11.625m^3/second (95%ile flow).

Table 5.3. Water abstraction by Merck Sharp and Dohme

Year	**Direct water abstraction from Suir** (Annual: m^3)
2009	865,684
2010	622,300
2011	734,745
2012	749,160

The CDM economic analysis of water use in Ireland (2004) referenced in Chapter 2 allocated values to water use based on output from agriculture and industry, i.e. gross valued added and annual values, including for the SERBD. Water use values relate to abstraction on a per-unit output basis, in-stream water use and other water use values associated with designated conservation areas and non-use values informed by overseas data. Some uses, such as river transport, are not included in the estimates. Furthermore, information specific to the Suir could not be estimated. Although data was compiled for 2004, this could be updated.

Table 5.4: Estimated Water Use Benefits for Key Sectors:

	Gross Value Added (€)	
A. Estimated Economic Impacts	***National***	***SERBD***
Agricultural	1,323,293,567	296,097,207
Industrial	31,971,000,000	4,321,121,052
	Annual Value of Water Use (€)	
B. Estimated Water Uses and Respective Values	***National***	***SERBD***
Agricultural	122,991,821	28,049,653
Industrial	75,374,122	10,972,574
Domestic	201,565,415	30,339,197

Source: CDM (2004)

Value of water to agriculture

A netback approach can be used to value water as an input based on final output values minus non-water input costs. The method is described in Chapter 3. In practice, little use by Irish agriculture is made of abstraction from rivers. Where abstraction does occur, this is mostly from groundwater with approximately10% supplied by mains water (Hess et al., 2012).

There is no national register of surface water abstractions for agriculture and there is no information on direct surface water abstractions from the River Suir. Nevertheless, despite the absence of detailed information and the limited dependence of Irish agriculture on surface water, it is still useful to demonstrate the value of surface water abstractions.

Transferred values from the (Moran and Dann, 2008a) study need to be adjusted to reflect the unique characteristics of Irish agriculture.

- Approx. 40% of the Irish potato crop is capable of being irrigated and where irrigation takes places this is primarily for fungicide application[52].
- In 2010 the CSO Census of Agricultural Production (www.cso.ie) identified 1,560 farms growing a total of just under 12,198 hectares of potatoes nationally. In Waterford, 69 hectares of land were allocated to potatoes, equating to 28 hectares of irrigated land (assuming 40% is capable of being irrigated and this area unchanged since 2010).
- In 2010, UK National Statistics (www.defra.gov.uk/statistics) estimated an average annual volume of water applied for irrigation for main crop potatoes of 1,060m^3)/ha.
- Assuming average volume of water use is similar to that in the UK and similar UK water use values per m^3 apply (between £0.23 and £1.38 per m^3), the estimated potential annual value of water to the potato crop in Waterford is between €8,055 and €48,330.

[52] In conversation with Michael Hennessy, Teagasc.

Table 5.5: Value of water for irrigation

	Value of water for irrigation (m^3) UK £ (2003)	**Value of water for irrigation (m^3)** UK £ (2013) *	**Value of water for irrigation** (m^3) € (2013)#	**Value of water for irrigation in Waterford~** (annum)
Agriculture: Potatoes	0.23	0.31	0.37	€10,871
Agriculture: Potatoes	1.38	1.84	2.14	€64,000

* Inflation adjustment based on Bank of England CPI data, rounding to two decimal place.
# Exchange rate as of **xx** March 2013 (See: http://www.x-rates.com/). ~ (1,060m^3/ha/yr * 28)*€0.27 and €2.14

The values identified in the table above are indicative values only. A valuation of water for use in agriculture requires a more complete understanding of all abstractions, i.e. surface and groundwater, which can then be combined with detailed information on agriculture costs to identity the value of water. Water would also be consumed by livestock for which potentially replacement costs of piped water provision could be estimated, although most of this consumption would be from small tributaries rather than the Suir itself. These estimations are beyond the scope of this case application but could in principle be undertaken if a specific assessment is required.

Value of water to industry

A similar method to that employed above for agriculture can be used to identify the value of water to industry. This relies on the transfer to Irish industry of values identified in the (Renzetti and Dupont, 2003a) study that was described in Chapter 3. This provides an estimate of the marginal value of water to different industries, but in the absence of information on the volume of water use does not provide a total value of water. However, the validity of applying these values from Canada to the industries found along the River Suir is doubtful given differences in the relative nature of production and water use even prior to considering the changes in water use efficiency that are likely to have taken place in the period since 1991. The sectors used in the Canadian study (Standard Industrial Classification) are also not directly comparable to those in Ireland, i.e. NACE.

Allowing for these limitations, the table below identifies some indicative values for IPPC licensed activities located along the river Suir. Table 5.6 indicates no clear association with the imperative of using high quality clean water in the production process relating to food or beverages. Only MSD draws water from the Suir for which the estimate is €55,100 before consideration of the proportions used for general purposes or in the production process itself.

Table 5.6. Indicative value of water to industry

Firm	**Sector**	**Annual Water Consumption (2011)**	**Marginal Value of Water** (per m^3) € (2013)
ABP Waterford	Food	212,795 m^3	0.022
Waterford Brewery	Beverage	210,463 m^3*	0.044
Dawn Meats	Food	144,274 m^3	0.022
Bulmers	Beverage	449,866 m^3	0.044
ABP Cahir	Food	190,246 m^3	0.022
Merck Sharp and Dohme (MSD)	Chemicals	734,745 m^3	0.075

* 2010

Value of water use to households

Households do not pay water charges, but local authorities' water services are directly funded by central government. Information on water services income for each of the four local authorities is presented below.

National water pricing policy provides for full cost recovery without profit, including capital, operation and maintenance costs through non-domestic charges. The application of this full-cost recovery varies across local authorities and there is likely to be varying cross subsidisation between domestic (central government funding) and non-domestic (water charges) users. Moreover, the price applied to non-domestic users does not reflect true use value. The information below should be viewed in this context.

Table 5.7. Indicative value of water to households

Council	Income	
	Government Grants	***Goods & Services****
Waterford City	€100,000	€3,904,848
Waterford County	€1,029,305	€4,144,680
Kilkenny County	€1,309,100	€4,660,400
South Tipperary County	€789,780	€5,820,525

* Includes commercial water, commercial waste water, superannuation, Agency Services & Repayable Works, Local Authority Contributions, other income. Sources: Kilkenny County Council adopted budget 2013, South Tipperary County Managers Draft Budget 2010.

Values for remediation

Liability issues could arise where a reduction in water quantity or quality impacts on the use of water for agriculture, industry or public consumption. This could impact on treatment costs or direct use. In the case of the latter, the netback approach could be applied, but requires primary data rather than unreliable transfer values from secondary sources such as those discussed above. The actual impact could be greater in that an absence of usable water could impact on a farm or company's ability to produce a final product. However, alternative sources would likely be available so the actual cost is more likely to that of switching sources. In any case, most agricultural and industrial water in the south-east is taken from groundwater aquifers. An interaction exists between groundwater and surface water in that excessive use of the former could impact on the availability of water for aquatic ecosystems particularly in the event of a drought. Changes in land use or surface vegetation could lead to greater interception of water or release of water to evaporation and reduce ground water recharge.

5.2.3 Regulating services

Introduction

Current water charges do not capture either the marginal cost of supplying water of satisfactory quality or of protecting the regulating ecosystem service. At present, charges only represent an annual charge set by local authorities at the beginning of each year that reflects expected annual average costs. In principle, water quality is related to ecosystem services though the regulating service provided by aquatic ecosystems in terms of their capacity to assimilate waste or to "self-clean". As discussed in Chapter 3, this service is performed by species found at various trophic levels and is facilitated when the oxygen content of water is raised, for example by natural cascades or by weirs. It is an intermediate service that reduces the risk to health from water-borne pathogens encountered when consuming fish or through direct contact recreational activities. However, it also reduces the degree of water treatment needed when water is abstracted as a final good. The river's assimilative capacity provides the same kind of regulating service (but in reverse from a human perspective) by reducing the treatment required when waste water is returned to the river.

The WFD requires the water quality of the receiving river to be preserved or improved to good status. These objectives, while established in an institutional context by policy, are a measure of society's willingness-to-pay as indicated by the amounts spent on water treatment. If disproportionate costs are identified (and accepted by the Commission) then these could be interpreted as defining a threshold to society's willingness to pay.

Sources of information

Some local authorities are more advanced than others in the implementation of full-cost recovery. This section examines the value of regulating services via the **averting expenditure** or the **replacement cost** method. First, the cost of the rising level of treatment needed to maintain the quality of drinking water in the face of a deterioration at source. Second, changes relating to waste water treatment costs to infer the value of waste sink/assimilative capacity.

For water treatment at source, the quality of the output (drinking water) is defined in terms of drinking water regulations and standards. For wastewater. treatment is required to a level that would not affect the quality of the receiving waters or their assimilative capacity. A change in the cost of water treatment at source following a reduction in raw water input quality can be used an indicative value of the lost ecosystem purification services. Alternatively, a value can be provided by expenditure on catchment management programmes focused on maintaining and improving source water quality.

The presence of defined water quality standards for receiving waters for wastewater discharges recognises the needs of both other uses of the resource and the limitations of regulating ecosystem services. Discharge licences take account of the ability of receiving waters to assimilate wastewater in such a manner that other uses are not impacted. The licenses typically relate to defined quality standards for specific uses, i.e. discharges are licensed once it can be demonstrated that the discharge can be assimilated and, for example, bathing water quality standards maintained. Changes to wastewater treatment costs can reflect either a change to required water quality or changes to the demands placed on the receiving waters. For example, additional treatment may be required to remove specific pollutants from wastewater discharge following the development of a new drinking water abstraction point downstream. Alternatively, the ecosystem may be unable to meet existing quality standards as additional pressures (point and diffuse pollution) have caused the assimilative capacity to be exceeded. In this instance additional treatment of wastewater discharges will be required to reduce overall levels of pollutants to a point where they can be assimilated and the waters return to existing water quality standards.

The new wastewater plant for Waterford City is located on the Lower Suir at Gorteens on the edge of the freshwater and transitional environment. It is designed for a current residential population equivalent of 47,000 and a projected population of 68,600. On the basis of Irish estimates of daily wastewater output per person of 225 litres (EPA, 1999) and UK studies for similar sized urban areas, wastewater from the current population would amount to 225,000 litres per day or 82 million litres per year. In addition, commercial premises are responsible for additional discharges bringing the total population equivalent to an estimated 143,550 in 2012 or an output discharge of 250 million litres. Sludge removal is estimated at 95,000 tonnes per year. Celtic Anglian Water International manages the Waterford plant. In the UK, the company reported operating costs to Ofwat (the British water regulator) for its divisions of 66p per thousand litres of treated sewage in 2009/10. This could imply costs in Waterford of €165,000 per year. A more precise estimate would require information on the local level of treatment required to meet environmental criteria.

Image 5.1 Waterford Wastewater Treatment Plant

Source: Waterford City Council

Costs would rise were the wastewater plant required to achieve higher standards to avoid damage to the receiving environment. Biological Oxygen Demand (BOD) measured in receiving waters provides an estimate of the amount of organic matter, or effluent, in wastewater. The Urban Wastewater Treatment Regulations set a limit of 25mg/l in sewage outfall. The removal of organic material related to achieving this limit was estimated at 8,613kg per day for the new Waterford plant as of 2013, although actual removal has tended to be less than 5,000kg per day (Mott McDonald, 2013). Removal costs depend on plant size, the degree of treatment and the latitude permitted by the assimilative capacity of the receiving environment. On average, for plants sampled for Ofwat, a 4% increase in costs in realised for a 10% reduction in BOD (Oxera, 2006).

Table 5.8 demonstrates the costs involved based on the Ofwat sample (Oxera, 2006). The organisation acknowledges that these examples of costs may not apply to all plants. The loadings in the Waterford plant have rarely exceeded 15mg/l raw sewage and are generally around 10mg/l. This suggests that BOD reduction in the Waterford plant is costing around 63c (£0.50) per kilogramme. On this basis, the treatment cost for BOD removed from the Waterford plant would be in excess of the figure given above and equivalent to €1.2 million. Were a 10% reduction to be required to protect the receiving environment, these costs could rise to by up to €740,000 per year.

Table 5.8 Cost of removal of BOD per kg (£/kg)

	10 mg/l			15 mg/l		
Plant size	2.5% lower bound	Central estimate	2.5% upper bound	2.5% lower bound	Central estimate	2.5% upper bound
<500 pe	8.7	13.0	19.4	7.6	10.3	13.9
500-4,000pe	4.1	5.5	7.5	3.5	4.4	5.4
4,000-50,000pe	1.0	1.3	1.8	0.9	1.1	1.3
Pe > 50,000	0.3	0.5	0.7	0.2	0.4	0.6

Pe. = Population equivalent. Source Oxera c/o Ofwat. Oxera cautions that results were based on a limited number of companies and only a minority of plants serving more than 50,000 population equivalent.

In addition, excess ammonia also needs to be removed to avoid additions to existing environmental levels of 0.15 and 0.22 mg/litre. Some forms of ammonia have the capacity to be toxic to invertebrates and fish in the marine environment (Seager et al., 1988; Nison et al., 1995) at levels above the EQS of 0.021 mg NH_3-N1^{-1}. The potential for environmental impacts is higher in locations of low salinity and low oxygenation as could apply to the parts of the Suir. BOD and ammonia are often treated together, but this process does involve some cumulative costs. Oxera (ibid) supply a similar table to that above for ammonia for which median estimates of costs for plants with a population equivalent equal to Waterford are £4.10, £3.30 and £2.60 per kilogramme removed for loads of 3mg, 5mg and 10 mg respectively. Incoming ammonia levels at the Waterford treatment plant have been below 1.0 mg/day and so the removal of nitrates could be estimated to average €2.60 per kg for the relevant population equivalent. The removal of other nitrates, while based on a limited sample, is estimated to cost £2.20 for plants with a population equivalent of over 100,000.

Phosphorous is not addressed in the Oxera report. Phosphorous can be removed either chemically or (at more expense) biologically, but would typically be undertaken only with tertiary treatment. Grease or maximum coliform counts are not addressed in the report either, although it was expected that the Waterford plant would reduce the latter to counts of 5,034 per 100 litres. Although subject to a wide range, this is an improvement of the prevailing maximum of 33,238 per litre. This reduction has particular benefits to aquaculture and shellfish consumption.

Although wastewater standards are founded on policy and political consensus over what would represent acceptable or disproportionate treatment costs, measures of nutrient pressures may go unnoticed by the public until they reach levels which have a clearly deleterious impact on wildlife, recreation or aesthetics.

Values for remediation

The averting expenditure method can be used to estimate the costs of additional treatment of water at source or of wastewater treatment following an environmental incident. The two forms of treatment measure rather different things. The cost of treatment at source responds to environmental impacts, but these would need to be sufficient to cause the contamination to exceed the threshold at which water must normally be treated for purposes of protecting human health. The source of the contamination could relate to terrestrial impacts as much as direct impacts to the aquatic ecosystem. The cost of wastewater treatment reflects both changes in the nature of the wastewater, but also the need to raise treatment levels to reduce the risk to surface water bodies from an environmental impact. Capital costs should be included in the equations if the damage is expected to be persistent and significant compared with routine water treatment costs.

A practical problem is obtaining information on water treatment costs from local authorities. In the first instance, the type of contamination and the response (physical or chemical) required for a specific incident should be understood. However, most local authorities contacted for this project were either unwilling or unable to provide an estimate of these costs. Work loads in preparation for the transfer of responsibilities to Irish Water were a factor in this respect, but it also evident that this type of information is not readily available.

In both cases, the cost of treatment does not reflect the full welfare value of protecting the aquatic ecosystem and its natural regulating service. However, standards are set by the WFD. Ultimately, the acceptability of these costs depends on public support and willingness to pay. If the costs of treatment are argued to be "disproportionate" then this could be an indication that the public's willingness-to-pay (or at least water services managers' assessment of their willingness-to-pay) for infrastructure to maintain or improve water quality has been exceeded.

5.2.4 Cultural services

Angling

Introduction

The River Suir is an important salmonid river and, as such, is valued by anglers. It is particularly favoured for trout. At one time, salmon would have been commercially caught by draft net fishermen on the lower reaches of the river, but since 2002 this activity has been restricted nationally to catchments identified to have surplus stocks. The River Suir, along with the Nore and Barrow, are now only open on a catch-and-release basis, i.e. of value for recreational angling.[53]

Nature of the ecosystem service

The overall good status of water quality, and the presence of suitable habitat both within and alongside the River Suir, provide the river with the potential to sustain a good salmonid population. A survey of the river on behalf of the Southern Regional Fisheries Board (SRFB) (O'Grady and Delanty, 2006) presented a positive assessment of the river's status concluding that there were good stocks of brown trout and salmon at densities comparable to the previous 2004/05 survey. In addition, it found that there is a widely distributed (but small) eel population. The report classifies the river into three segments on the basis of width, flow and salmonid potential. The three segments are upstream of Thurles, Thurles to Ballycamus and downstream of Ballycamus. It comments that compared with the 1980s there has been a significant improvement in the ecological status of the river between Thurles and Ballycamus due to a new wastewater treatment facility and the closure of a meat processing plant. Fish stocks were found to be good downstream of Ballycamus, particularly for trout for which the habitat is most suited. Overall, the survey indicted a capacity for higher catches, but with the potential for further improvement given the fundamental characteristics of the river.

Sources of information

The main source of information about fish stocks is maintained by Inland Fisheries Ireland and specifically the SRFB based in Clonmel.

Identifying liability

Although the river is agreed to be in generally good condition for fisheries, the SRFB report does acknowledge a continuing risk from pollution, specifically inflows with high nutrient levels. The observation is made that the trout population has not much recovered its potential despite the upgrading of sewerage works and suggests that this could possibly be due to occasional discharges from the plant. The report also observed evidence of serious pollution from wastewater at Loughmore in the upper catchment.[54] Water quality issues associated with inadequate wastewater treatment were also observed on the Moyle tributary at Lisronagh and the Ara River at Tipperary. Diffuse pollution is not discussed, but is an issue given the presence of much productive farmland in the catchment. The presence of pollution from diffuse sources reduces the margin of flexibility available to discharges from waste water treatment plants. Diffuse pollution is not addressed by the ELD, although point discharges of nutrients, for example, from animal housing or intensive chicken or pig enterprises is covered.

Water quality is not the only factor impacting on fisheries and so the source of impacts needs to be carefully identified. Other constraints on the angling value of the river discussed by SRFB report include a lack of habitat diversity in formerly drained parts of the upper and middle catchment. Additional problems are past drainage works, shading from dense tree cover and obstructions, mainly weirs. Former incentives to drain land have been reduced and are no longer available in the vicinity of salmonid rivers. Nevertheless, the report acknowledges that independently financed ill-

[53] In 2013, 94 rivers were open for angling/fishing, 32 of which are only open on a catch-and-release basis. 58 rivers are closed as they have no identified surplus population.
[54] Upgrading of the wastewater plant at Loughmroe was underway as of 2006 but at an early stage.

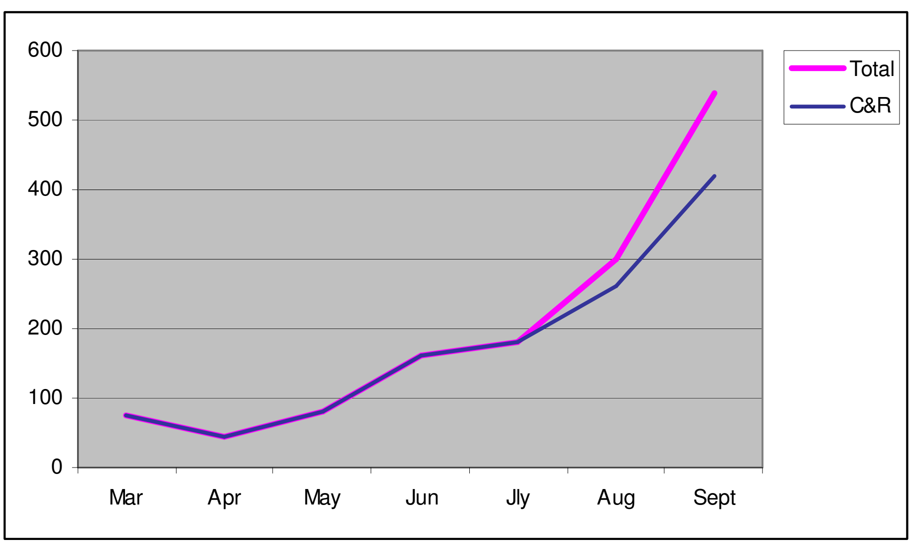
planned drainage could still have an impact as could the poaching of riverbanks by livestock in locations adjacent to spawning grounds.

Salmonids

The River Suir is renowned for game angling and has no coarse fishing. It is the fourth largest salmon river in terms of catch and accounts for around 4% of the total angling effort with an annual catch of around 1,380 in 2011 (of this number 10.6% are caught and released (see Figure 1)). The IFI estimate of spawning salmon has varied between just under 10,000 in 2006 to around 17,000 in 2007, but was 11,500 in 2010, the latest year for which data has been compiled (IFI, 2011a). The Conservation Limit is 16,482. However, the river is best known for brown trout for which the main angling season occurs between March and September. There is less statistical data for trout, but the SRFB report describes good carrying capacity along the river's full length.

Revealed preference, in the form of sales of rod licenses and permits, provides a lower bound indication of the value of angling. In the South Eastern Region in 2011 there were 299 annual, 881 district and 80 juvenile salmon rod license sales together with 151 21-day and 48 single day sales (IFI, 2011b). A licence does not confer the right to fish and local permits must be obtained from one of around twelve principal angling associations or clubs or from one of the several private fisheries. Typical permit prices are between €15-€30 per day or €75-€150 per week. These figures could therefore imply annual state licence revenue of €44,000 and permit revenue of €365,000 relating to salmon assuming that the Suir accounts for around 50% of salmon angling effort in the South-East. A licence is not required for trout, but assuming that similar angling effort is expended, the additional permit sales would bring the total revenue to over €750,000 per year. On the basis of the recent IFI survey (2013) (see Chapter 2), this level of sales suggests an annual angling value of over €5 million once expenditure on angling related activities (boat hire, gillies, etc) is taken into account along with indirect expenditure on accommodation.

Figure 5.1 Reported salmon catch by month on River Suir 2011 (IFI, 2011b)



Salmon: Economic estimates

Quantification of the economic impact of an incident relevant to the ELD would begin with an ecological assessment of the consequences for fish stocks. Possible incidents could relate to pollution or sedimentation affecting spawning grounds. For salmon, productivity is known to be impaired in waters of moderate to poor quality (IFI, 2011a).

There are spatial and temporal problems in estimating the impact. The interim losses could be significant, but take time to be realised. There would be a delayed response in the sales of licenses and permits with the reduced income likely to be greater for prolonged reductions in stocks. Impacts could be more apparent in the seasons following an event. An economic assessment would need to take into account the value of the angling before and after based on the angling response for past incidents or similar catchments. This comparison could also help to indicate the duration of interim losses before recovery occurs or of the merit of re-stocking. Longer duration impacts could even impact on the capital value of land and fishing rights. An economic assessment would need to be undertaken on the basis of local research given that the value of permits is not widely published and no register is maintained by IFI. An authority could also choose to take into account the value of lost business to the local economy. In many locations, there are businesses, such as guest houses, which draw much of their custom from the angling community.

In addition there are spatial considerations. The location of an impact could be different from that where the ecosystem services costs are experienced. Salmon could be most vulnerable to impacts upstream or during migration, but the losses from these impacts could be realised most at downstream angling locations. The assessment would therefore need to take into account the impact downstream, noting also the characteristics of the location and dependence on the spring or autumn run. The spatial discrepancy makes it important to scientifically link the location of the damage to the receptor site.

A spatial characteristic also applies to anglers themselves. The more valuable stretches of the river will be fished by tourists and well-heeled individuals. However, impacts on other stretches could affect the well-being of local or less affluent anglers and those with less opportunity to substitute with visits to alternative sites. There could be a case for applying weights to account for the values held by different socio-economic groups and the businesses that depend on them.

Image 5.2 River Suir

Source: Irish Fly Fishing

Eel

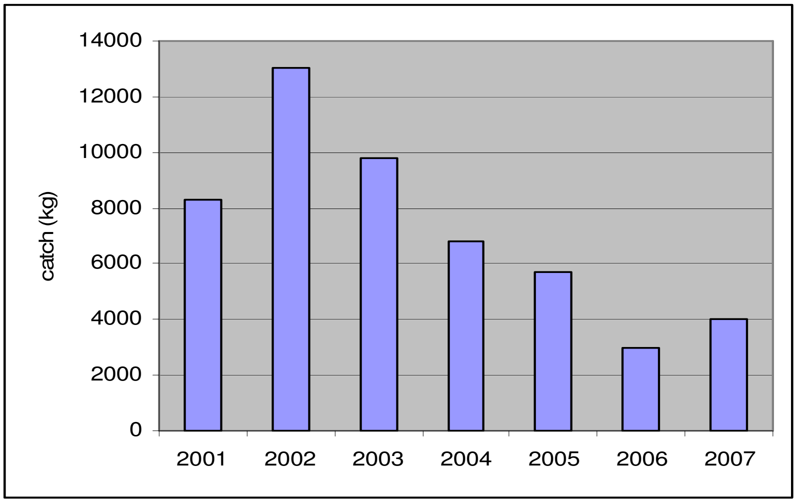
Although potentially a provisioning service, catches of eel are not currently caught for sale. The rivers Suir and Barrow, along with Waterford Harbour, supported an eel fishery prior to the national ban on the commercial fishing of eel (*Anguilla anguilla L.*) in 2009. The estuary yielded between 3.4 and 8.6 tonnes per year (average 4.9mt) between 2001 and 2008. Between 35 and 38 licenses were issued each year for brown eel of which around 20 were for the Waterford District. The majority were for baited pots laid on the lower Suir Estuary and four were fyke net licenses for the upper Suir Estuary. Research samples undertaken as part of a European-wide monitoring programme reported better catches from transitional waters of the Suir and Barrow (O'Leary et al., 2012). These provided a conservative estimate of eel density of 9-58 eels per hectare Eels were also caught in the Wexford District with the majority of the catch deriving from the Slaney Estuary which empties into Wexford Harbour where catches varied between 0.5 and 10 tonnes (averaging 3.8 mt).

Pristine escapement from the Suir catchment to the sea was estimated at by O'Leary et al (ibid) at 16 tonnes. Actual levels are around 9 tonnes. Across Europe eel populations have been falling due to a variety of reasons of which exogenous factors relating to changes in sea currents and climate are suspected to be significant. However, overfishing, morphological changes to river channels, diffuse pollution and the spread of the *Anguillicola* parasite are also reported as problems in the South East.

Hydrological damage or pollution of the kind addressed by the ELD may therefore be only one factor in the species' decline. Indeed, it is unclear to what extent poor water quality has an effect of eels. However, persistent organic pollutants do not appear to be elevated in eel samples (elevated dioxin levels were reported for the Burrishoole catchment in County Mayo indicated spot pollution at this location) (EPA, 2010d). A point pollution incident could potentially have a catastrophic impact if it were to occur at the wrong time due to the unhealthy state of the population and its confinement to particular catchments. The sensitivity and poor condition of the population means that even the closure of the Waterford fishery was expected to result in only a small temporary recovery before trends resume a downward to below 10% of pristine escapement (SRFB, 2008).

Altogether the catch from the Waterford and Wexford Districts accounted for 13% of the national catch of brown eel and 1.7% of silver eel or elvers. The national catch varied in ROI between 86 and 220 tonnes between 2001 and 2007 and supported the livelihoods of 150-200 part-time fishermen. The annual value of the catch was between €500,000 and €750,000 suggesting that revenue the Waterford and Wexford catches were €65,000 and €98,000. This revenue compares with an annual reported catch of 700 tonnes and turnover of £5.5 million in Ireland's largest fishery managed by the Lough Neagh Fishermans Eel Cooperative (although stocks are declining in Lough Neagh too, the fishery was exempted from the 2009 ban as recruitment meets the EU targets). Across Ireland, commercial eel fishing took place in 4.6% of catchments. Current levels of recruitment are low everywhere. Analysis of historic data indicates productivity of 0.9 to 5.5 kg/ha (the latter for the productive River Moy) providing for a previous potential total national production of 595 tonnes per year compared with current rates of between 1.3 kg/ha to 2.7 kg/ha. As a percent of historic production, escapement averages 24% compared with an EU target of 40% (DCENR, 2008). For these reasons the national ban on eel fishing was extended in December 2012 until 2015.

Figure 5.2 Reported brown eel catch in the SE RBD



Eel: Economic consequences

Although a catch ban is currently in place, eels are potentially a commercial species. Prior to the ban, catches from the Suir were well below potential given that national average recruitment is only half that of EU targets. Water and habitat quality have an evident impact on the health of the eel population, but exogenous factors are at least as important. A major pollution incident at the wrong time in the wrong place could impact seriously on the eel population. However, the immediate economic losses would be nil due to the ban. Based on the previous value of catches in Waterford and Wexford the value of eels moving into Waterford Harbour could between €32,500 and €50,000 at current prices with the Suir taking a share of this total along with the Barrow and the Nore. However, this is very much a minimum value given the higher catches that the river should support and the importance of eels to wildlife on the river, e.g. herons and egrets. Given the poor condition of the stock combined with the well-known attachment of eels to particular rivers, an incident, were it to occur in the wrong place or at the wrong time, could have an impact that extends to many seasons or even a catastrophic impact on the viability of the population that removes the prospect for future restoration of the fishery.

Values for remediation

It would be difficult to predict economic damage to the eel population given the continued downward trend in numbers away from levels that would permit a reopening of the fishery. Remediation should focus on primary restoration, ideally exceeding no-net-loss to account for the heightened vulnerability placed on the eel population by an incident.

For economic estimates of the impact of reduced fish stocks on angling, much ecological information would be needed on the consequences of an impact for the whole extent of the river and the time required for recovery. The minimum interim damage would be represented by revealed preference evidence of reductions in angler numbers during this period together with the consequences for landowners and businesses dependent on this trade. Perceptions would play a part in this response and evidence from previous events or similar catchments could be used to indicate the magnitude of the impact. There would also be impacts on the wellbeing of local anglers and local communities which should be taken into account through remediation exceeding no-net-loss.

5.3 Transitional and inshore waters

5.3.1 Supporting services

Introduction

Waterford Harbour is not an especially unique or exceptional habitat, although the estuary is relatively deep and allows for considerable mixing of fresh and saline waters. It provides important supporting ecosystem services and consequent benefits in terms of regulating, provisioning and cultural services. The estuary, along with Coolfinn Marsh, supports large numbers of migratory wildfowl and waders including Greenland White-fronted Geese, Greylag Geese, Lapwing and Curlew (NPWS Site Synopsis).

Nature of the ecosystem service

The richness of the local ecosystem supports large numbers of wintering and migratory birds. In turn, this provides a cultural ecosystem service for bird watching. The conservation status of White-fronted Goose is listed as 'amber' as the majority of its population winter at less than ten sites in Ireland including also the nearby Wexford Slobs and Tacumshin Lake. There has been less reference to Waterford as a wintering location for White Fronted Geese in recent Birdwatch Ireland on-line articles. The population has fallen significantly in recent years due apparently to poor breeding success in the Arctic. The reason for this is not known for sure, but could be a consequence of displacement by Canada Geese in or changes in spring grass growth due to climate change. This makes good wintering habitat, such as that found in Ireland, all the more important.

Lapwing and curlew have both declined dramatically as breeding species in Ireland due to drainage or transformation of wet meadows and loss of pristine blanket bog. In the 1988-91 Birdlife Ireland/BTO Atlas of British and Irish Birds, 5,000 pairs of curlew were reported from Donegal and Mayo where now only six sites have been found to hold breeding pairs. The curlew is globally threatened and therefore is a 'red' listed species. Irish breeding birds, at between just 100-200 pairs, are far outnumbered by the wintering population. Waterford Harbour has in the past held nationally important numbers of these migrants.

In addition to its habitat value for the more visible species, Waterford Harbour, in common with most estuaries, supports tremendous amounts of primary and secondary production and nutrient cycling. The microfauna provide for the regulatory services described below while supporting species higher up the food chain. Many of the macrofaunal species at the next trophic level are prey for the large migratory bird population. However, they also support fish stocks, both commercial and recreational. There is also a local population of grey seals. Common seals are sometimes seen in the area too.

The Lower Suir below Waterford City is an SAC. There are narrow bands of salt marsh habitat in the vicinity of Little Island and between Ballynakill and Cheekpoint. The marsh at Little Island is dominated by Atlantic Salt March species including twitch with an additional third of a hectare represented by non-annex *Spartina* grassland of value to wintering birdlife. Land around the designated area has become quite built up in recent years and the quality of the habitat in one area has been impacted adversely by infilling and the line of a sewage pipe. The site is notable for the scarce Meadow Barley and is typical of a brackish estuary salt marsh. Introduced Cordgrass is also present but does not appear to have spread too much at present. Although the salt marsh area is stable and may even have expanded in recent times, the overall NPWS assessment of the habitat is unfavourable, although the inventory does note that the status of the core Atlantic Salt Marsh is reasonably good.

Sources of information

The **NPWS**, including its local office, would be the main source of information. **Birdwatch Ireland** and local bird enthusiasts would also be able to provide some additional information.

Values for remediation

Given the importance of Waterford Harbour for species such as white-fronted geese and curlew any remediation following an incident should aim to exceed primary restoration or no-net-loss complementary remediation given the potential knock-on impacts during the interim period for breeding populations. Transitional environments provide intermediate supporting services, but remediation should also not lose sight of the value of the estuary's primary and secondary production for important regulating, provisioning and cultural services.

5.3.2 Regulating services

Waste assimilation

Introduction

The coastal ecosystem provides an important bioremediation service. The regulating service is efficient, but can potentially be overwhelmed and diverge from the levels of environmental quality that the EPA is looking to protect. Water quality could be adversely affected by human impacts, for example a toxic pollution plume that could either kill those organisms responsible for assimilating waste or reduce their capacity to perform the service so efficiently. Lower levels of pollution may also affect the balance of species types, but could still permit the eventual breakdown of the pollutants. In either case, an interim impact could accumulate in higher order species such as shellfish, finfish and birdlife of evident cultural or provisioning value to human beings.

Targets of environmental quality are set within the WFD. This is a regulatory constraint, although the targets are presumed to be within the bounds of environmental quality that people most prefer and value. Agri-environmental measures aim to manage diffuse pollution from farming. Likewise, the new regulations for septic tanks aim to reduce pollution. However, because diffuse pollution is difficult to control, the emphasis has been placed on point source pollution such as sewage outflows and wastewater treatment plants. If the quality and assimilative capacity of the receiving environmental needs to be protected against diffuse pollution - or in response to a single pollution incident - more stringent wastewater standards may be needed at higher cost.

Nature of the ecosystem service

Organisms bury and transform waste through assimilation and chemical re-composition. A minority of this waste is buried permanently removing carbon and nutrients in an era where human environmental impacts have caused these to enter the ecosystem in excess. In effect, the ecosystem performs a service that would otherwise be undertaken by wastewater treatment plants. Although there are now modern wastewater treatment plants, for years this service was only performed by the ecosystem. Even now, the ecosystem continues to have the effect of treating waste to tertiary level, i.e. a level beyond that which is currently performed by manmade infrastructure.

Sources of information

Information on transitional water quality is maintained by the **EPA**. The Lower Suir is listed among the sensitive tidal waters under national regulations (S.I. 254 of 2001 and S.I. 440 of 2004) implementing the Urban Waste Water Directive (EPA, 2007). As discussed in Chapter 3, the Trophic Status Assessment Scheme (TSAS) is used to assess the impact of nutrients contributing to eutrophication. For dissolved oxygen, the target threshold levels for tidal, freshwater and intermediate waters are within 70-130% saturation. Saturation levels below 60% are indicative of oxygen depletion while levels above 100% indicate quantities in excess of those needed for primary production. Levels above a threshold of 130% can result in supersaturation. Daily shifts between the two states can occur due to the effect of daytime phytoplankton photosynthesis prior to depletion at night.

On the TSAS index, the Suir and Barrow compare relatively favourably to Wexford Harbour and the Blackwater Estuary both of which are rated as eutrophic. In the 2002-06 EPA sampling period, BOD was reported to be at just acceptable levels of 4-6 mg per litre. By the following 2007-09 sampling period, BOD had declined to between 0-2mg/l on the Lower Suir but increased to 6-8.5 mg/l on the Middle Suir. In neighbouring catchments the situation was worse. On the Upper Slaney, Bandon and Blackwater Estuaries, BOD was elevated at 6-8 mg/l (by comparison BOD levels in the Swilly Estuary in Donegal levels had reached *adverse* levels of between 8-10mg/l.).

The Suir fares less well on the index for Dissolved Inorganic Nitrogen (DIN). During 2002-06, the Lower Suir breached thresholds for DIN in both winter and summer (EPA, 2007). These had fallen to 1.6-2.4 mg/l by 2007-09, but values remain at the threshold level. Indeed, nearly all the rivers on the south and south-east coasts exhibit excess nitrogen concentrations. All of these rivers have experienced reductions in phosphorous due to improved agricultural practice and sewage treatment. However, there has been less change in the use of fertilisers based on inorganic nitrogen and so environmental concentrations have proved more resilient. Nitrogen, particularly in the absence of the stimulus to algal growth provided by phosphorous, is increasingly being flushed out to sea where it has the more limiting role. In addition, both the Suir and the Nore had high winter concentrations of ortho-phosphate (MRP) in 2007-09 (EPA, 2010d).

As a consequence of this nutrient pollution, the upper stretches of the Suir and Barrow estuaries are classified as potentially eutrophic while lower stretches are intermediate. The Lower Suir is relatively deep (and dark) and the mixing of freshwater and saline waters means that phytoplankton has less time on the surface to photosynthesise.[55] The mixing of waters results in nutrients being flushed out to sea where the more limiting nitrogen causes growth in diatomaceous algae on the surface which then sink to form beds of organic matter offshore. This often covers fishing gear laid out in Waterford Harbour, but its full implications are little understood. Excess nitrogen coupled with reduced oxygen, particularly in the event of algal blooms (particularly *Karenia*), does present a risk of hypoxia to fish and to aquaculture.

In all these respects, the estuaries of the south-east coast each show distinct characteristics that are determined by physical factors of depth and mixing usually to a greater extent than ecosystem services. The Lower Suir, being deep and subject to regular flushing, in contrast to the enclosed and shallow Wexford Harbour, presents a greater risk from nutrients to the outer areas of Waterford Harbour and to the coastal environment. However, while the physical environment may have the greater influence, this results in a varying role for ecosystem services at different locations in the river, estuary and coastal areas. For example, the large mussel beg located in the relatively shallow Slaney estuary may be having a significant influence in the rapid spatial transformation of this waterbody from eutrophic to good status.

Values for remediation

Primary and secondary treatment was proposed for the expansion of the Waterford City wastewater treatment plant. This is having the effect of removing a considerable quantity of organic matter and some phosphorous, but more complete removal of phosphorous and nitrogen would really require tertiary treatment. The reduction in organic material is coincident with anticipated BOD removal of 94% equivalent to discharges of 0.04-0.08 mg/litre compared with previous rates of 0.4-0.12mg/litre. Environmental levels of BOD in the Lower Suir and the estuary (Cheekpoint) have been estimated by SERBD at between 1.55mg/litre and 2.05mg/litre for the period 2005-07 which is consistent with the EPA figure reported above for 2007-09. The previous level of discharges would

[55] *Pers comm*. With Shane O'Boyle EPA.

not necessarily have led to a eutrophic situation, but the capacity of these discharges to tip the balance towards eutrophication depends on the condition of the receiving waters at any one time or season. The new wastewater plant has made this risk more remote.

Although the load of nitrates in Waterford's sewage appears to be low, phosphates and nitrates are barely removed by the treatment process. Additional nitrates are discharged into the river from diffuse sources such as agriculture. As discussed above, the environmental levels of nitrates in the Lower Suir are equivalent to the threshold for eutrophication in freshwater and transitional environments. Consequently, while levels of BOD have become more acceptable, a pollution incident could easily raise the level of nitrate in the estuary to levels above the threshold at which eutrophication could occur. If the standard of water treatment needs to be raised to mitigate this impact, then additional costs of between €2.60 and €4.84 per kg removed could be incurred based on the Ofwat figures given in Section 5.2.3.

Averting expenditure methods could therefore be used to demonstrate the additional costs needed for waste water treatment to ensure that the effects of a pollution incident are neutralised. In the meantime interim losses may already have occurred for wildlife populations and for businesses that depend on high water quality such as fishing and shellfish, ecotourism and water-based recreation. Revealed preference methods could be used to examine the extent of these losses, although as in other examples listed above, it could take time until the public realise the extent of the impact and change their behaviour, for instance by choosing to visit alternative sites.

Image 5.3 Waterford Harbour

Carbon sequestration

Introduction

Carbon sequestration contributes positively to maintaining atmospheric carbon levels in the context of the impact of anthropogenic emissions on climate change. Dissolved organic carbon (DOC) can also be released, for example from cutaway bogs, but this too can find its way into emissions via the ecosystem or itself can have adverse impacts on the ecosystem functioning. As discussed in Chapter 3, wetlands can be either sequesters or carbon or emitters of carbon depending of the level of saturation and environmental factors such as temperature. Many plant communities found in the estuarine and inshore environment have a positive regulating value due to net carbon sequestration and/or storage as well as for the bioaccumulation of harmful heavy metals.

Nature of the ecosystem service

Chapter 3 notes that the regulating value of salt marsh for carbon sequestration has been estimated at between 0.64-2.19 t C $ha^{-1}yr^{-1}$ (Cannell et al., 1999; Angus et al, 2012). There is a rather small area of salt marsh in Waterford Harbour of up to 8 km^2. On the basis of the above estimates, this marsh would be sequestering between 500 t C and 1750 t C per year with a value of anything between €10,000 and €87,600 depending on the method of monetisation used to value emissions. However, inspection of the area would suggest that much of what is broadly described as salt marsh actually more closely resembles periodically flooded meadow.

A carbon sequestrait6on service is also performed by kelp, but the area of kelp is small and located in the outer harbour. There do not appear to be any sizeable areas of seagrass in Waterford Harbour. More detailed field data would be needed on the area of mix of inter-tidal mud and sand than can be acquired for this study.

Sources of information

Primary mapping and field surveys of the extent, integrity and composition of salt marsh and exposed or regularly submerged muds and sands would need to be linked to the available information on carbon flux from these environments, e.g. Alonso et al (2012)

Values for remediation

Adverse impacts of anthropological origin are likely to be most evident where they affect salt marsh or exposed sands and gravels. There is no clear information on the effect that a changes in the biota could have on the carbon flux and there are no primary measurements for any of these environments in Waterford Harbour. If data were available, this could be used to indicate the extent of carbon sequestration losses and these could be valued in terms of the carbon prices on the ETS or savings on abatement as described in Chapter 3. Without obvious recipients for compensation, remediation should focus on providing an additional area of complementary habitat.

Natural hazard reduction

Introduction

Natural ecosystems around the coast including those within Waterford Harbour have the effect of dissipating wave energy and therefore are of value in mitigating the level of damage that could follow from floods or storm surges. These services are provided by environments that include kelp beds, shellfish reefs, mudflats and saltmarsh. A serious level of degradation, i.e. one is excess of that at which impacts begin to be of ecological significance, would be required before these services are undermined.

Nature of the ecosystem service

Mudflats are a product of sediment deposition due to the dissipation of wave energy. They represent a very dynamic environment and may be rather ineffective in preventing damage from major storm events. Salt marsh, on the other hand, forms where vegetation has an opportunity to get established and is known to have an impact on storm mitigation and erosion. As noted in Chapter 3, wave attenuation is significant at low water depths, i.e. 87%, remaining at a respectable 72% even at greater depths (Moller et al., 2001).

Sources of information

The **Irish Coastal Protection Strategy Study (ICPSS)** for the south coast (OPW, 2011) identifies areas at risk of flooding under 0.1% and 0.5% AEP[56] scenarios. Mapping would be required to identify the extent of salt marsh and periodically exposed mud and sand.

Parts of Waterford City or its environs have been flooded on 13 previous occasions and the location is categorised as 3 (out of 4) on the OPW Flood Risk Assessment Report. Floods in 2004 cost the city council €12,000. Passage East has been flooded on three previous occasions and is categorised as a level 2 risk. Areas identified as being at risk by the ICPSS include much of Waterford City along the stream outflowing from beside the People's Park, narrow sections of the north bank of the Suir east of the city and along the south bank east of Little Island, and a wide corridor between Kilmannock and Fishertown on the River Barrow. Aside from much of the city (which is protected by hard structures), most of these areas are agricultural, albeit with scattered farms and housing.

An exception is the mainland area south of Little Island beside a residential area at Ballinakill Downs (The Pines and Waterside). The vulnerable waterside here is up to 1000m in length. There is no salt marsh as such, but rather salt meadows (wet grassland) that are around 200m in length or approximately 3km^2. Adjacent agricultural land here is protected by a bund. Critical infrastructure here is represented by a water pumping station. Another small area of salt meadow is located to the north of Ballinakill and is 300m in length or around 3km^2 in area. There would be some vulnerability to flooding here, although the area at potential risk is small and receives some protection from bunds. Waterford Harbour was not amongst the seven south coast sites that were identified in the ICPSS as being an erosion risk hazard. The OPW CFRAM study has not yet been published.

Values for remediation

Figures provided in Chapter 3 would suggest that the combined area of salt marsh at Ballinakill would contribute to savings on sea wall construction costs of between €2.3-€4.1 million if raised to today's prices and converted into euro. On this basis the salt marsh could be worth €5,330 per hectare. However, given the location of the marsh in the upper estuary it is unlikely to be causing waves to break and so its true value would only be in terms of containing high tides and flood waters. There would be no need for a sturdy seawall.

Salt marsh further down the estuary could be dissipating wave energy, but the rationale for seawall construction is reduced by the absence of residential properties. Damage to grazing land from freshwater flooding has been estimated by Penning-Rowsell et al (2005) at between €100 and €750 per hectare per event depending on season and intensity. The saline water from a coastal flood event could have a more detrimental impact on agricultural land. However, while there is evidence of past reclamation in Waterford Harbour, there is no evidence of recession of the salt marsh in recent times. On the contrary, the marsh area may have extended in the last century due to dredging in the main channel.

Ecosystem service values depend on the importance of the habitat for natural hazard reduction, the extent of habitat lost or severely damaged, and the number and type of receptors at risk. In the short-term, a lower bound value of the interim losses would be represented by the cost of building artificial bunds or sea walls. In the case of Waterford Harbour, the natural hazard value of the salt marsh is judged to be slight (this situation could change as sea levels rise). The risk to the alternative marsh itself from upstream pollution is slight too and the greater impact would be represented by on-site physical damage, drainage or removal. The disturbance that has occurred to date has had some impact on the biological integrity of the salt marsh, but not on its regulating ecosystem service value.

[56] Annual exceedence probability

Image 5.4 Tidal section of Suir at Ballinakill

5.3.3 Provisioning services

Coastal Fish

Introduction

Estuarine habitats are used by fish, including commercial species, for spawning and shelter, or by juveniles. As discussed in Chapter 2, the relationship between estuaries and fish species has been explored in more detail in North America where some estuaries and bays are both large and very productive. Less attention has been given to the relationship in Europe and it is impossible to identify the relative role of Waterford Harbour compared with other estuaries or shallow sea areas. Shellfish, being more sedentary, are better understood, although the movement of juveniles between locations, the role of different environments and the impact of predators is little known.

Nature of the ecosystem service

Coastal areas, and estuaries in particular, provide a supporting and provisioning ecosystem service as spawning and nursery grounds. They may also provide shelter in severe weather. The south coast is classified as a "low integrity" spawning ground for cod and whiting compared with "high integrity" lengths of the north-east coast and east coast of Northern Ireland. Plaice also use the Waterford coast as a spawning ground (Ellis et al., 2012). However, most of the sampling behind this assessment was undertaken in UK waters for CEFAS. The Atlas of Commercial Fisheries around Ireland (BIM, 2009) reveals the seas off the Waterford coast as receiving the highest fishing effort (hours per nautical square mile for vessels > 15m) for demersal fish of the coast around Ireland at between 84 and 150 hours. This pattern is particularly the case for Irish registered vessels operating out of ports such as Dunmore East.

Table 5.9 lists the landings of various species caught in the three fishing boxes adjacent to the ports of Kilmore Quay, Dunmore East and St. Helen's/Duncannon/ of which Dunmore East is the port closest to the Waterford Harbour. The landings data in the table was supplied by the SFPA statistical unit. In addition, the table identifies estuary-dependent species as listed by Potter, Claridge and Warwick (1986). This source is relatively old. A little more is now understood about estuary dependence, although expert input may be needed to assess the situation for a particular species in Ireland. Some species such as cod are now known to use estuaries at some stage in the life cycle. Others, such as flatfish, use benthic habitats that are typical of estuaries, but which can also be found in shallow seas. More information is available about estuary dependent species in the North American literature, e.g. Able (2005), possibly due to the importance of fishing in some of these large continental estuaries.

Of the European literature, Hinz et al (2006) acknowledge the practical difficulty of linking flatfish species to specific conditions on the sea or estuary bed due in part to competing factors such as prey abundance and the need to avoid predation. For plaice, they find a strong association with sand, but little or no relationship with mud, gravel or hard surfaces. Sole have a clear relationship with sand too, but a variable relationship with mud and no evident relationship with gravel or hard surfaces. Lemon sole has a clear relationship with sand, a weak relationship with gravel or hard surfaces, but no evident relationship with muddy conditions.

Table 5.9 The Sea Fisheries Protection Authority catch for 2012 for the three fishing rectangles of Waterford coast and local catch into Kilmore Quay, Dunmore East and Duncannon.

	Local catch	Total catch	Total catch value
Estuary dependent species			
Flounder	1	1	€625
Lemon sole	467	545	€1,355,960
Black sole	155	230	€2,313,570
Sprat	6836	7232	€1,569,344
Plaice	623	738	€1,423,602
Whiting	2315	3266	€3,759,166
Herring	1868	3489	€1,542,138
Blue mussel *	390	390	€580,000
Clams *		85	€32,300
Razorshell	1	61	€173,911
Scallop *	909	994	€7,371,504
Whelk	183	183	€147,864
Cockle		13	€26,234
Crab *	235	289	€400,843
Lobster European *	4	12	€148,680
Lobster Norway *	1149	2038	€8,575,904
Cod *	1652	2012	€4,297,632
Other (pos dependence)			
Other	5101	6149	approx €13,000,000
Haddock	10507	12357	€16,496,595
Hake	265	335	€613,385
Megrim	1587	1826	€5,300,878

* species that are not necessary estuary dependent, but are likely to use the local estuarine environment.

Sources of information

The south coast is an important area for sea fishing with ports located at Duncannon/St Helens, Dunmore East and Kilmore Quay. Data on fish landings is maintained by the **Sea Fisheries Protection Authority (SFPA)**. Out of total national landings by Irish vessels in 2012 of 198,937 tonnes, landings in St Helens/Duncannon were 2,195 tonnes, Dunmore East 8,518 tonnes, Kilmore Quay 3,722 tonnes, and Waterford 426 tonnes. (BIM, 2009) The total volume catch has been falling gradually for some years due to the impact of quota restrictions, but annual figures for different species and ports tend to vary from year to year.

Fish from the south coast are also landed at other Irish ports and in the UK. White fish are the main quarry. Most of Ireland's cod are caught in this area of the Celtic Sea (up to 300 kg per nautical square mile) along with much of the country's catch of haddock (up to $500kg/nm^2$), hake (up to $230kg/\ nm^2$), ling (up to $104kg/\ nm^2$), megrim (up to $232\ kg/\ nm^2$), monkfish (up to $301kg/\ nm^2$) and plaice (up to $91kg/\ nm^2$) The area is relatively important for ray, black sole and whiting, although catches are rather small in comparison with the above species. Only small amounts of herring and mackerel are caught from this area (BIM, 2009). Data on landings, productivity and fishing effort are available from **SFPA**, the **Marine Institute** and **Bord Iascaigh Mhara (BIM)**. The **IFI** (http://wfdfish.ie/index.php/category/transitional-waters-2008/) also undertakes regular surveys of fish stocks in key estuaries, although the list does not currently include the Suir or Waterford Harbour. The **UK Centre for Environment, Fisheries and Aquaculture Science (CEFAS)**, a division of Defra, are also useful source of information on fish populations.

Values for remediation

It can be presumed that impacts affecting water quality in Waterford Harbour or the condition of certain types of sediment will impact on the species associated with these habitats. A production function approach would therefore be the appropriate method of valuation, but unfortunately there is no reliable quantification of the ecological relationship or function. For instance, there would be great uncertainty as to whether an impact would affect spawning or adult fish. Catches, and landings value, also vary considerably from year to year making it difficult to distinguish impacts. The catch of several species is well below what would have occurred in the past and what might be possible in the future under a policy of an Ecosystem Approach to Management (EAM) and more sustainable local management.

It is therefore not possible to directly demonstrate the value of Waterford Harbour based on these catches. Possibly it could be assumed that catches into Dunmore East are most dependent on the estuary, but there is no certainty of this as too little information is publicly available on where and by whom fish are caught. There is also no information on discards. All that can be said is that there are significant local catches of sprat, whiting and herring, all of which are estuary-dependent species and that Waterford Harbour is an input in a total local catch from the three fishing rectangles of St Helen's/Duncannon, Dunmore East and Kilmore Quay of species with at least some estuary dependence that was worth approximately €69 million in 2012. Of this sum approximately €55 million was landed in the three ports. If an incident were to occur that impacted on the estuary, it would be necessary to examine the importance of the timing and the specific location for particular species depending on their use of estuary within their life cycle as currently understood. As catches and values vary considerably from year to year and are subject to quota, the impact may need to be related to an average catch for perhaps the previous five years.

Shellfish

Nature of the ecosystem service

Good water quality, together with a variety of suitable estuarine habitat, is also important for shellfish. Unlike sea fish, however, adult shellfish are sedentary and may be vulnerable to impacts in particular locations. Shellfish are also filterers. They can accept an amount of organic debris in their environment, but to a varying degree by species. Certain serious pollutants will also accumulate in shellfish populations making them unsafe for human consumption.

Under the Shellfish Directive (2006/113/EC) and Section 6 of the Quality of Shellfish Water Regulations (2006), Member States are required to develop Pollution Reduction Programmes for shellfish waters. These are supplementary to the programmes contained in River Basin Management Plans. The regulations apply to bivalves, but not crustaceans. They set various guideline and mandatory values amongst which are thresholds for suspended solids (not to exceed the content of unaffected waters by 30%), dissolved oxygen (≥60%), hydrocarbons, metals and faecal coliforms. Impact on sediment levels on water quality can arise from various land uses and activities (including diffuse pollution, industrial discharges and waste water plants) as well as activities within the estuary itself (including dredging, bottom fishing, construction, harbour boat movements and works) .

The shellfish area within Waterford Harbour is 9.32km^2 in extent. The harbour has a small aquaculture sector producing Gigas oyster and blue mussel. BIM records reasonably reliable figures on aquaculture production. In the last six years, production has averaged 1,100 tonnes and in 2012 was 1,287 tonnes with a value of €5 million.

Wild shellfish landings are clearly more reliant on good environmental quality, stable substrata and reliable sources. Figures on wild shellfish landings are available from the SFPA, but are less reliable than other landings data in that they are dependent on voluntary reporting by operators. Waterford Harbour is harvested mainly for cockle. The total national landings of cockle were 643 tonnes in 2007, but just 5 tonnes in 2010. The estuary is the third largest centre for cockle after Dundalk and nearby Tramore Bay. The species is found in two inter-tidal locations in the estuary (Woodstown and Passage East). Recruitment, however, has been poor in recent years and the total biomass was estimated at 643 tonnes in 2007. While there were 15 vessels involved in cockle harvesting in 2007 (BIM, 2008), the poor recruitment and the SAC status of the estuary means that no commercial harvesting has been permitted since this year, although small-scale private harvesting is allowed.

Surf clams are also landed into Waterford too. This species is rather demanding of particular substrates of coarse sand, is slow growing and vulnerable to fishing pressure. Total landings into Waterford were 162 tonnes in 2010 and 73 tonnes in 2011 with values of €486,000 and €219,000 respectively. There has been poor recruitment in recent years and the Total Allowable Catch was set at 150 tonnes. However, this was not realised in 2011 due to limited uptake arising from the poor economic return.

Values for remediation

No cockle harvesting in Waterford Harbour has occurred in recent years. Recruitment of both cockle and surf clams has been poor. Eutrophication is a possible cause of the poor recruitment of the former.

The Waterford Harbour Shellfish Area Characterisation Report (Environ, 2010) labels the overall status of the transitional waters of Waterford Harbour as 'moderate' and therefore unsatisfactory due to dissolved inorganic nitrogen. However, the upper section of the estuary is listed as 'good' as is the coastal section and these locations are therefore judged to be satisfactory for shellfish production. Shellfish samples have provided no evidence of metals contamination, but have been found to have high coliform counts that could be due to urban wastewater plants or discharges from domestic systems upstream, but also from the port or ships. These landings fall within the Class B category which allows them to be purified over a time in clean water tanks before human consumption. Costs can be estimated for this period, but the shellfish are in principle non-compliant with the regulations.

A pollution event or impact on shellfish habitat could potentially be quantified in economic terms based on the value of the shellfish for human consumption. The scale of the impacts would depend on whether extra purification is needed or in terms of the value of harvest lost. As with finfish it could be difficult to identify impacts on reproduction and juvenile shellfish. Water quality impacts on aquaculture would be easier to identify than impacts on wild shellfish populations. Impacts would be most severe where catches fall below sustainable yield or economic viability. These could be difficult to separate from poor stock management. Evidently, improvements in local management of the cockle fishery are needed if and when this fishery resumes.

In addition, impacts could also affect other valued elements of the ecosystem. The poor performance of the Cockle population has had an impact on wintering bird population for whom cockle are a major food source. Coastwatch has previously criticised the impact of Cockle dredging in Waterford Harbour.

5.3.4 Cultural services

Introduction

Coastal tourism is not a major draw in the immediate vicinity of Waterford, although there are areas of sand and car parking along the eastern fringe of the harbour below Passage East. A popular pub with beach access is also located in this area as well as some up-market housing which benefits from the sea-views. Tramore is a more significant destination for beach tourism followed by the smaller community of Dunmore East. Both are more frequented by domestic tourists. Tramore is a popular surfing centre and Dunmore East is home to an adventure centre. The spectacular Copper Coast west of Tramore has been designated as a European Geopark. Most wildlife viewing is by local people and only a minority of birdwatchers arrive from outside the county.

Nature of the ecosystem service

Clean water is an important feature demanded of water-related tourism. However, the Mourato et al survey (2003) quoted in Chapter 2 is illuminating in this respect in that it found that more half of respondents never went into the sea and that people rather identified beach cleanliness and facilities as being the most important beach attributes. Litter is a key factor in the former, but evidence of pollution on the beach or the water can be expected to have a similar detrimental impact. Many pollution incidents may not apparent to beach users unless accompanied by evidence of oil, dead fish or birds. Along the coast, interest in water quality would be high amongst those engaged in water-based recreation such as surfing.

Waterford Harbour is an important bird habitat as discussed above and therefore important to birdwatchers too. Most wintering birds occupy the shallower west side of the estuary except at Cheekpoint where the good numbers of wigeon and black-tailed godwit are viewable at a distance. Passage East and Geneva Strand are amongst the other main vantage points. Further along the coast, Tamshunkin Lake is a well-known site of national importance.

Sea angling is a popular activity. The coast of Waterford hosted Ireland's first bass fishing event in 2012. Cheekpoint (just outside Waterford City) and Passage East as the two primary destinations listed on the Sea Angling Ireland website. Bass catches at the these main locations appear to have fallen, although good catches of flatfish, codling, whiting and coalfish are to be found. It is noted that those in Passage East may be vulnerable to the local seal population. Further out at Woodstown Strand bass, dabs and sole can be fished. Around the headland at Dunmore East is good for plaice and dabs while the harbour area is fished for founder, rockling, mullet or congor eel.

Waterford Harbour is also home to a sub-aqua club. The waters off Hook Head are especially popular with other dive sites located off Dunmore East and Tramore.

Sources of information

Information on tourism is available in the numerous research reports that have been prepared by Failtte Ireland (www.failteireland.ie). There were 4.4 million domestic holiday trips in 2010 from which revenue of €1.1 billion was earned. Beautiful scenery was the main reason given for choosing a destination. Twenty-two percent of domestic holidaymakers visited the South East where they spent approximately €21 million. Nationally, 20% engaged in water sports (the nature of which are not specified) and 5% in angling.

The South East attracted 650,000 overseas visitors in 2010 (the most recent year for which data is published) of whom 384,000 were holidaymakers. These visitors accounted for revenue of €175 million, although the figure is well down on recent years due to the recession. There has also been a longer term decline from an estimated 1.1 million visitors in 2000. Waterford was visited by around 204,000 of the 650,000 overseas visitors.

Information on visitor numbers is available for specific or paid attractions, but not at local level below that for major cities. Information on the economic value of water-based tourism and recreation was last prepared by the Marine Institute/ESRI in 2003. Other information could be obtained directly from specialist operators such as sea angling charters and angling shops, adventure centre operators and surf schools/shops.

The welfare value of wildlife in the vicinity of Waterford Harbour is modest and the expenditure related valued is slight. However, the main sites such as Passage West are known to active birdwatchers and will be visited on a regular basis mainly by locally-based individuals. The main birdwatching attractions are to the east in Wexford at the NPWS Wexford Slobs wetland reserve and at Tamcumshin Lake and St. Lady's Lake which are well-known locations for migratory birds and vagrants. A total of 21,413 people visited Wexford Slobs in 2012, although this figure is down on previous years due to the effect of the recession on tourism. For example, in 2005 the reserve received 41,479. A sizeable proportion of these numbers are represented by school parties as the reserve and visitor centre are an important educational resource. Tamcumshin Lake and St Lady' Lake are visited by small groups of dedicated birdwatchers most days.

Values for remediation

The direct pollution risk to water quality from vessels entering Waterford Harbour is not substantial as the port mainly handles containerised traffic, although there are facilities for the receipt of oil shipments. Flushing of tanks is a serious international problem, but vessels would not risk doing this within sight of the coast. The new port is located downstream of Waterford City at Belview. Eutrophication would affect water quality, including turbidity, rducing the attractiveness of seaside locations especially for swimming. Where levels of nutrients are high beach surfaces risk receiving an unsightly covering of algae.

Environmental impacts would also affect birdwatching, sea angling and the general amenity of people visiting or living along the coast. The possible direct impact of cockle fishing and declining shellfish stocks on bird populations was noted above. There are no Irish estimates by which to calculate the welfare impacts of any incident. Utility values for seaside bathing water quality could potentially be estimated using benefit transfer form the UK figures provided by Mourato et al (2003), Hanley, Bell and Alvarez-Farizo (2002a) and Hanley and Kristrom (2002b). The figures provided by Birol and Cox (2007) for wetland habitat in the Severn Estuary could be applied to habitat in Waterford Harbour, although the size and nature of this habitat is very different. All these figures relate to maintaining or improving water quality. No figures could be found for Ireland, the UK or Europe on the welfare impact of avoiding pollution impacts on wildlife.

The Waterford County Council website acknowledges the attraction of angling in County Waterford and provides information here for prospective tourists. At least three sea angling charter vessels operate from Dunmore East. Information on the local scale of operations and charters is available from www.sea-angling-ireland.org and http://www.fishinginireland.info

Image 5.4 Beach at Woodstown in Waterford Harbour

5.4 Summary – River Suir and Waterford harbour

Policy acknowledges that water is a critically important natural resource deserving of protection and significant investment to raise its quality to at least good ecological status. However, demonstrating this is another matter. The Suir application has revealed many of the challenges of applying ecosystem service valuation to aquatic environments. For example, the purest way to reveal the value of regulating ecosystem services is through a production function approach to valuation. However, estimates based on this approach are elusive given the considerable lack of knowledge and uncertainty that attaches to many key ecosystem processes, particularly those performed by invertebrates and micro-organisms. In addition, there is the problem of the dynamic nature of these processes and because their outcomes often vary considerably from one location to another. Much of the natural water treatment performed by the aquatic ecosystem is determined by the maintenance of oxygen levels in the face of nutrient inputs and is largely a consequence of physical factors such as turbulence.

While Chapters 3 and 4 discussed how ecosystem processes continue under varying levels of pollution at least up to a critical threshold, it is clearly those ecosystem services associated with clean water that are most valued by society. A partial estimate of the value of water quality could therefore be supplied by the level of expenditure that policy has deemed necessary for wastewater treatment. This expenditure will vary with both the volume and composition of the effluent and changes in the natural assimilative capacity of the receiving waters. However, this cost-based measure is a inferior and partial indicator of the welfare value of high water quality. It is also subject to the particular nature of the ecosystem itself and its interaction with a water body's physical characteristics. Estimates of the costs have also been elusive given the un-centralised manner in which such information is currently retained by local authorities..

Therefore, the direct elicitation of society's willingness-to-pay for quality water should, in principle, provide an efficient means to circumvent a need to fully understand the contribution of ecosystem process to ecosystem services. Public surveys have habitually been used to provide estimates of the welfare value of water using stated preference methods. However, Chapter 2 demonstrated the difficulty of communicating water quality to the public and the very understandable limits to the public's perceptions of quality and of ecosystem services. The chapter also discussed the weaknesses of the benefit transfer approach, at least where applied across international boundaries. Eventually, the extent to which society is willing to pay for the improvements demanded by the WFD will be tested through public acceptance of the investment needed or, alternatively, through applications for derogation based on grounds of disproportionate cost.

The current chapter sought to estimate, or examine the potential to estimate, some of these values for the River Suir and Waterford Harbour. However, the uses made of the river are modest. In Ireland, there is little use made of irrigation, much abstraction occurs from groundwater and river-based recreation or tourism is largely limited to angling. The river is used for abstraction for public supplies, but even a cost based approach to ecosystem service valuation is restricted by the current absence of water charges.

By comparison, the transitional and coastal sections of the Suir or Waterford Harbour and its vicinity do provide ecosystem services in terms of fish and shellfish production. The former could be of significant value if, indeed, the harbour functions as a nursery environment. Cultural ecosystem services are of modest value within the area of the estuary, but of significant economic importance along the adjacent coast. As with the freshwater environment, the physical characteristics of the estuary appear to play a fundamental part in determining the level of regulating services. The importance of ecosystem services varies considerably both within and between transitional and coastal environments but with there being little reliable evidence of the nature of the processes on which they are based.

6 Report Summary: Ecosystem services, impacts and synergies

6.1 Introduction

The Environmental Liability Directive (2004/35/EC) applies a common liability approach to instances of environmental damage throughout the European Union. It aims to prevent and remedy environmental damage by holding those responsible financially liable for remediation. The objective of ECORISK has been to explore methods whereby the valuation of ecosystem services can be used to supplement established methods of environmental damage assessment based on physical, biological or chemical criteria. In brief, the main findings for the study are listed in the box below:

Overall findings and recommendations

- Remediation should take account of impacts on ecosystem services of value to human beings. It should aim to restore these ecosystem services or compensate for interim losses.
- In some cases the value of these ecosystem services can be quantified in monetary terms.
- Various economic valuation methods are available including cost-based methods, revealed preference and stated preference techniques. As the last of these can be time-consuming, benefit transfer methods are also recommended if the source study has been applied to a similar Irish or UK environment.
- Many ecological functions are not well understood, but often data on distinct environmental changes in outputs (e.g. in fish stocks, bird populations, etc.) is sufficient for environmental valuation.
- When valuing environmental damage, ecologists, the public and specific stakeholders are most likely to value avoidance of dangerous environmental thresholds or tipping points.
- Where monetary quantification is difficult or data unavailable, the scale of these ecosystem services should still be assessed along with the number and identity of recipients. Where ecosystem service losses have occurred in an interim period but cannot be quantified remediation should aim to exceed a no net loss situation.
- Procedures should be put in place to improve the availability of data for local impact assessment, for example data on public and private water abstraction (location, quantity, recipients), data on water and waste water treatment costs, and data on visitor and tourist numbers. Public bodies should be obliged to collect this data and to make it more freely available.
- More primary economic surveys are needed to establish the value that the public places on the quality of freshwater and coastal water bodies and on wildlife and wildlife habitat.
- This research report provides examples of valuation methods but official guidelines would be useful for the valuation of different types of ecosystem services.

On freshwater bodies:

- Rivers and lakes supply a key ecosystem service in the form of waste assimilation and other service benefits in the form of water supply, angling and various types of recreation.

- A water body's capacity to assimilate waste is strongly related to water quality and is best valued through primary stated preference valuation or benefit transfer. Population is a factor, but it is important to define the extent of the spatial catchment in which values are held.
- Angling and recreation values can be measured through a combination of production function methods and revealed preference, i.e. participation, fishing permit sales, boat hire, travel cost and local expenditure.
- Some local authorities have insufficient data - or insufficiently accessible data - on water abstraction, waste water treatment and respective costs.

On estuarine and inshore coastal water bodies

- Estuaries and coastal areas supply key ecosystem service benefits in the form of waste assimilation, fin fish, shellfish, and recreation, including wildlife related recreation. However, ecosystem services valuation can be challenging because many of the relevant ecosystem functions are still poorly understood.

Of the three areas to the ELD applies, ECORISK was asked to focus on impacts to the natural environment, namely damage to water or to protected species and natural habitats. To quality as a significant impact under the ELD this damage must be significant enough to affect the quality status of a water body as defined by the Water Framework Directive (WFD) or sufficient to undermine the achievement or maintenance of "favourable conservation status" of protected species or natural habitats. The report has discussed EU conservation and water policies, but the potential scope of the ELD is wider than for the Birds and Habitats Directives in that it refers to protected species and natural habitats wherever they occur and not just within the confines of Natura 2000 sites.

Where damage has occurred the ELD allows for three types of remediation:

- Primary remediation to restore a damaged resource or impaired service to its baseline condition,
- Complementary remediation in cases where primary remediation would fail to fully restore a site to its baseline conditions using primary remediation alone. This could include improvement to habitat at another site which is geographically linked in terms of species/habitats or human interactions.
- Compensatory remediation where there are interim losses until primary or complementary measures take effect. This includes temporal loss of ecological functions.

Complementary and compensatory remediation requires an "operator" to scale the level of remediation to compensate for the loss of environmental resources. In practice, where restoration is not feasible, complementary remediation has involved habitat restoration, enhancement or creation elsewhere (Mayes, 2008). Measures have generally been decided on a case by case basis whereby environmental protection agencies agree on the level of remediation required.

The definition of complementary and compensatory measures has been informed by Member States' experience of implementing the Habitats Directive, often following legal debate on specific cases. The directive is strict in its interpretation of remediation. Priority is firmly placed on the avoidance of impacts to protected species and natural habitats. Impacts are only conceded for planned projects of imperative reasons of overriding public interest (IROPI) and then only after an 'appropriate assessment' of the implications for the site's conservation objectives. According to Guidance provided by the Commission on Article 6(4) (EC, 2007), compensatory measures are independent and additional to any mitigation required for a project. They are intended to offset certain negative consequences while restoration is undertaken to return a site to the reference biological integrity that justified its designation.

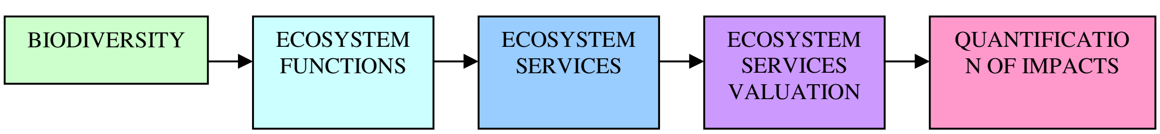
By its nature, a key distinction of the ELD is that it deals with unplanned environmental impacts. A further distinction is that it acknowledges both the importance of environmental functions (including services between species) *and* services to human beings. The ELD notes that where restoration of these services "is not possible, alternative valuation techniques shall be used" to ensure that the environmental resource is remediated to a level equivalent to that which has been lost. This distinction is important in that it permits resource equivalency to be pursued through a hierarchy of resource-to-resource, service-to-service or value-to-value approaches. Where possible, emphasis is placed on remediating the ecological resource by means of resource equivalency analysis (REA) expressed in physical units such as bird species or animals. The service-to-service approach is often described under the heading of habitat equivalency analysis (HEA) and refers to areas of specific habitat including the ecological functions this provides. A value-to-value approach applies the alternative metric of a monetary valuation of the ecosystem service element.

The case for compensatory remediation emerges for interim losses including where physical remediation is prolonged or not possible within a meaningful timeframe. These situations are likely to occur for many habitats that are complex or which have evolved over long time periods. Figure 1.1 helps to demonstrate how interim losses can be relatively significant. Whatever approach is taken, the ELD addresses physical remediation rather than financial compensation, but where interim losses occur, remediation is likely to differ from the replacement of damaged resource and is more likely to be quantified in terms of an equivalent level of functions or services.

6.1.2 Ecosystem services

Ecosystem services are ecological outputs that contribute to human wellbeing. The study has described how the Millennium Ecosystem Assessment (MA, 2005) categorised these flows into supporting, regulating, provisioning and cultural services and how subsequent work by TEEB and for the EEA CICES (Haines-Young and Potschin, 2013b) have focused on the end point benefits from regulating service, provisioning services and cultural services including settings for amenity and non-use values. Ecosystem services are linked to biodiversity through ecological functions. High levels of biodiversity may be associated with high levels of ecosystem processes or functions, but the relationship is not inevitable due in part to a dependency on context, the presence of species redundancy (where a species provides the same functions as another) or, conversely, keystone species which are critical to certain functions or habitats (Naeem et al., 2002). Ecological functions provide for ecosystem services, but not all ecological functions are valuable to ecosystem services. On the other hand, some functions may contribute to more than one ecosystem service.

Figure 6.1. The relationship between biodiversity, ecosystem services and quantification of impacts



6.1.3 Calculating resource equivalency

Calculating resource equivalency in either physical units (REA) or services (HEA) is, in principle, straightforward. It involves the calculation of the resource or service loss (debits) arising from the damage, estimating the expected benefits per unit of remediation (credits), and then dividing the debit by the per-unit credit to determine the total amount of remediation required (REMEDE (Lipton et al., 2008)). This analysis proceeds in five steps, i.e.

1. The initial evaluation
2. Determination of the damage (debit)
3. Determination of the gains from remediation (credits)
4. Scaling the complementary or compensatory remediation
5. Monitoring and reporting

6.2 Valuing ecosystem services

Many ecosystem services are associated with public goods and are unpriced by the market. As a consequence their value may not be appreciated and there is the prospect that they will be mismanaged or damaged. The ELD acknowledges the relationship between human wellbeing and the natural environment. The Birds and Habitats Directives provide for the protection of biodiversity, but the overt acknowledgement of the relationship between human wellbeing and the natural environment by the ELD strengthens the incentive for good environmental management.

The benefit of applying economic values to debits and remediation credits is the use of units that are common to those often used to measure human welfare. This presumes that the ecosystem services can be quantified and then valued in monetary terms. The following non-market valuation methods were described in the report:

- Productivity or production function methods where some market data is available, for example the pricing of an ecosystem service input in terms of its contribution to a marketed output.
- Damage avoided, replacement costs or avertive expenditure. Damage avoided as a measure of the benefit of an ecosystem service, the cost of replacing an ecosystem service with an artificial alternative or of supplementing or strengthening the capacity of the ecosystem to provide a service.
- Revealed preference methods. Based on observation of behaviour associated with the environmental good, e.g. travel costs or property prices.
- Stated preference. Establishing willingness to pay directly by through the use of surveys, e.g. contingent valuation or discrete choice experiments.

Of these methods, the production function approach is useful where an ecosystem provides a provisioning service to a market good, e.g. the contribution of fungal or microbial processes in the soil to a standing crop of timber. Replacement cost or avertive expenditure could relate to investments in flood or storm defences that were previously supplied by the regulating services of wetlands or dunes. Travel costs to a natural destination provide evidence of revealed preference for the cultural service of recreation and can be complemented with data on local expenditure. Of all the methods, only stated preference can capture non-use values and a proportion of total consumer surplus value. The report's database reveals that stated preference has typically been used for cultural services, although with some examples for regulating services too. As ecosystem services have only been formally conceptualised in recent years, most of these studies address elements of Total Economic Value (TEV) rather than ecosystem services. However, within TEV, "use values" relate to provisioning and some cultural services, but "indirect use" and "non-use" values will have been variously relevant to the valuation of supporting, regulating and cultural services.

Stated preference surveys are time consuming to apply, need to undertaken with care and should include documented steps to allow for replication. Contingent valuation approaches are vulnerable to various potential biases while statistical and cognitive limitations restrict the number of attributes that can be included in a choice experiment and therefore the potential scope of the exercise.

Table 6.1 Potential applications of valuation methods and limitations

Method	application	limitations
Production function	Estimating the value of provisioning services inputs to marketed products, .e.g. pollination for agricultural products, soils for forest products, water for manufactured products.	Can be difficult to identify the precise contribution of the ecosystem service.
Replacement cost	To estimate the cost of replacing an ecosystem services such as the hazard protection provided by dunes with an artificial sea wall.	The replacement cost relates to the cost of the replacement rather than the value of the dunes. The replacement good may not provide the same quality of service.
Averting expenditure	Replacing an ecosystem services with an artificial alternative such as hand pollination, use of pesticides, use of fertilisers, water purification, etc.	The replacement cost relates to the cost of the replacement rather than the value of the ecosystem services. The replacement may not provide the same quality of service.
Revealed preference	Estimating the value of a beach in terms of the travel costs that people endure to visit it as an amenity. Capturing the value of an amenity in property prices	Does not capture the full value of the ecosystem services. Need to distinguish the contribution of the good, e.g. the beach from other reasons for visiting. For travel cost time cost may be most important, but unclear how it should be valued
Stated preference	Estimating people's willingness-to-pay to protect or enhance an environment or ecosystem services, or willingness-to-accept compensation in return for its loss.	Captures a greater part of the consumer surplus that other methods, but subject to various biases including due to the typically hypothetical nature of the scenario and the payment vehicle.

Sometimes local data is available for ecosystem services valuation as illustrated by the values estimated within the project's case study of the River Suir and Waterford Harbour. However, in other cases, it can be challenging to obtain such data. The challenges include spatial considerations where an ecosystem services at one location contributes benefits at another (or where damage to an ecosystem service at one location presents external costs for a community at another location). There are also possible temporal considerations such as when the implications of ecosystem damage may not be realised in the short term as with the case of the impact of peatland drainage on carbon emissions and climate change.

Fundamentally, there is often limited ecological evidence of the how environmental conditions contribute to ecological functions and how these in turn contribute to ecosystem services. The report has shown how this lack of knowledge applies especially to some invertebrates and microbes. Ideally, economic analysis requires marginal data to construct a continuous demand or supply function. In practice, this information may not be available and economists will have to settle for more discrete data. This is not necessarily a problem for stated preference surveys as various stated preference studies referenced in the main report demonstrate that the public can be rather insensitive to ecological change where this is subtle or gradual rather than sudden or visual. Therefore, there will be occasions when it is sufficient to present information on ecosystem services *outputs* rather than on underlying ecological processes. However, this may not capture the full range of values for the ecosystem services associated with some environmental benefits such as water quality where the public may only become aware of water quality when it deteriorates to a bad status rather than slipping below the threshold of the good status sought by the WFD. Most values may be attributed to the avoidance of bad status, but good status is required for healthy ecosystem functions.

Furthermore, the stated preference method most readily captures utility values (willingness to pay) at the level of the individual. However, people may value the environment from a biocentric or citizen perspective (Sagoff, 2008). The environment may have cultural, social or ethical meaning to people or be of value at a shared or community level. This can lead to situations in which survey respondents are reluctant to trade-off scenarios of environmental change - even hypothetical scenarios - and especially to agree to trade-offs in monetary terms. A related observation is that people tend to value losses more than gains, i.e. they are more averse to losses (Kahneman and Tversky, 1979; 1984). The ELD addresses environmental losses, but many of the contingent valuation studies listed in the database, including those undertaken for water quality valuation in Ireland and elsewhere in Europe, have estimated willingness to pay for environmental protection or enhancements. Willingness-to-*accept* is more appropriate to an environmental loss, but this type of question is subject to strategic bias and may provide less reliable data.

Despite these challenges or reservations, environmental valuation is preceded by an identification of what ecosystem services are present and who they benefit. This alone can demonstrate the impact of adverse environmental impacts on human wellbeing. Valuation provides a quantification of the impact. The approach therefore ensures that the social value of the environment is not ignored and can be accommodated within remediation measures.

A key requirement is to avoid damage in the vicinity of environmental tipping points. Various researchers have written on the differences between ecological and economic values, for example O'Neill (1993), Spash (2000), Sagoff (1994; Sagoff, 2004) and Admiraal (2013). However, ecology and economics share an interest in identifying the location of critical thresholds at which impacts to the prevailing ecosystem and associated social values would present serious consequences. Identifying and quantifying these thresholds is of much relevance to the ELD. Some of our most valued species and habitats, e.g. high quality rivers, are very sensitive to anthropocentric impacts and pollution impacts can easily transform the valued ecosystem into something rather different and less welcome. The associated ecosystem services may be lost, but they could alternatively be provided by less desirable ecological systems (e.g. algal growth).

Key points – remediation and ecosystem services

- The ELD allows for primary, complementary and compensatory remediation.
- Remediation must restore the lost resource or provide an equivalent nature, degree, area or extent of remediation to the resource lost.
- It is generally assumed that remedial actions should veer towards no net loss or over-compensation.
- Compensatory remediation is especially relevant to interim losses of ecosystem functions and ecosystem services.
- Ecosystem services are the outputs of ecological functions that provide benefits to human beings. Many of these benefits are non-market or public goods.
- There can be considerable uncertainty attached to the relationship between ecosystem functions and ecosystem services. Conventional ecological research has tended to address

impacts on biodiversity rather than of ecosystem functions or especially ecosystem services.

- Uncertainty also attaches to temporal and spatial discrepancies in the supply and demand of ecosystem services.
- Many species (and environments) that are valued by society are sensitive to anthropocentric impacts such as elevated nutrient pollution which can quickly transform the ecosystem into another state, often one that is less desirable from a human perspective.

6.3 Water

To examine the practicality of ecosystem services valuation, ECORISK looked at water, one of the three foci of the ELD, but also of significance to the favourable status of many protected species and natural habitats. The project examined freshwater, estuarine and inshore coastal waters and their ecosystem services. In addition, it undertook a modest case study of the River Suir and Waterford Harbour. Table 6.2 lists the main groups of ecosystem services provided by rivers, lakes, estuaries and inshore coasts and Table 6.3 summarises estimates of ecosystem services value for the Suir.

Table 6.1 Ecosystem services by habitat

Rivers	
Supporting	Wildlife habitat
	Genetic diversity
Regulating	Assimilation (and removal) of waste and nutrients (high)
	Biological control
	Flow and flood moderation
	Sediment capture and deposition
Provisioning	Water supply, fish, reed, etc
Cultural	Contact and non-contact recreation and amenity, especially angling
Lakes	
Supporting	Wildlife habitat
	Genetic diversity
Regulating	Assimilation of waste and nutrients (low)
	Biological control
Provisioning	Water supply, fish, reed, etc.
Cultural	Contact and non-contact recreation amenity, esp angling, swimming, water sports
Freshwater Wetlands	
Supporting	Wildlife habitat including migration/wintering
	Genetic diversity
Regulating	Assimilation of waste and nutrients (high)
	Biological control
	Flow and flood moderation (high)
	Groundwater recharge
	Carbon storage/flux
Provisioning	Water supply, fish, reed, etc.
Cultural	Contact and non-contact recreation and amenity, especially wildlife/ecotourism
Estuaries	
Supporting	Wildlife habitat especially migration/wintering
	Genetic diversity
Regulating	Assimilation of waste and nutrients (high)
	Biological control
	Hazard reduction, e.g. mudflats/salt marsh and coastal flooding

	Carbon storage/flux
Provisioning	Fish nursery habitat & harvest of fish, shellfish, etc.
Cultural	Contact and non-contact recreation and amenity, especially wildlife /ecotourism, sailing.
Coastal	
Supporting	Wildlife habitat especially migration/wintering
	Genetic diversity
Regulating	Assimilation of waste and nutrients (high)
	Biological control
	Hazard reduction, e.g. kelp, shellfish beds, dunes and coastal flooding
	Carbon storage/flux
Provisioning	Fish nursery habitat & harvest of fish, shellfish, seaweed, mearl, etc.
Cultural	Contact and non-contact recreation/amenity, especially beach, swimming, wildlife, sailing, scuba, sea angling.

Appendix 5 of the report provides matrices listing some of the key habitats and species to be found in fresh and coastal waters along with the ecosystem services they support.

Table 6.2 Value Estimates for the River Suir and Waterford Harbour

Activity	Direct revenue	Indirect expenditure	Consumer surplus	Costs averted
Regulating service				
Assimilation of waste	none	n/a	Infer from SP surveys	Additional wastewater treatment
Hazard mitigation	none	n/a	Minimal (as right).	Salt marsh: in theory €2,3-€4,1m. In practice minimal. Alluvial vegetation minimal.
Provisioning service				
Water				Additional water treatment
Eel catch (prior 2008)	€32-€50,000 ^			n/a
Sea fish nursery	< €32m ^ *			n/a
Shellfish	€5.5m ^			Purification in event incident
Cultural services				
Angling	€750,000 **	< €5 million ***		n/a
Birdwatching	No data			n/a
Envir and tourism	No data (likely to be modest)	No data	No data	n/a
Use + existence value	n/a	n/a	SP survey	n/a

^ Waterford Harbour, * Dunmore E catch (low risk).** salmon and trout. *** Based on IFI projections (2013).

For water, ECORISK examined the potential for the use of alternative valuation methods. However, although clean water is presumed to be highly valued, evidence of this value can be difficult to quantify for Irish rivers as levels of recreational use and abstraction are often low. Data on abstraction omits many industrial and small scale private schemes. In principle, the assimilative capacity of the aquatic ecosystem should reduce expenditure on water treatment at source. However, routine treatment must be carried out for reasons for public health. Costs are modest and, while they rise significantly in relation to threats such as *Crytosporidium*, these problems tend to arise from failures in catchment management rather than of the aquatic ecosystem. Other contributors to rivers' capacity to self-clean are only indirectly associated with natural ecosystem services, for example the effect of sunlight on bacterial survival and main-made weirs (albeit in association with microbiological processes). The relatively good water quality of many Irish rivers is due to the regular flushing they receive due to our high rainfall.

Use values are largely restricted to angling and some cruising. Regulating ecosystem services are of direct value to the former and indirect value to the latter. Public access to river banks is rather restricted, but rivers do contribute cultural services through the value the public attaches to wildlife and landscape. Lakes are of more significant value for amenity and tourism. More significant use values are also associated with lakes and coastal waters. However, local data is rather limited. Furthermore, while coastal waters are of provisioning value for fish and shellfish, our understanding of the value of estuaries and bays as fish nursery areas is minimal. We also have only a limited understanding of the value of coastal regulating services for waste assimilation, nutrient cycling and carbon sequestration.

A rather roundabout measure of the value that we attach to the environment is presented by the considerable sums now required for investment in wastewater plant and treatment. Where the assimilative capacity of the receiving environment has been diminished by environmental impacts, there is a need for avertive expenditure on treatment to protect water quality. There is a marginal cost associated with removing more phosphorous or nitrogen or of moving from secondary to tertiary treatment. In principle, the standards of treatment are set by the WRD in relation to the capacity of the receiving water body, but stated preference can provide us with a handle on the value that the public attaches to good water quality status. At present, the public is generally unaware of the cost, but, ultimately, the standards should reflect society's willingness-to-pay for this additional treatment,

Indeed, the costs may be acceptable. A recent Eurobarometer survey (EC, 2012) indicated that 67% of Irish people claim to be conscious of serious water quality problems. Moreover, 63% agree, or tend to agree, that the price of water should reflect the environmental impact of water use (EC, 2012). These findings will be put to the test when water charges are introduced and once derogations are requested with respect to the WFD targets for raising water quality. If the WFD succeeds in introducing more transparency and public scrutiny then the case for stated preference is strengthened as the public will be better informed.

To date, only one primary survey has so far been undertaken in Ireland, namely Stithou et al (2011b), although Norton et al (2012) demonstrate how these values can potentially be transferred to other rivers. More primary studies are needed including of rivers of varying quality and remoteness as well as of the coastal environment. However, the Stithou et al study, along with other European studies, examined the value of an improvement to the aquatic environment rather the (potentially higher) value of *a loss* of environmental quality as would be addressed by the ELD.

Some key findings on ecosystem services and water

- Some data is available on which to base economic estimates of the value of ecosystem services, but only for provisioning and cultural services associated with a handful of direct use activities.
- Ecosystem functions and values are often spatially distinct. For example, the upper stretches of rivers provide spawning habitat for salmon, but the value is realised downstream in terms of angling numbers, capital values, permit revenue and local expenditure.
- Some ecosystem services vary from one location to another and for reasons that may be poorly understood. This makes any value estimate location specific and not very transferrable.
- Estuaries provide nursery habitat for fish, but little information is available on this function and mature fish may be caught in a different location. Even shellfish are mobile at the larvae stage.

- There can be considerable collinearity with other factors. Two locations may provide similar cultural types of ecosystem service, but it may be more developed and valuable at one location than another. This could limit the opportunity for using transfer values.

6.4 Biodiversity offsets and banking

Biodiversity offsetting is a means of providing an equivalent measurable conservation outcome to compensate for adverse biodiversity impacts.[57] Offsets provide for a formalised process of like-for-like complementary remediation and aim to achieve *no net loss* of biodiversity. The report described how biodiversity (or conservation) banking could be used as an extension to offsetting that allows for more flexibility that the bespoke exchange of habitats. Banking involves the purchase of credits in exchange for environmental damage. These credits are then matched against one or more receptor sites or consolidated with others to achieve an equivalent level of remediation. Banking initiatives such as the Williamette Partnership in Oregon www.willamettepartnership.org incorporate ecological functions in the accounting process. Their spreadsheet includes reference to provisioning services and public use, but does not attach economic values to these services. In contract, the Environment Bank being supported under the pilot offsetting scheme[58] in the UK has emphasised the role of ecosystem services. It has been argued that potentially well-managed biodiversity banking could "mainstream biodiversity" (Crowe and ten Kate, 2010) and deliver new habitats additional to those protected by designation (McManus and Duggan, 2011). This does not have to be limited to remediation after damage to protected species or natural habitats, although remediation under the ELD could be a component.

6.5 The Beneficial Use Index and its relevance to estimates of environmental liability

The EPA is developing a Beneficial Use Index (BUI) by which water bodies can be scored to prioritise where resources need to be invested for the WFD. Data is available on abstraction points, designated areas (SACs and SPAs), bathing areas and shellfish waters. This can be assembled into a spatial database.

Potentially, data on economic welfare values could be added to this information. Population data could be combined with transfer values from stated preference surveys to estimate the welfare value of rivers and lakes using distance decay factors as demonstrated by Norton et al (2012), Bateman et al (2006b) and Hanley et al (2003). However, without more primary surveys it is difficult to predict how these values relate to existing water quality, how closely use values relate to the number of places where riverbanks can be accessed, and importantly, the significance of non-use values (values not associated with use).

The value of many regulating ecosystem services could be captured by related provisioning services (water consumption, abstraction by farms) and cultural services. Spatial considerations are a challenge though in that the value of a particular stretch of river may depend on regulating services upstream and the avoidance of impacts both to the river and the wider catchment.

While ecosystem services are extremely relevant to a Beneficial Use Index, in the short term it could be difficult to incorporate much quantified economic data from national data sources. Locally, straightforward market research methods could be used to refine estimates of user expenditure and to collect additional relevant data such as permit sales/purchases, distance travelled, etc. More location specific information on angler or tourist numbers could also be provided through inter-agency cooperation between local authorities and state bodies such as the EPA, Inland Fisheries

[57] See also http://bbop.forest-trends.org
[58] Piloted by the Department for Environment, Food and Rural Affairs,

Ireland (IFI) and Failte Ireland. There is the potential for the Beneficial Use Index, along with the accounting of natural capital required by 2020 under the EU Biodiversity Strategy, to spurn more initiatives for the collection of relevant data.

Over time, the Beneficial Use Index could inform environmental liability estimation and provide the basis for local data collection in the event of an incident. For the purpose of remediation, this economic data can be included within estimates of species or ecological function credits.

6.7 Summary

The ECORISK study has demonstrated the extent to which ecosystem services of social and economic value can be identified to inform remediation. Consideration of ecosystem services should always apply to interim losses addressed by compensatory remediation, but should also be accounted for in the restoration of ecological functions for both primary and complementary remediation. ECORISK has described some of the key ecosystem services performed by freshwater, transitional and inshore marine water bodies through the example of the River Suir and Waterford Harbour. Many of the ecosystem processes and functions behind these services are complex and little understood. However, knowledge of the essential relationship between the ecosystem services and human wellbeing is often sufficient rather than a comprehensive understanding of processes and functions.

ECORISK has identified some regulating services that could be valued through indirect non-market valuation. There is a role for revealed preference where market prices are not available. There is also a role for direct valuation methods involving public survey methods. There will, though, be a need for guidelines to ensure a consistency of approach. For the purposes of the ELD, there is also a need to use these same stated preference methods to estimate the value of losses rather than gains alone.

Ecosystem service valuation would also support a programme of biodiversity offsetting that could deliver conservation gains in return for losses. This could be applied to impacts on protected species and natural habitats addressed by the ELD, but also extend to lesser impacts on more familiar habitats. The quantification of ecosystem service values in monetary terms is feasible in many cases, but will often require on-the-ground data collection in areas where environmental impacts have occurred. Until more detailed environmental use data is collated and more primary valuation studies are undertaken, approaches such as the proposed Beneficial Use Index can provide broad estimates of the significance of impacts.

Appendix 1 - Glossary

AEP	Annual Exceedence Probability (for flooding)
BIM	Bord Iascaigh Mara (State agency for developing seafood industry)
BOD	Biological Oxygen Demand (measure of organic pollution)
CFRAM	Flood Risk Assessment and Management
Defras	Department for Environment, Food and Rural Affairs (UK)
DOC	Dissolved Organic Carbon
EAM	Ecosystem Approach to Management
EQS	Environmental Quality Standard
HEA	Habitat Equivalency Analysis
ICZM	Integrated Coastal Zone Management
IFI	Inland Fisheries Ireland
IPPC	Integrated Pollution Prevention Control
NACE	The European industrial activity classification
Natura 2000	EU wide network of protection areas for nature and biodiversity
NHA	Natural Heritage Area (as listed under national legislation)
NPWS	National Parks and Wildlife Service
Ofwat	UK water regulator (UK)
Q-value	Ecological quality rating used by the EPA
RBD	River Basin District
RBMP	River Basin Management Plan
REA	Resource Equivalency Analysis
Red List	List of threatened species approved by International Union for Conservation of Nature
SAC	Special Area of Conservation (as listed under the Habitats Directive)
SERBD	South East River Basin District
SPA	Special Protection Areas (as listed by the Birds Directive)
TAC	Total Allowable Catch

TSAS	Trophic Status Assessment Scheme
WFD	Water Framework Directive
WTP	Willingness to pay (a measure of economic value as determined by personal utility)

Appendix 2 - Wetlands in Ireland
(after Fossitt, 2000)

* indicates that only some examples of habitat are wetlands

Source: DEHLG (Environ, 2011)

Freshwater wetlands
Dystrophic lakes
Acid oligotrophic lakes
Limestone/marl lakes
Mesotrophic lakes
Eutrophic lakes
Turloughs
Reservoirs
Other artificial lakes and ponds
Eroding/upland rivers
Depositing/lowland rivers
Canals
Drainage ditches
Calcaleous springs
Non-calcareous springs
Reed and large sedge swamps
Tall-herb swamps
Marsh

Marsh
Marsh

Heath and Dense Bracken
Wet heath

Peatlands
Raised bog
Upland blanket bog
Lowland blanket bog
Eroding blanket bog
Rich fen and flush
Poor fen and flush
Transition mire and quaking bog

Woodland and scrub
Wet pedunculate oak-ash woodland*
Riparian woodland
Wet willow-alder-ash woodland
Bog woodland
Scrub*

Exposed rock/disturbed ground
Non-marine caves*

Coastland
Rocky sea cliffs (springs and seepages)*
Sea stacks and islets (springs and seepages)*
Sedimentary sea cliffs (springs and seepages)*
Lagoons and saline lakes
Tidal rivers
Lower salt marsh
Upper salt marsh
Dune slacks
Machair

Marine littoral (intertidal)
Exposed rocky shores
Moderately exposed rocky shores
Sheltered rocky shores
Mixed substrata shores
Sea caves
Sand shores
Muddy sand shores
Mud shores
Mixed sediment shores

Sublittoral (subtidal)
Exposed infralittoral rock
Moderately exposed infralittoral rock
Sheltered infralittoral rock
Exposed circalittoral rock
Moderately exposed circalittoral rock
Sheltered circalittoral rock
Infralittoral gravels and sands
Infralittoral muddy sands
Infralittoral muds
Infralittoral mixed sediments
Circalittoral gravels and sands
Circalittoral muddy sands
Circalittoral muds
Circalittoral mixed sediments

Marine Water Body
Open marine water
Sea inlets and bays
Straits and sounds
Estuaries

Appendix 3 - Habitats Directive Annex 1 habitats

(after Fossitt, 2000)

- 1110, Sandbanks which are slightly covered by sea water all the time
- 1130, Estuaries
- 1140, Mudflats and sandflats not covered by seawater at low tide
- 1150, Coastal lagoons
- 1160, Large shallow inlets and bays
- 1170, Reefs
- 1230, Vegetated sea cliffs of the Atlantic and Baltic coasts
- 1310, *Salicornia* and other annuals colonizing mud and sand
- 1330, Atlantic salt meadows (Glauco-Puccinellietalia maritimae)
- 1410, Mediterranean salt meadows (Juncetalia maritimi)
- 1420, Mediterranean and thermo-Atlantic halophilous scrubs (Sarcocornetea fruticosi)
- 2170, Dunes with *Salix repens ssp. argentea* (*Salix arenariae*)
- 2190, Humid dune slacks
- 21A0, Machairs (* in Ireland)
- 3110, Oligotrophic waters containing very few minerals of sandy plains (Littorelletalia uniflorae)
- 3130, Oligotrophic to mesotrophic standing waters with vegetation of the Littorelletea uniflorae and/or of the Isoëto-Nanojuncetea
- 3140, Hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp.
- 3150, Natural eutrophic lakes with Magnopotamion or Hydrocharition-type vegetation
- 3160, Natural dystrophic lakes and ponds
- 3180, Turloughs
- 3260, Water courses of plain to montane levels with the Ranunculion fluitantis and Callitricho-Batrachion vegetation – UPLAND
- 3270, Rivers with muddy banks with Chenopodion rubri p.p. and Bidention p.p. vegetation
- 4010, Northern Atlantic wet heaths with *Erica tetralix* (flushes only)
- 6410, *Molinia* meadows on calcareous, peaty or clavey-silt-laden soils (Molinion caeruleae)
- 6430, Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels
- 7110, Active raised bogs
- 7120, Degraded raised bogs still capable of natural regeneration
- 7130, Blanket bog (*active only) (flushes only)
- 7140, Transition mires and quaking bogs
- 7150, Depressions on peat substrates of the Rhynchosporion (wet heath only)
- 7210, Calcareous fens with *Cladium mariscus* and species of the Caricion davallianae
- 7220, Petrifying springs with tufa formation (Cratoneurion)
- 7230, Alkaline fens
- 8310, Caves not open to the public
- 8330, Submerged or partly submerged sea caves
- 91D0, Bog woodland
- 91E0, Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (Alno-Padion, Alnion incanae, Salicion albae)

Appendix 4 - Habitats Directive Annex II species dependent on wetlands
(after Fossitt, 2000)

Species Code	Species name	Common name
1013	*Vertigo geyeri*	Whorl snail
1014	*Vertigo angustior*	Whorl snail
1016	*Vertigo moulinsiana*	Whorl snail
1024	*Geomalacus maculosus*	Kerry slug
1029	*Margaritifera margaritifera*	Freshwater pearl mussel
1065	*Euphydryas aurinia*	Marsh fritillary
1092	*Austropotamobius pallipes*	White-clawed crayfish
1095	*Petromyzon marinus*	Sea lamprey
1096	*Lampetra planeri*	Brook lamprey
1099	*Lampetra fluviatilis*	River lamprey
1102	*Alosa alosa*	Allis shad
1103	*Alosa fallax*	Twaite shad
1106	*Salmo salar*	Atlantic salmon
1355	*Lutra lutra*	Otter
1393	*Drepanocladus vernicosus*	Shining sickle moss
1395	*Petalophyllum ralfsii*	Petalwort
1421	*Trichomanes speciosum*	Killarney fern
1528	*Saxifraga hirculus*	Yellow marsh saxifrage
1833	*Najas flexilis*	Slender naiad
1990	*Margaritifera durrovensis*	Nore pearl mussel

Appendix 5 Matrices of ecosystem services

Matrix 1 Freshwater ecosystem services see page 139

Matrix 2 Inshore and coastal ecosystem services see page 141

Key or Characteristic	STATUS									ECOSYSTEM SERVICES																					
Habitat or Speciess										Supporting services							Regulating services					Provisioning				Cultural					
pink = listed in Habitat (Annex II) o	Rarity (rare, scarce, common)	Condition (bad, poor, good)	Vulnerability to chemical pollutio	Vulnerability to organic pollution	Vulnerability to sediment	Vulnerability to invasive species	Vulnerable to changes in hydrolo	Vulnerability to habitat loss	Vulnerability to disturbanace	Primary/secondary production	Food web dynamics	Nutrient cycling	Pollination	Fish nursery / migration	Bird/animal habitat / diet	Bird migration	Climate regulation	Natural hazard regulation	Water quality	Sediment quality/control	Biological control	Fish	Drinking water	Other water supply	Agricultural use	Angling	Boating/Sailing	Passive	Birdwatching/nature	Direct water contact	Aesthetic
Dystrophic lakes		B												xx																	
Oligotrophic lakes	C	B		x		x			x					xx								x	x	x	x	x			x	x	
Mesotrophic lakes		B		x	x	x			x					x		x						x	x	x	x	x		x	x	x	
Naturally eutrophic lakes	S	B		x		x			x							x						x	x		x	xx		x	xx		
Turloughs (gd water dependent)	S	P		x			x	x								x		x							x			x	xx		
Upland rivers	C	G			xx				x					x	x				xx			x	x	x	x	xx		x	x	x	
Lowland rivers	C	P		x	x	x			x					x		x						x	x	x	x	xx		x	xx	x	
Alluvial woodland (terrestrial)	S	B					x							x			x														
Bog woodland (terrestrial)	C	P					x																								
Wet heaths (terrestrial)							x																						x		
Raised bog (terrestrial)	S	B					xxx	xx	xx								xx									x		x			
Blanket bog (terrestrial)	C	P					xx	x	x							x	xx	x	x						x			x	x		
Transitional mire/fen (terrestrial)	S	B		x			x	x	x							x	xx	x	x			x		x				x	x		
Calcarious fen (gd water dependent)		B		x			x		x			xx		x			xx														
Alkaline fens (gd water dependent)		B		x			x		x					x			xx														
Reed swamp												xx			x	x		x				x		x					x		
Tall herb swamp						x		x	x			xx	xx		x	x	x	x						x					x		
Freshwater marsh	S								x			xx	x	x	xx	x		x	x					x		x		x	xx		
Macrophytes																															
bladderworts																															
pondweeds																															
bogbean																															

Key or Characteristic	STATUS									ECOSYSTEM SERVICES																						
Habitat or Speciess										Supporting services							Regulating services					Provisioning				Cultural						
	Rarity (Rare, Scarce, Common)	Condition (bad, poor, good)	Vulnerability to chemical pollutio	Vulnerability to organic pollution	Vulnerability to sediment	Vulnerability to invasive species	Vulnerable to changes in hydrolo	Vulnerability to habitat loss	Vulnerability to disturbanace	Primary/secondary production	Food web dynamics	Nutrient cycling	Pollination	Fish nursery / migration	Bird/animal habitat / diet	Bird migration	Climate regulation	Natural hazard regulation	Water quality	Sediment quality/control	Biological control	Fish	Drinking water	Other water supply	Agricultural use	Angling	Boating/Sailing	Passive	Birdwatching/nature	Direct water contact	Aesthetic	Culture
mosses, liverworts and lichens	Rivers = 1 RE,		x	xx					x								xx	x	x													
common reed	C																		x													
duckweed																			x													
sedges	C																		x													
marsh fratillary																																
yellow march saxifrage																																
Salmonids	VU C	P	xx	xx	xx	x	x							x	x							x				xx						x
Brown trout			xx	xx	x	x	x															x				xx						
Shad	VU	P	xx	xx	x	x	x																									
Char	VU R	P	xx	xxx	x	x	xx																									
Pollan	VU	P	xx	xx	x	x																										
Angling coarse fish				x		x																				xx						
Brook/River Lamprey	NT	G	xx	xx	x	x			xx																							
Eel	CR R	B	x	xx		x	x				x			x								x										
Common frog	C		x	x	x				x																							
Smooth newt	S		x	x	x				x																							
Invertebrate community																																
white-clawed crayfish		P	x		x	x									x													x				
nore pearl mussel	R	B																														
pearl mussel	R	B		xxx	xx	x			xx																							
whorl snails	Most sp	P		x			x	xx	x																							
caddisfly																																
mayfly	6 threat	B/P	x								xx				xx											xx			x			
stonefly nymphs																																
dragonfly/damselflies	Total 2 EN, 2 VU, M			x							x				x											x		x				
Zooplankton										xx	xx	xx					xx															
Phytoplankton										xxx	xx	xx					xx															
Microbial population											xxx	xxx					xx		xxx	xxx	xx											
Bacteria																																
Otter	C	P	x	x					x																				xx			x
Kingfisher	S		x	x	x				x																				xx			x
Lapwing (breeding)		P					xx	xx	x																				xx			x
Redshank (breeding)		P					xx	xx																					xx			
Sawtooth ducks/grebes	R-C		x	x				x	x																				xx			
Ducks/swans	R-C																												xx			
Wintering geese	S-c							x	xx																				xx			x

Habitat/Species type C. Bullock (ECORISK) adapted from Fletcher et al (2012) and UK ValuingNature Network	STATUS Rarity (rare, scarce, common)	Condition (bad, poor, good)	Vulnerability to chemical pollut	Vulnerability to organic pollutic	Vulnerability to sediment	Vulnerability to invasive specie	Vulnerability to disturbance	ECOSYSTEM SERVICES Supporting services Primary production	secondary production	Food web dynamics	Nutrient cycling	Fish nursery/migration	Bird/animal habitat/diet	Regulating services Climate regulation	Natural hazard regulation	Water quality	Sediment quality/control	Chemical absorption/indicators	Biological control	Provisioning Fish	Fertiliser	pharmaceuticals	Blue biotechnology	Cultural Angling	Boating/Sailing	Passive	Birdwatching/nature	Direct water contact	Aesthetic	Culture
TRANSITIONAL HABITATS																														
Sandbanks (covered by water)	C		x	x	x							x	x		x					x							x			
Mudflats (not covered)	C		x	x	x			xx	xx	xx	xx	x	xx		x					x							xx			
Estuaries	C	P	xx	xx	x		x	xx	xx	xx	xx	x	x		x	xx	x			xx				x	x	x	xx	x		xx
Lagoons	S			x			x	xx		xx	x		x												x	x	xx			xx
Shallow inlets	S			x			x			x		x	x							xx				xx	xx	xx	x	xx		xx
Saltmarsh								xx			x	x	x	x	xx		x	x		x							x			
COASTAL / INSHORE MARINE																														
Intertidal rock								x	x						xx								x							
Intertidal reefs	S						x	x	x			x			xx					xx			x	x			x	xx		
Vegetated sea cliffs	C	G											xx													xx	x			xx
Dunes	C	B			xx		xx						x		xx		x									xx	x	xx		xx
Machair	s	P			x								x				xx						x			x	x	x		xx
SPECIFIC HABITATS																														
Mussel beds									x	x	x		x		x	xx	x	x		xx			x				x			x
Oyster beds									x	x	x	x			x	xx	x	x		xx										x
Maerl								x	x	x	x	x	x						x	x										
Seagrass								xx	xx	xx	xx	x		x	x	x	xx		x	xx	x			x						
Seaweeds								xx	xx	xx	xx	x		x	x	xx	x	xx	x	xx	xx	x	xx	x						x
SPECIES																														
Important functional groups:																														
Bioturbators									xx	xx	xx		xx			xx	xx													
Grazers (e.g. Limpets)										xx	x					xx			x											
Charismatic or protected mammals and birds																														
Salmon																				x				xx						x
Dolphins & purpoises										x																	xx			x
Otter																											xx			x
Seals (grey, common)										x																	xx			x
Common seal										x																	xx			x
Migratory waders	S-c																				x						xx			
Wintering duck/grebes																											xx			
Wintering geese																											xx			x

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[i] The shortage of information on the character of ecosystem functions that support ecosystem services derives from the manner in which the study of natural science has evolved and is understood. Ecology is itself an offspring of the fundamental sciences of biology and botany. Its distinctive feature is its interest in the complex interactions that exist between living things. However, while ecology has made us aware of the complex relationships that exist between living things, our own relationship with it has been neglected. Although there is an obvious and growing consciousness of the dependence of human beings on the environment, the specifics of this relationship, as it is realised throughout the hierarchy of living things and their interaction with a dynamic, changing physical environment, remain poorly understood. Out of practically, ecological research typically focuses on the relationship between small groups of individual species at one point in time. To understand the functions behind ecosystem services, a wider picture must be pieced together from various studies even assuming that related studies exist. Very often they do not.

The Directive also provides for the future repeal of a number of existing Directives and decisions:

- Groundwater Directive (80/68/EEC)
- Surface Water Abstraction Directive (75/440/EEC)
- Shellfish Directive (79/923/EEC)
- Fish Directive (78/659/EEC)
- Discharge of dangerous substances (Directive 76/464/EEC
- Drinking Water Measurement Directive 79/869/EEC
- Council Decision 77/795/EEC establishing a common procedure for the exchange of information on the quality of surface freshwater

It also sets standards in relation to the following Directives:

- [iii] Urban Waste-water Treatment Directive (91/271/EEC);
- Nitrates Directive (91/676/EEC);
- Integrated Pollution Prevention Control Directive (96/61/EC).
- Mercury Discharges Directive (82/176/EEC);
- Cadmium Discharges Directive (83/513/EEC);
- Mercury Directive (84/156/EEC);
- Hexachlorocyclohexane Discharges Directive (84/491/EEC);
- Dangerous Substance Discharges Directive (86/280/EEC)