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Environmental Protection Agency
An Ghníomhaireacht um Chaomhnú Comhshaoil

Benefit Transfer for Irish Water

STRIVE

Environmental Protection Agency Programme

2007-2013

SSTI Strategy for Science, Technology and Innovation

Comhshaol, Pobal agus Rialtas Áitiúil
Environment, Community and Local Government

Environmental Protection Agency

The Environmental Protection Agency (EPA) is a statutory body responsible for protecting the environment in Ireland. We regulate and police activities that might otherwise cause pollution. We ensure there is solid information on environmental trends so that necessary actions are taken. Our priorities are protecting the Irish environment and ensuring that development is sustainable.

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EPA STRIVE Programme 2007–2013

Benefit Transfer for Irish Water

Using Benefit Transfer Techniques to Estimate the Value of achieving 'Good Ecological' Status in Irish Water Bodies

(2010-SD-DS-1)

STRIVE Report

Prepared for the Environmental Protection Agency

by

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Executive Summary

The aim of the Water Framework Directive (WFD) (2000/60/EC) (WFD) is 'to establish a framework for the protection of inland surface waters, transitional waters, coastal waters and ground waters'. The Directive calls for integrated catchment management plans to be prepared for all river basins in order to achieve 'good ecological status' (GES) in all EU waters by 2015. As such, the Directive aims at a minimum for a 'good' and 'non-deteriorating status' for surface, underground and coastal waters and sets common approaches and goals for water management in EU member states. An important element of the Directive is that it calls for a consideration of the economic costs and benefits of improvements to ecological status in catchment management plans, along with the introduction of full social cost pricing for water use. Hence, benefits play an important role in the assessment of the proportionality of costs in the implementation of the WFD. This report explores the use of benefit transfer (BT) techniques in placing a value on achieving GES, as specified in the WFD, across water bodies in Ireland. Given that no major valuation exercises on water quality in Ireland have been conducted, BT will be crucial for estimating these benefit/cost ratios, and thus identifying cases of disproportionate costs for which derogations can be sought. This project aims to identify the most appropriate BT methodology to use in the Irish situation and apply it to a number of catchment policy sites.

Benefit transfer involves taking valuation estimates from primary valuation studies and applying them to an alternative site where one is valuing the same environmental good or service as in the primary study. When analysed carefully, information from past studies published in the literature can form a meaningful basis for water management policy valuation through transferring values from a study site to a policy site. This study used a number of BT approaches to estimate the value of achieving GES under the WFD. It first carried out a simple unit BT (where the unadjusted willingness to pay [WTP] estimate from one or more study sites was used to apply their average value to the policy site) to estimate the value of achieving GES based on the change in water status across 151 water management units (WMUs) in Ireland. Next, an adjusted (for distance decay) BT unit transfer approach was used to measure the value of achieving GES for the Boyne catchment. A primary contingent valuation (CV) method estimate of the value of achieving GES in the Boyne was used to examine the transfer error arising from this BT. Finally, a BT function transfer approach was used to look at the value of a number of catchments achieving GES where the value function – with associated attribute coefficient values – used in the BT process was taken from a primary valuation study, and input information for the water bodies examined (in terms of the catchments' environmental attribute levels) was provided by experts in each river basin district.

The unadjusted unit transfer values provided total catchment benefit values ranging from €388 for the Sheen WMU to €2,800,352 for the Tolka. The overall value for achieving 'at least good ecological status' in the Boyne catchment using the adjusted BT approach and accommodating distance decay effects was estimated to be €13,600,000 with a 95% confidence interval between €7,100,000 and €20,200,000. In terms of the function transfer approach, in the original study the authors estimate a compensating surplus value associated with the Boyne catchment achieving GES (per household/year) of €32.7 with a 95% confidence interval between -€55.26 and €114.68. Based on the expert opinion for the river attribute levels for the Boyne catchment a BT mean value of €51.73 was estimated. A comparison of the results of these BT approaches to the results in the primary studies in the Boyne gave transfer errors of 29% and 58% for the distance decay unit transfer and the function transfer approaches respectively.

Overall, results show that the uncertainty in value transfers can be quite large. It can be argued however that the transfer errors calculated for the BT estimates for the Boyne catchment are not overly large when one compares them to estimates elsewhere in the literature. It could be argued that any BT estimates produced in order to quantify the benefit value of a water body achieving GES should only be used to compare the relative values across water bodies or where the demand for accuracy is relatively low. The use of BT estimates for making decisions in relation to disproportional costs at single sites is not recommended. In the limited cases where policy-makers feel that the costs of achieving GES may be higher than the aggregate benefits from such a policy intervention, then a primary survey should if at all possible be carried out to determine those aggregate benefit values as accurately as feasible.

1 Introduction

The aim of the Water Framework Directive (WFD) (2000/60/EC) (WFD) is 'to establish a framework for the protection of inland surface waters, transitional waters, coastal waters and ground waters'. The Directive calls for integrated catchment management plans to be prepared for all river basins in order to achieve 'good ecological status' (GES) in all EU waters by 2015. This concept is a broader measure of water quality than the chemical and biological measures that were previously dominant. As such, the Directive aims at a minimum for a 'good' and 'non-deteriorating status' for surface, underground and coastal waters, and sets common approaches and goals for water management in EU member states. An important element of the Directive is that it calls for a consideration of the economic costs and benefits of improvements to ecological status in catchment management plans, along with the introduction of full social cost pricing for water use. Hence, benefits play an important role in the assessment of the proportionality of costs in the implementation of the WFD.

The WFD was adopted in October of 2000 and is considered an ecosystem-based approach to the management of water resources within Europe. It has been described as 'possibly the most ambitious and far-reaching piece of environmental legislation to have originated from the European Union' (Moran, 2008). It sets out a framework for achieving its aims rather than imposing rigid and overly prescriptive regulations – thus, the methods and legal instruments used may differ between member states. The WFD seeks to establish sustainable use of water by member states while concurrently protecting aquatic ecosystems and their dependent habitats. Previous efforts at the supranational level had focused on specific water-quality problems mostly connected with human health or direct uses of water such as for drinking (Drinking Water Directive [DWD] [98/83/EC]), bathing (Bathing Water Directive [76/160/EEC]) and shellfish (Shellfish Waters Directive [2006/113/EC]). Previous directives dealing with the improvement of water quality were in relation to the discharge of pollutants into water bodies, such as the Urban Waste Water Treatment Directive (UWWTD) (91/271/EEC) and the Integrated Pollution Prevention and Control Directive (IPPC) (96/61/EC).

The number and breath of previous EU directives related to water protection resulted in a patchwork approach to water management, which the WFD aims to replace with a more coordinated method. The WFD mirrors other recent EU legislation aimed at protecting the environment through integrated ecosystem approaches, such as the Marine Strategy Framework Directive (MSFD 2008/56/EC) and the proposal for a Soil Framework Directive (COM(2006) 232)). Adopting an ecosystem approach changes the previous emphasis from a narrow focus on a few physio-chemical water quality indicators to a broader basket of achievements, encompassing not only certain water quality indicators but also obtaining water bodies with proper ecosystem functioning and GES.

At a minimum, the Directive aims to achieve at least at 'good' and 'non-deteriorating status' and at the same time standardise the methods which EU member state countries use to achieve these objectives. With this in mind, member states are responsible for the preparation and implementation of management plans at the natural hydrologic (river basin) level (known as river basin district [RBD]) instead of other administrative or political boundary levels. If an RBD crosses national boundaries then both member states are required to work together to produce a joint RBD management plan. Table 1.1 outlines the timeline for the implementation of these plans and for the implementation of the WFD as a whole. The plan set out for each RBD follows a DPSIR (Drivers, Pressures, State, Impact, Response) Framework (Fig. 1.1) which is often adopted for implementing environmental policy (Borja et al., 2006). Each RBD management plan identifies the environmental pressures on the water bodies (pollutants or abstractions) and the socio-economic drivers that cause these pressures (population, agriculture and industry).

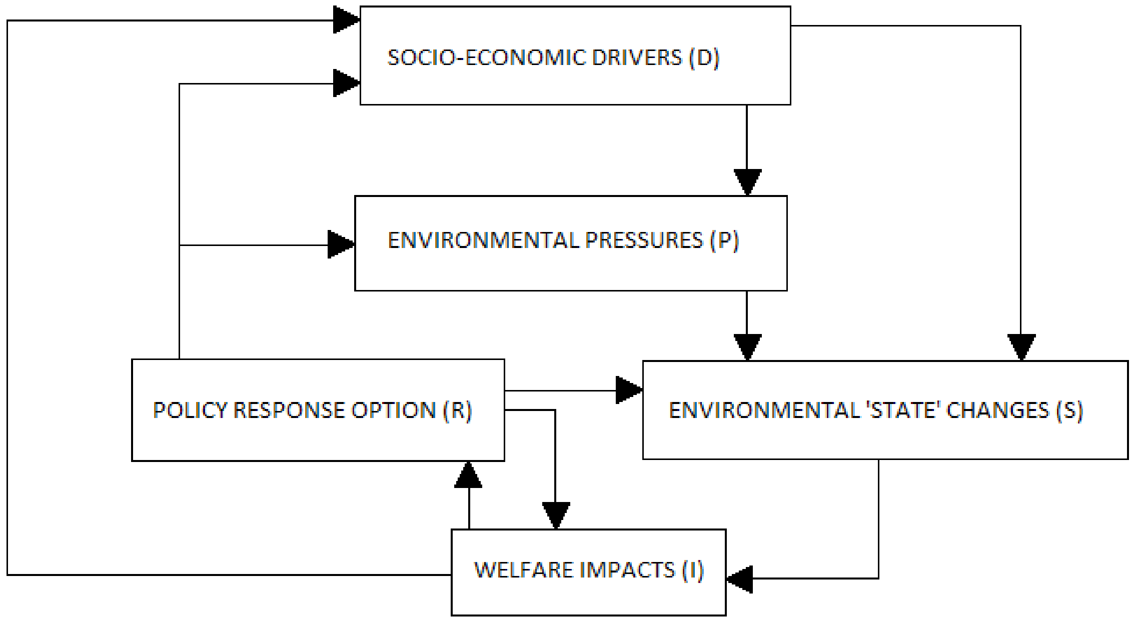

Figure 1.1. Drivers, pressures, state, impact, response (DPSIR) framework.

Source: adapted from Turner et al. (1998)

The impacts of these pressures and the sub-catchments (known as water management units [WMUs]) which are affected are identified. In turn, the RBD plan proposes to either change the state of each WMU (i.e. from poor to good) or maintain high status, and identifies what response policy-makers should take. Initially, these responses are to apply and enforce existing environmental protection legislation and policy (known as 'basic' regulatory measures) more stringently. Where it is felt that the 'basic' regulatory measures are failing or will fail to achieve the stated aims of the RBD management plan in time, additional measures (known as 'supplementary' measures) may be imposed. Both authorities at a local (RBD) and national (member state) levels will be responsible for the development and implementation of sufficient measures that will lead to good surface water status (GSWS).

The WFD also requires that charges for water services should adopt the principle of full-cost recovery and, in accordance with the 'polluter pays principle', provide incentives for water-use efficiency. At the same time, common methods to estimate these costs are yet to be determined and it is expected to be quite challenging in a number of member states where water in the domestic and agricultural sectors is subsidised (Spain, Greece, Portugal) or where water pricing is almost completely absent (Ireland). In the latter case, the political cost of asking households to pay for environmental improvements when sources of diffuse pollution are not fully checked is expected to be high. Furthermore, pricing mechanisms imply 'benefit pricing' based on willingness to pay (WTP) (Morris 2004).

Ireland is somewhat behind in terms of measuring the economic value of achieving GES under the WFD across catchments. This report aims to fill this gap in the research by exploring the use of benefit transfer (BT) in placing a value of achieving this main objective (i.e. achieving GES) of the WFD across water bodies in Ireland.

It is recognised internationally that water resources are necessary inputs to production in economic sectors such as agriculture (arable and non-arable land, aquaculture, commercial fishing, and forestry), industry (power generation) and tourism, as well as to household consumption (Birol et al., 2006). However, water resources are also valued for more than purely production purposes. The value of clean water resources can arise from the many non-market benefits that are provided, including recreation use, and maintenance of biodiversity (Brouwer et al., 2009).

Table 1.1. Water Framework Directive timetable.

Year	Issue	Reference
2000	Directive entered into force	Art. 25
2003	Transposition in national legislation	Art. 23
2004	Identification of River Basin Districts and Authorities	Art. 3
	Characterisation of river basin: pressures, impacts and economic analysis	Art. 5
2006	Establishment of monitoring network	Art. 8
	Start public consultation (at the latest)	Art. 14
2008	Present draft river basin management plan	Art. 13
2009	Finalise river basin management plan including programme of measures	Arts 13 & 11
2010	Introduce pricing policies	Art. 9
2012	Make operational programmes of measures	Art. 11
2015	Meet environmental objectives	Art. 4
2021	First management cycle ends	Arts 4 & 13
2027	Second management cycle ends, final deadline for meeting objectives	Arts 4 & 13

Source: http://ec.europa.eu/environment/water/waterframework/info/timetable_en.htm

It should be noted that some European water bodies have become so polluted that it may prove too expensive to restore their quality to GES by 2015 or, in certain cases, even beyond the deadlines set for the second and third WFD planning cycles (2016–2021 and 2022–2027). The WFD has allowed for this possibility through Art. 4(4), which allows member states to extend the deadline for achieving GES by up to 12 years beyond 2015 if it is technically infeasible, disproportionately expensive or if natural conditions do not allow improvement within that time scale. Extensions beyond 2027 may be allowed where natural conditions do not permit the achievement of GSWS within the previous three planning cycles.

Proving that achieving GES is disproportionately expensive requires comparing the costs of putting in place a water management plan to achieve GES with the benefits that might come about as a result of achieving GES. Given that no major valuation exercises on water quality in Ireland have been conducted, BT will be crucial for estimating these benefit/cost ratios, and thus identifying cases of disproportionate costs for which derogations can be sought. This project aims to identify the most appropriate BT methodology to use in the Irish situation and apply it to a number of catchment policy sites. By so doing, the research can add significantly to the national knowledge base in relation to water quality valuation and be in line with other international research such as Hanley et al. (2006a) and Morrison and Bennett (2004).

Given this brief introduction to the WFD (and some key terms from the direction shown in Box 1.1), the research objectives of this project sought to address a number of methodological issues and gaps in the water quality valuation literature. In particular, the project uses catchment area analysis, Census of Population 2006 statistics and geographic information system (GIS) techniques to define and model the geographic extent of the market for water quality valuation at individual catchment sites. The research also contributes to the literature by conducting the first BT exercise for water quality valuation in Ireland in order to value the benefit of achieving GES under the WFD in a number of catchments. The estimated BT benefit values are also validated against the results of a choice experiment (CE) that was conducted in late 2009 and early 2010 in the Boyne catchment to value the benefit of achieving GES under the WFD in this catchment. This also tested whether the benefit estimates for the policy site used in the transfer are statistically the same as the actual values that were collected for the original Boyne study (Stithou et al., 2011).

Box 1.1. Important Definitions in Water Framework Directive

Ecological status is an expression of the quality of the structure and functioning of aquatic ecosystems (WFD, 2000) and is measured using a cross-section of biological parameters, supporting physio-chemical parameters, and supporting hydrology and morphology conditions. Biological parameters include the composition and abundance of aquatic flora (diatoms, phytoplankton, macrophytes) and aquatic fauna (benthic invertebrates, fish). Physio-chemical parameters include oxygen, nutrients, water clarity, temperature, acid status and salinity. Hydrology and morphology conditions include flow, depth, water level and bankside conditions.

Chemical status is a measure of the concentrations of priority substances and priority hazardous substances at EU level, e.g. pesticides, hydrocarbons where 'good chemical status' is achieved when the concentrations of pollutants do not exceed the environmental quality standards.

Good surface water status refers to the status achieved by a surface water body when both its ecological status and its chemical status are at least 'good'. One of the main aims of the WFD is to develop a combined approach of assessing emission limit values and quality standards to manage water quality and quantity (Carter and Howe, 2006).

In the case of 'artificial/modified' waters serving economic activities (i.e. canals or weirs) the aim of the WFD is to achieve **good ecological potential** rather than *good ecological status*, in which only *chemical status* is needed to achieve good status. Conversely, for protected areas (e.g. SACs or SPAs), designated for the protection of water-dependent habitats or species- and nutrient-sensitive areas, more stringent requirements may be applied in order to achieve the WFD objectives.

The rest of the report is structured as follows. Section 2 examines how the WFD aims to create an efficient allocation of water resources. Section 3 then reviews the implementation of the economic assessments required under the WFD in Ireland. Section 4 looks at alternative valuation methodologies that researchers can use to value the benefit of achieving GES, while Section 5 reviews previous valuation studies that have examined the issue of modelling and valuing water-quality improvements. Section 6 outlines the alternative BT methodologies that could potentially be used to place a value on the benefit of a water body achieving GES under the WFD. Section 7 then presents three alternative BT applications to estimate the value of a number of Irish water bodies achieving GES and tests the validity of the results. Finally, Section 7 concludes with some recommendations for future research. While the alternative BT methodologies are discussed in Sections 6 and 7, a separate guidance document is provided in the Appendix. This sets out in more depth the steps available to a practitioner undertaking a BT which estimates the non-market benefits resulting from a surface water body achieving GES under the WFD.

2 Water Framework Directive Economics

From an economic perspective, water resources are not efficiently allocated and may be overexploited due to the existence of market and government failures at different levels (local, national and international). This phenomenon primarily occurs because of the 'public good' nature of water resources and secondly because of the complexity that characterises their value (including use and non-use values) such that they are not traded in markets as private goods. Hence, when people use water resources they don't pay their scarcity rent[1] (both in terms of quantity and quality) but instead pay for the private extraction cost (Birol et al., 2006). Due to these unrecognised scarcity rents, private costs and benefits diverge from social costs and benefits leading to social welfare losses that are expressed as inefficiently high extraction levels or pollution over time and space (Pearce and Turner, 1990 and Koundouri, 2000). The nature of the economic development and growth path that is followed is such that the value of environmental goods such as water resources has often been overlooked in development decisions.

'Economic efficiency' occurs at the point where net social benefits (i.e. benefits minus costs) of an economic activity are maximised, or equivalently, when the marginal benefits are equal to marginal costs, in order to implement the most efficient social and economic policies that prevent the excessive degradation and depletion of environmental resources. As a result, it is necessary to establish their full value and to incorporate this into private and public decision-making processes (Birol et al., 2006). The WFD was formulated with this objective in mind. In particular, the EU WFD is one policy initiative that aims to ensure the sustainable management and conservation of this valuable resource, along with other international efforts such as the 1971 Ramsar Convention on Wetlands of International Importance (Ramsar, 1996). In order to achieve this, the WFD promotes the concept of water as an economic commodity, while maintaining its focus on water's broader and often intangible value. It also recognises the importance of economics by integrating it in different ways in order to guide decisions that are in line with the Directive's objectives. Particularly, economic principles are to be applied in four main respects within a river basin (Morris, 2004):

1 The estimation of the demand for water and the valuation of water in its alternative uses (Art. 5);
2 The identification and recovery of costs, environmental and resource, associated with water services, having regard for the polluter pays principle and the efficient use of water (Art. 9);
3 The use of economic appraisal methods to guide water-management decisions (Art. 11);
4 The use of economic instruments to achieve the objectives of the WFD, including the use of incentive pricing and market mechanisms (Art. 11).

Article 9 stresses the need for users (i.e. industries, farmers, and households) to be charged a price that reflects the full costs of the water services they receive. As a result, the price attached to water services is expected to depend not only on the operational, maintenance and costs invested in infrastructure but should also cover environmental and resource costs. Although the maintenance costs of existing water supply systems and the investment costs of new water supply or wastewater treatment systems are easier to calculate, the environmental and resource costs are not so straightforward and do not appear on financial balance sheets. These later costs relate to the damage caused by pollution to, or abstraction from, the ecosystem and to interventions that can potentially cause water scarcity and also harm the ecosystem. It is expected (according to Annex III of the Directive) that when all water-related activities are targeted and when prices reflect the real cost of water consumed, that a more sustainable and efficient use of the resource will be achieved. Full-cost pricing is also a mandatory part of any river management plan.

The economic analysis of water use will support not only the development of water-pricing policies but will also provide for the possibility of derogations under the umbrella of 'disproportionate' costs. With regard to the

1 Scarcity rent can be thought of as the cost of 'using up' a finite resource as the benefits of the extracted resource are unavailable to future generations.

latter concept, Art. 4 states that exemptions are possible if the costs of reaching the GES are disproportionate. However, in order to evaluate the extent to which this is the case and to assess 'disproportionality', one also has to know the costs and benefits associated with reaching the environmental objectives, in both qualitative and quantitative terms. In order to pass the test, costs should exceed benefits by a significant margin in a cost-benefit framework. Cost-benefit analysis (CBA) is an analytical tool based in welfare theory, which is conducted by aggregating the total costs and benefits of a project or policy over both space and time (Hanley and Spash, 1995). A project or policy represents a welfare improvement only if the net benefits are positive. As different management options yield different net benefits, the option with the highest net benefits is the preferred or optimal one. In this context, CBA will be a valuable economic analysis tool in terms of deciding whether a management plan to achieve GES should be put in place in cases where it seems it may be 'disproportionally expensive' to achieve the WFD objectives by 2015.

Monetary valuation is highlighted within the Directive through Arts 9, 11 and 4. As noted above, Art. 9 introduces the notion of cost recovery of water services for all member states. That implies that water services should not be subsidised from the general budget as is the case in Ireland but instead the price of water should include and reflect the environmental and resource costs of water services. Environmental valuation methods can be employed to assess the magnitude of these environmental and resource costs. As specified in the AquaMoney Policy Brief No. 1 (Aquamoney, 2006) 'in the context of selecting cost-effective programmes of measures (Art. 11), environmental and resource costs and benefits can signal to what extent the environmental objectives are met, and if not, what the associated costs are, including residual environmental damage costs and any costs arising as a result of an inefficient allocation of water and pollution rights'.

As noted above, Art. 4 sets out the environmental objectives of the Directive as well as the exemptions from these objectives based on disproportionate costs. Hence, it introduces the need to value benefits in monetary terms in order to make them comparable to the costs. Although the overlying assumption of positive net benefits justifies the use of cost-effectiveness analysis rather than CBA, wherever it appears that the first is not demonstrably worthwhile, CBA is necessary (Morris, 2004). Generally, the fact that the environmental standards to be met are predefined makes the issue of economic efficiency obsolete – requiring only a CBA in cases of derogation.

As a result, the requirements of the WFD implementation involves estimating both direct and indirect costs and benefits to be considered in each management plan (Hanley and Black, 2006). Regarding the nature of benefits, the Directive supports economic analysis that considers direct benefits, such as reductions in the cost of drinking-water treatment downstream when less pollution is discharged into a river, and indirect benefits, such as an increase in jobs if cleaner coastal waters lead to higher tourism levels. Furthermore, more-difficult-to-quantify benefits such as recreation and the availability of healthy ecosystems will also be included. The contribution of valuation methods can be seen as useful in quantifying these latter non-market goods and services. In general, this is an important but the same time difficult task for river basin authorities and will involve them having to consider and evaluate costs and benefits – including environmental criteria. As noted in Kallis and Butler (2001, p.137) 'recovery of environmental costs still remain too vague a mandate in the Directive and is unlikely that it will considerably affect prices in the near future, more so given the lack of a common framework for the monetarisation of environmental costs from the different authorities and Member States'. Therefore, although the concept of environmental and resource costs and benefits plays an important role in the economic analysis of the Directive, so far no practical guidelines or common framework exists for their assessment.

In general, environmental economics is expected to play an important and supportive role in WFD implementation and in particular in justifying spending on environmental protection where applicable. Another challenging area for economists will be the search for cost-effective measures and an integrated approach to address 'land-management' issues because of the significant environmental costs related to diffuse pollution from agriculture (Morris 2004). It can also be argued that the inclusion of benefits values in the assessment of the implementation of the WFD will be important for three main reasons:

1 Their inclusion will promote transparency in water-related decision-making as the trade-offs that motivate a decision will be highlighted;

2 The inclusion of benefit estimates will also assist in identifying policies that produce welfare gains to society and provide information on how the water body is being used, by whom and how these uses will be affected;

3 It will deliver an assessment of how the general public perceives the current status of water resources, how well it is informed about water issues and how relevant water-quality improvements are perceived (AquaMoney, 2007).

In summary, if member states, including Ireland, decide to use CBA in water-management decision-making, then the valuation of the non-market benefits of implementing the WFD will be needed. In order to achieve maximum economic efficiency (where marginal social benefits are equal to marginal social costs) it is necessary to establish the full value of achieving GES of water resources and to incorporate this into private and public decision-making processes (Birol et al., 2006). In this context, the objective of this project is to examine the possibility of employing BT valuation techniques to measure the potential value of achieving GES as defined by the WFD. It should be noted that while some valuation studies for water resource benefits have been undertaken in Ireland (Curtis, 2002, 2003; Hynes and Hanley, 2006; Hynes et al., 2009) there is no comprehensive set of values. This study has therefore the potential to inform the policy debate on a number of levels by exploring the value of achieving GES under the WFD and assessing how successful BT might be in estimating the associated non-market values.

3 Implementation of the Economic Assessments required under the Water Framework Directive in Ireland

Ireland is currently up to date with the requirements of the WFD's implementation timetable. In particular, in 2004 Ireland undertook a characterisation and analysis of all RBDs as required by Art. 5. The report (ERBD, 2005) provided an analysis of the characteristics of RBDs and undertook a review of the impact of human activity on the status of waters. As referred to in its executive summary, 'the report serves as a comprehensive assessment of all waters (groundwater, rivers, lakes, transition and coastal waters), establishes a baseline and identifies priority actions for subsequent stages in the river basin planning cycle' (p.E-4).

As part of the 2005 *National Summary Report for Ireland* (ERBD, 2005), a baseline economic analysis was completed with a preliminary assessment of the value and costs associated with water resources in Ireland. In this context key information gaps were identified along with a proposed strategy to address them. The results presented in the final report, *Economic Analysis of Water Use in Ireland* (Camp, Dresser and McKee [CDM] Associates 2004), provided the foundation for the economic component of the summary national characterisation report under Art. 5 of the Directive. The methodology used for the estimation of water-use benefits in the CDM report suggested an economic impact assessment of key water-using activities and the valuation of abstractive and in-stream water resources in each RBD.

It should be noted that the number of studies that have applied stated preference techniques in the context of valuing economic benefits that derive from the WFD is large and increasing across Europe (Baker et al., 2007; Brouwer 2006; Kontogianni et al., 2005; Spash et al., 2009). A considerable number of these studies have applied the CE method (Álvarez-Farizo et al., 2007; Brouwer et al., 2010; Hanley et al., 2007a; 2006a, 2006b; Kataria 2009; Kataria et al., 2009; Lago and Glenk, 2008). The studies vary in terms of purpose of the study, geographic scale (local, regional, national) and hence the affected population. They also differ with regard to the description of the environmental good, the change in ecological status, the payment vehicle, the survey mode and the validity of the results. This makes comparisons difficult but nevertheless they provide an indication of related values and demonstrate how the idea of valuing benefits within the WFD may be approached, since there is no specific guideline from the EU on how to proceed in this regard.

In the case of Ireland, valuation studies with a focus on river quality improvements are limited. Those studies that are available focus on valuing water-based leisure activities. Hynes and Hanley (2006) estimated through the travel cost (TC) method the mean WTP of the average kayaker using the Roughty river in Co. Kerry, in order to shed light on the conflict between commercial interests and recreational pursuits on Irish rivers. In Hynes et al. (2009) the authors examined the welfare loss to recreationalists from a reduction (50%) in the recreational rating of a river caused by water diversion for agricultural use or the implementation of a hydro scheme. This study used revealed preference data to estimate values for a range of river attributes relevant to kayaking. Another study (Curtis, 2002) applied the TC method to estimate the demand and economic value of salmon angling in Co. Donegal. In addition, in Curtis (2003) the demand for water-based leisure activity (sea angling, boating, swimming and other beach/sea/island day-trips) in Ireland was examined based on data from a nationally representative telephone survey.

A number of other economic studies in Ireland also involve some form of economic appraisal of water-based activity, but do not however measure water-related benefits directly. For example, Lawlor et al. (2007) conducted an economic evaluation of selected water investment projects in Ireland. The authors estimated 'required WTP' with respect to the local population. An apportionment of benefits was made between local and non-local beneficiaries, based on the relative importance or popularity of the water body in question. However, the study did not provide benefit values of use in the appraisal of water-resource initiatives. Bullock et al. (2008) carried out an economic assessment of the value of biodiversity in Ireland and considered the economic and social benefits of biodiversity across a range of sectors, including water. Consumer surplus figures were produced for specialist and general users of rivers and lakes based on certain population assumptions. However, the findings were indicative only and not based on any primary valuation studies.

The only study to date that has estimated the value of achieving GES in an Irish water body directly has been Stithou et al. (2011). In this case the authors used a CE method to estimate the value of improvements in a number of components of ecological status on the Boyne river. The study determined what value the targeted population of the catchment placed on the non-market economic benefits of moves towards GES. In addition, the effect of various factors of observed individual heterogeneity on choice was explored. The study found significant marginal values attached to improvements in a number of river attributes and estimated a welfare impact of €32.70 per person per year for a 'high impact' policy that resulted in the achievement of GES across the entire river catchment. Stithou (2011) also used a contingent valuation (CV) model to estimate the value of achieving GES for the same river and estimated a welfare impact of €30.54 per person per year. These estimates are used later in this report to quantify the transfer errors[2] from the BT methodologies applied and ultimately to determine the validity of this study's BT approaches.

Despite the aforementioned studies that have explored aspects of water quality and valuation in Ireland, no major valuation exercise on achieving GES across a range of Irish water bodies has been conducted to date. This is what this project attempted using BT techniques. Before discussing this secondary approach to non-market valuation in Section 5, Section 4 outlines a number of the primary valuation methods that have been used to value improvements in the ecological status of water bodies.

2 Transfer error is a measure of the difference between the transferred values and available primary estimates for the policy site

4 Methods for Valuing the Benefit of achieving Good Ecological Status

According to Bateman et al. (2006a, p.222) 'the economic benefits [of implementing the WFD] are likely to be many although only a minority are likely to be easily amendable to quantification, for example, reduced water treatment costs. One important motivation for the WFD appears to be the creation of non-market environmental benefits, such as open-access recreation'. In terms of economic measurement tools, valuation methods that could be used to estimate the non-market values that Bateman et al. (2006a) refer to and which are associated with achieving GES under the WFD can be separated into two typologies: (i) 'revealed preference' and (ii) 'stated preference' methods (Hanley et al., 2007).

Revealed preference methods are based upon data drawn from observations of behaviour in real markets from which inferences may be drawn on the value of a related non-market good. The real market acts as a proxy market for the environmental good or service. The use of observable market behaviour and an identifiable link to the non-market good forms the basis for revealed preference valuation techniques, which include TC modelling and the hedonic pricing (HP) method. When estimated, these revealed preference models draw statistical inferences on values from actual choices people make within markets (Boyle, 2003). Less commonly used revealed methods, which may be useful in certain situations, include the replacement cost (RC), averted expenditure (AE), production function (PF) and cost of illness (COI) methods. Market prices could also be considered a revealed preference methodology as the market reveals the price of goods and services based on how people interact within the market.

The HP approach is based on the characteristics theory of value. This states that goods can be described as a bundle of characteristics and therefore the price of the good is a function of these characteristics and the levels of these characteristics (Lancaster, 1966). Hedonic pricing is most commonly used with house or land prices to determine the values of the surrounding environmental levels, such as air quality, distance to amenities or a clean water body. The TC method is used to estimate the value of sites which people travel to for recreation (including hunting and fishing and wildlife viewing). This is based on the theory that the time taken and travel costs represents the price of access to the site. The distance and number of trips can be modelled to represent the WTP of individuals for the site. Various TC models can be used, including count models and random utility models, the latter of which allows for variation in levels at the site and for potential substitute sites (Hynes et al., 2008).

In general, revealed preference methods can only value the use value of an environmental good or service. In order to measure the non-use values of a site or service and also to get the total economic value (TEV) for environmental goods (including both use and non-use values), stated preference techniques are necessary. Stated preference methods are used for eliciting values where there are no markets or proxies for markets. Such values are non-use values which are often a large component of TEV (Stevens et al., 1991). Stated preference techniques derive estimates of consumer surplus via constructed hypothetical markets through which individuals are asked to express their willingness to pay for environmental goods and services such as recreation opportunities. For example, the CV method can be used to seek information directly from survey respondents regarding their maximum willingness to pay (or minimum compensation demanded) for a recreation opportunity (or some other change in environmental quality) or for some specified change in a recreation experience, all within the confines of a hypothetical market. Other methodologies within the stated preference stable include the CE and contingent ranking (CR).

The CV method was the first approach to ask people directly to state their value of a non-market good. The technique, which was initially suggested by Ciriacy-Wantrup (1947), was first applied by Davis (1963) to estimate the benefits of game hunting in Maine, USA. Direct non-market valuation techniques such as CV and CEs do not rely on actual market data to produce welfare estimates but rather produce estimates based on respondents' inferences or choices in a survey. The value elicited through the CV method is dependent on the nature of the hypothetical or simulated market conveyed to the respondents. The CV method normally consists of three major parts: (i) the scenario or description of the policy or programme by which the good/service is going to be provided; (ii) the value elicitation mechanism; and (iii) the socio, economic, demographic and environmental factors that could potentially influence the value placed by individuals (Mitchell and Carson, 1989).

On the basis of the information obtained from the CV survey the WTP of each person can be obtained and aggregated to give the total value of the hypothetical change to the environmental good or service. A respondent's choice or preference may be elicited in a variety of ways in a CV survey. These include an open-ended question where respondents are asked a direct question on their WTP. Alternatively, respondents may be given a value and asked would they pay that amount. If they respond positively they are then asked if they would pay some higher amount. If they respond that they would not be willing to pay the first monetary sum presented to them then they are presented with a lower figure and asked if they would pay that amount instead. This is known as the 'discrete' or 'dichotomous' choice questionnaire format. Other common elicitation formats used in the literature include payment cards and multiple-bound CV methods. Each has been shown to have strengths and weaknesses (see Boyle, 2003 for a full explanation of these techniques).

One major limitation of CV is that it is only capable of considering the value of one hypothetical change or scenario. However, in the context of an environmental good such as ecological status, there are many characteristics to consider – such as biodiversity levels, recreation values, etc. As a result of its ability to measure the marginal value of these characteristics separately, the use of the CE methodology has become very prominent in the environmental valuation literature. In CEs goods are described in terms of various characteristics which can be represented by different levels. For example, a river may be described in terms of its water-quality levels, recreational potential levels, appearance levels and biodiversity levels. Choice experiment can be used to measure non-use values and is thought to be superior to CV (Brouwer et al., 2009).

The first application of CE in the environmental economics literature was a study conducted by Adamowicz et al. (1994) to measure use values associated with water-based recreation. Since then, a large number of CE studies have looked at the non-market benefits of environmental goods. Although the literature demonstrates a particular interest in the use of CE for valuing wetlands (Morrison et al., 1999; Carlsson et al., 2003; Othman et al., 2004) a growing number of CE studies have been used to value river improvements, including benefits arising from the WFD.

5 Valuation Studies examining Water-quality Improvements and achieving Good Ecological Status under the Water Framework Directive

This section is not intended to give an exhaustive review of studies that have examined the non-market benefits of the WFD. Rather, it demonstrates the differences in how researchers have approached WFD valuation within the literature. In the first of these studies, Hanley et al. (2006b) used a CE to estimate the value of achieving GES for two small catchments in Scotland: the Motray and the Brothock. All respondents surveyed lived in the river catchment. Both rivers suffered from low-flow regimes and nutrient-concentration problems, mainly as a result of agricultural impacts on the catchments. Characteristics of the CE, related to the river catchments used in this study, were linked to flora and fauna, appearance of the river and smells emanating from the river. The study observed that a decrease in low-flow days and an improvement in river ecology had a positive effect on survey-respondent utility. It also found that all coefficients relating to preferences for the river attributes had the expected sign. A ‘big improvement’ in river ecology had a household WTP of £24–28 per annum whilst a ‘slight improvement’ in river ecology had a household WTP of £9–17.50 per annum. A reduction in the number of low-flow instances was valued at £2.70–£3.87 per household per month.

In another more recent study, Brouwer (2008) used a dichotomous-choice CV study to estimate the WTP of Dutch households to improve water quality under the WFD. A WTP per household per year was found to range from a mean of €90 (95% confidence interval [CI] €80–100) to a mean of €105 (95% CI €75–135), both in 2003 euro. The average Dutch household pays €470 in water taxes and charges. This study estimated, therefore, that respondents would be willing to pay 22% of what they are currently paying in water taxes or 0.4% of the average household income to achieve the river quality standards proposed under the WFD. Interestingly, respondents who engage in water-based recreation have a WTP of €110 per household per year, which is significantly more than respondents who never undertake recreation in or near water (where their value is approximately €30 per household per year).

Elsewhere, Witteveen en Bos (2006) undertook a market and non-market BT study of the WFD for the Netherlands. They estimated that the bulk of the benefits (40%) would be due to the increased value of houses located near water bodies with improved water quality. The other benefits that they found were associated with the economic values attributed to nature conservation (30%), recreation (15%) and biodiversity (15%).

Bateman et al. (2006c) undertook both a CV and a CR exercise to estimate the value of change in water-quality status for the River Tame in the UK. The study estimated the values for a small, medium and large change in river water quality elicited using both methods. The payment vehicle was a council tax rather than a water rate increase, as a rate increase was felt to be inappropriate and likely to generate high protest values (mainly because of the degree of distrust and opposition to private water institutions stemming from the privatisation of water companies during the 1980s). The CV element of the study found that 23% of respondents were unable to state a WTP and 39% stated a WTP of zero. Including protest votes gave a mean WTP per household per annum of £7.60 for a small improvement, £12.07 for a medium improvement and £18.12 for a large improvement, whilst excluding these protests gave a mean WTP of £9.60 for a small improvement, £15.24 for a medium improvement and £22.89 for a large improvement. Using CR, the WTP per household per annum estimated was £8.64 for a small improvement, £21.34 for a medium improvement and £31.50 for a large improvement. Previous empirical comparisons of the above valuation techniques have also found that values derived from CR tend to exceed those obtained from CV studies (Stevens et al., 2000).

Baker et al. (2007) undertook two CV studies and one CE study to estimate the value of implementing the WFD in England and Wales. The authors used both an open-ended payment card and a dichotomous choice for the CV studies. Over 1,480 interviews were undertaken and a scenario where 33% of improvements are achieved in

2015, 2021 and 2027 respectively were estimated. The payment card CV showed a mean WTP of £38.00 per household per year whilst the dichotomous choice CV produced a mean WTP of £143.50 per household per year for the above scenario. This finding is in line with literature on CV where the payment card method has been found to produce lower WTP estimates than the dichotomous choice method. The CE estimated WTP for two scenarios, the first being the case where 95% of water bodies achieve GES by 2015 and the second where 75% of water bodies achieve GES by 2015 and 95% by 2027. The estimated WTP per household per year for the former scenario was £299.90 and for the latter scenario was £260.00.

In more recent studies, CEs have established themselves as the method of choice in terms of estimating the values associated with improvements in water-related environmental features. Table 5.1 summarises a small number of the studies (whose number is increasing) that have applied the CE technique in the context of valuing economic benefits that derive from improvements in features of water. As will be noted, these studies vary in terms of the purpose of the study and the geographic scale (local, regional and national), and hence the affected population. They also vary in terms of the good, the baseline, the change, the payment vehicle and the survey mode. This makes comparisons difficult, but they nevertheless provide an indication of related values and demonstrate how the idea of valuing benefits associated with water using the CE method has been applied in the literature. A further breakdown of CE and CV studies that may be used in BT to estimate the value of achieving ecological improvements is contained in a database put together as an output of this project at: http://www.nuigalway.ie/semru/bt.html.

Observing the studies in Table 5.1 it is obvious that there is no common approach to water valuation. Even within CEs, there is a wide variety of attributes employed across studies. The ecology parameter may be present in all CE studies but even then it may be perceived and conceptualised differently. Another difference among studies that value water improvements is that there is no uniform approach in terms of how to measure the scale or boundaries of the 'good'. As a result, some studies focus on a specific part of the river, in some cases on its urban stretch (Hanley et al., 2006a), on the main channel of the river only (Kataria et al., 2009), the whole catchment (Hanley et al., 2006b), sub-basins zones (Brouwer et al., 2010), components of the river basin (Poirier and Fleuret, 2010) or on local, regional and national areas simultaneously (Baker et al., 2007). As a result, the main differences in the estimation benefits are observed in the degree of 'benefit inclusion' (valued as a bundle or separately) and the boundaries or size of the good. As a consequence of the latter, differences are also observed in terms of the heterogeneity of the affected population. Other elements that vary include the nature of the good (rivers, lakes, coastal water) and the payment mode (council tax, water rates, VAT increase). Hence, these differences influence the relative result, which means that making comparisons across different studies is difficult.

Table 5.1. Choice experiment (CE) studies on valuing water and water body features.

Study	Country	Attributes	Levels
Kragt and Bennett (2008)	Australia	(i) Native riverside vegetation (ii) Rare native animal and plant species (iii) Seagrass area (iv) One-off levy on rates collected by the Tasmanian government	(i) kilometres (ii) number of species present (iii) hectares (iv) $A 0, 30, 60, 200, 400 or 0, 50 , 100, 300, 600
Bennett et al. (2006)	Australia	(i) Fish species and populations (ii) River's length with healthy vegetation on both banks (iii) Native waterbird and animal species with sustainable populations (iv) River suitable for primary contact recreation without threat to public health. (v) Compulsory one-off payment to trust fund	(i) % of species (ii) & (iv) % of river adapted to the background environment of each of the three rivers considered. (iii) number of species (v) $A 0, 20, 50, 200

Continued overleaf

Table 5.1. Choice experiment (CE) studies on valuing water and water body features. *cont.*

Study	Country	Attributes	Levels
Van Bueren and Bennett (2004)	Australia	(i) Species protected (ii) Farmland repaired or bush protected (iii) Waterways restored for fishing or swimming (iv) People leaving country areas every year (v) Annual household levy	Levels for national CE: (i) No. of species protected (ii) Millions of hectares rehabilitated (iii) No. of km (iv) No. of people leaving annually (v) $A 0, 20 to 200
Robinson et al. (2002)	Australia	(i) Riparian vegetation (ii) Aquatic vegetation (iii) Good or very good appearance (iv) Additional levy on council rates (per year)	(i) To (iii): % of river length (iv) Among others $A 0, 40, 60
Heberling et al. (2000)	USA	(i) Uses of stream (ii) River restored (iii) Travel time from home to site (iv) Easy access points (v) Increased water bill payments per year (for next 10 years)	(i) 'Drinkable, fishable and swimmable' (ii) Miles (iii) 10min, 30min,2hs (iv) 'limited', 'excellent' (v) $5, 30, 100, 250, 500, 750
Poirier and Fleuret (2010)	France	(i)-(iv) Attributes are defined as components of the river basin: coastline, River Touques, River Dives, River Vie (spatial/site specific attributes) (v) Annual voluntary contribution	(i) to (iv) Two levels for each attribute : *status quo* level and good level (v) €0, 10, 20, 30, 40
Kataria (2009)	Sweden	(i) Fish (ii) Birds (iii) Benthic invertebrates (iv) River margin vegetation and erosion (v) Additional annual cost for the household	(i) % increase of fish stock (ii) improved conditions for birds' life: Yes, No (iii) Species richness: High, Moderate, Considerably reduced (iv) Broad to narrow beach combined with various degrees of plant species and biomass growth (3 levels) (v) 0, 200, 375, 600, 850, 1175, 1400 SEK
Baker et al. (2007)	England, Wales	(i) Status of local area in 8 years' time (ii) Status of England and Wales in 8 years' time (iii) Status of England and Wales and local area in 20 years' time (iv) Increase in water bill and other household payments	(i) Different combinations of % of low, medium and high quality in local area at time=0 (current conditions) and at time=8 (in 2015) (ii) Different combinations of % of low, medium and high quality in national area at time=0 (current conditions) and at time=8 (in 2015) (iii) 95%, 75% of current water-quality status (iv) £0, 5, 10, 20, 30, 50, 100, 200
Álvarez-Farizo et al. (2007)	Spain	(i) River ecology (variety of aquatic plants, fish and birds) (ii) Surroundings of the river (litter, smell, visual quality of water, riverside vegetation, erosion) (iii) Supplies of water for urban and agricultural purposes (iv) Increase in the cost of shopping basket	(i) High and low diversity (ii) High and low quality (iii) Guaranteed or subject to fluctuations (iv) Increases of €1, 2, 5, 8 and 15
Hanley et al. (2007a)	England	(i) No. of reaches treated (ii) Bad odour (iii) Ecological condition (fish deaths and invertebrate abundance) (iv) Increase in water bills per year	(i) None, reach 1, reaches 2, 3 & 4 (ii) Days a year (iii) Poor, small improvement, medium, large and very large improvement (iv) £0, 6, 12, 18, 24

6 Benefit Transfer

As the brief description of previous research above suggests, each economic valuation methodology has its own strengths and limitations, thereby restricting its use to a select range of goods and services associated with water quality. However, the policy tool of BT can take the results of these studies to form the bedrock of practical policy analysis. Primary valuation research, while being a 'first best' strategy, is very expensive and time consuming. Thus, secondary analysis of the valuation literature using BT is a 'second best' strategy that can nevertheless yield very important information in many scientific and management contexts (Rosenberger and Loomis 2000). When analysed carefully, information from past studies published in the literature can form a meaningful basis for water-management policy valuation.

Therefore, a more cost-effective approach for the valuation of water-quality improvements can be achieved through the application of BT where estimated values are transferred from previous studies of similar changes in environmental quality to a new policy situation. An example is that of Johnson et al. (2008) who used BT in a stated preference study in England and Wales in order to calculate public WTP for a reduction in risk of illness resulting from swimming in contaminated river waters in Scotland. The study was framed in the context of the EU Bathing Waters standards and the WFD. Elsewhere, the application of BT in the context of the WFD has been examined and tested in Hanley et al. (2006b) by applying CE in two similar rivers and then exploring the possibility of using BT.

It is worth noting that the UK's Environment Agency has previously collated UK studies on benefit valuation and has issued guidance on the use of such values in BT for the appraisal of water resource improvement initiatives. Goodbody (2008) evaluates the possibility of making use of values derived in other countries, in the absence of original studies in Ireland, and in particular benefit values from UK. The report concluded (p.23) that 'the benefit values mandated in the UK are the most appropriate as they refer to the benefits of incremental changes in water-quality status. However, these benefit values are the result of relatively few studies in some instances. There is also some evidence that the benefit values are low in relation to the few Irish estimates that have been made'. It could be argued that BT is a good alternative for Ireland given the large number of site-specific valuation studies that have now been conducted right across the EU and the almost complete absence of such studies in Ireland.

Brouwer (2000) has argued that the term 'value transfer' or 'environmental value transfer' should be used instead of BT as non-market costs can be transferred in the same manner as benefits. Both terms have found common usage in the literature (e.g. Wilson and Hoehn, 2006; Bateman et al., 2006b; Spash and Vatn, 2006; Brenner et al., 2010). However, as non-market costs were not examined here, the current study continues to use the term BT. Benefit transfer has been applied in a wide variety of natural resource, recreational and environmental contexts, including in the areas of water-quality management (Bergland et al., 1995; Luken et al., 1992), health risks associated with water quality (Kask and Shogren, 1994), air quality (Rozan, 2004) forest management (Bateman et al., 1995) and the WFD (Hanley et al., 2006a).

A number of methods are used to transfer values between the study and policy sites. The most straightforward is to use the un-adjusted WTP estimate from one or more study sites, and apply their average value to the policy site. This method is referred to as 'unit value transfer'. However, it has been noted that the simple unit value transfer approach may not be suitable for transfer between countries with different income levels and costs of living (Navrud and Ready, 2007; Ready et al., 2004), and also that the method neglects other sources of variation in values entirely. An extension to the unit value transfer method is where WTP values are adjusted for differences in real incomes, for example, between study and policy sites. If the researcher does use the income-adjusted unit transfer approach, it is recommended that the population characteristics between the study and policy sites should be as similar as possible (Navrud, 2007).

An alternative BT approach is to use a 'function transfer' method. Loomis (1992) argues that transferring the entire benefit function increases the validity and reliability of the transfer. Rosenberger and Stanley (2006) point out that, by transferring the benefit function, the practitioner can make adjustments to value estimates based upon a range of characteristics of the policy site as well as characteristics of the benefiting population. This involves using the original WTP function from a study site and using input values from the policy site to generate the mean WTP. Meta-analysis (MA) is a more complex form of value function transfer: this uses a value function estimated from multiple study results together with information on value determinants for the policy site to estimate policy site values. Meta-analysis is commonly described as 'the statistical analysis of a large collection of results for individual studies for the purposes of integrating the findings' or the 'analysis of analyses' (Glass, 1976). Meta-regression analyses assume the existence of an underlying meta-valuation function that relates the magnitude of empirical estimates of value to characteristics of the study site (Rosenberger and Stanley, 2006). Many meta-regression analyses have been conducted in environmental and natural resource economics; Johnston and Rossenberger (2010) are a general example and Wilson and Liu (2008) are a water resource example. The use of spatial micro-simulation techniques for BT is another form of value function transfer that has been recently suggested by Hynes et al. (2007) and Hynes et al. (2010).

A successful indication of the overall worth of BT approaches is whether or not the transferred values are similar to equivalent primary estimates for the policy site on the basis of some statistical criteria. The presence of a transfer error in the BT process is in the first instance caused by a number of factors, including the quality of the primary study undertaken and of the site data, similarity between sites and the similarity of type of good, service or change being measured (Brouwer, 2000). These transfer errors are of great concern in the BT literature, as they provide confidence in the final valuation of the policy site (Colombo and Hanley, 2008). While one of the main reasons for using BT to measure the value of achieving GES in Irish water bodies is the lack of primary estimates for the policy sites, this study's transfer approaches can still be tested against the Stithou et al. study conducted in the Boyne river catchment. Following Bateman et al. (2000), the transfer error for environmental good k (in this case GES value), is calculated as Eq. 6.1:

$$TransferError_k = \frac{TransferredEstimate_k - PolicySiteEstimate_k}{PolicySiteEstimate_k} \times 100 \qquad \text{(Eq. 6.1)}$$

Rosenberger and Stanley (2006) have broken down transfer errors into three types: (1) generalisation error; (2) measurement error; and (3) publication selection bias. Generalisation errors occur when estimates from study sites are adapted to different policy sites. Therefore, it would be expected that sites with similar geographies, market size and socio-economic characteristics will have lower generalised errors. Where a BT uses a function transfer or MA this is generally associated with a lower generalised error as the function or MA will allow for differences between the sites. Chattopadhyay (2003) found that function transfers outperform unit value transfer under circumstances where the dissimilarities are forced.

Measurement errors are generated from the primary research on which the BT processes are based. These are mainly caused by the decisions made by the researchers of the original study site on what methodology to use and which data to report. For instance, they may exclude results which although may be important to the BT are statistically insignificant to the authors of the original research and BT based on these studies may under- or over-estimate the value being transferred (Rosenberger and Stanley, 2006).

Publication selection bias occurs because journals have a preference for reporting novel methods and applying standard methods in newer situations. This is the opposite of what many in the BT literature (e.g. Brouwer, 2000; Johnston and Rosenberger, 2010; McComb et al., 2006) advocate, which is a more standardised methodology and numerous estimates of the same changes, goods and services in a variety of conditions, times and places, and including replicate studies.

Johnston and Rosenberger (2010) have noted that the general consensus within the literature is that function transfers generally outperform unit transfers. However, other studies show the opposite. Brouwer (2000) found that the unit transfer approach provided a lower range of transfer errors for half of the BT studies that he had reviewed. In his review of 12 studies the highest unit transfer error was 75% while the highest function transfer error was 475%. Rosenberger and Stanley (2006) also found high errors ranging from 475% for function transfer to 577% for unit transfer, and 7028% for meta-analysis. Despite the high transfer errors in some studies, Liu (2007) found that only 2.5% of peer-reviewed papers had transfer errors above 100% and that 40% of this literature had transfer errors less than 10%.

Johnston and Rosenberger (2010) point out that, notwithstanding a few exceptions, the literature in general provides little information with regard to which approach may reduce transfer error to 'acceptable levels'. Indeed, acceptable levels are still undefined in the broader literature (Spash and Vatn, 2006). While Loomis (1992) argued that the adoption of function transfer might reduce the need for transferring between similar sites, the current consensus in the literature appears to be that site similarity is the most important aspect in achieving low transfer errors. Site similarity includes similarity in terms of populations, resources, market structures and other attributes. Some authors note that MA may reduce transfer errors and lead to more robust estimates, but this will be dependent on the quality of the MA, which may be compromised by 'inadequate methods, insufficient commensurability across included studies and the difficulty of applying MA to non-experimental data' (Johnstone and Rosenberger, 2010, p.484).

Benefit transfer has attracted considerable controversy, some of which may relate to the practice of placing monetary values on the environment or ecosystem goods and services. Concerns are also raised over the quality of available data and the inadequacy of primary data (Green, 2003). However, Smith et al. (2002) comment that the inability to undertake additional primary studies because of resources constraints is the reason why BT is used in the first place. Therefore, the choice at hand is often between either a BT study or a qualitative judgement (Smith et al., 2002).

For smaller countries such as Ireland, where there is a limited amount of primary studies, BT using non-Irish primary studies is often the only option available. Ready and Navrud (2006) noted that this necessitates accounting for certain complications and it may be necessary to adjust for patterns in WTP. As noted above, factors that may be needed to be adjusted include currency conversion, wealth and income measures, user attributes, cultural differences, and the extent of the market. Bateman et al. (2006b) also point out that the extent of the market for the environmental good may be a more significant determinant of the aggregate value placed on an environmental good than the average value that an individual holds for a change in that good or service. The authors also point to the 'distance decay' effect – the further the population is from where the environmental benefit occurs, the lower their WTP will be. Finally, the authors showed that the 'economic jurisdiction' (an area that incorporates all those who gain economic value from a project) is often smaller than the political jurisdiction (the area within some administrative boundary).

Bateman and Langford (1997) found that for national sites of importance (e.g. Norfolk Broads National Park in Eastern England) the WTP was measurable across the country but decayed across three zones at distances of 40km, 150km and 260km by 30%, 64% and 63%, respectively. Bateman et al. (2006b) found that for a local river the size of the economic jurisdiction increased from 19.66km for a small improvement to 27.75km for a large improvement. Bateman et al. (2006b) have suggested using GIS to address this issue of 'distance decay' and market extent. Appendix A provides a guide that sets out on a step-by-step basis how to undertake a BT of the non-market benefits resulting in a change to GES in a surface water body.

7 Using Benefit Transfer to Estimate the value of Irish Water Bodies Achieving Good Ecological Status

In the case of the WFD, it has been suggested that there are too many water bodies and too little time in which to undertake a primary valuation study to decide if there are disproportionate costs in achieving GES (Hanley et al., 2006a, 2006b). This is especially true in Ireland where little work to date has been carried out in terms of quantifying the benefits associated with achieving GES across water bodies. Benefit transfer has been suggested as a methodology that could be used to rank the level of benefits associated with different water bodies achieving GES (Hanley et al., 2006a). It must be noted that using such an approach may not give the same level of rankings as a CBA might, as the BT rankings are not undertaken on the basis of net benefits but on the basis of total benefit value.

In this section, a number of BT approaches are used to estimate the value of achieving GES under the WFD. First, a simple unit BT is carried out to estimate the value of achieving GES across 151 water management units in Ireland. Next, an adjusted (for distance decay) unit transfer approach is used to measure the value of achieving GES for the Boyne catchment. The transfer error arising from this BT is examined using a primary CV method estimate of the value of achieving GES in the Boyne from Stithou (2011). Finally, a function transfer approach is employed to examine the value of a number of catchments achieving GES where the value function (with associated attribute coefficient values) used in the BT process was taken from Stithou et al. (2011), and input information for the water bodies examined (in terms of the catchments' environmental attribute levels) was provided by experts in each RBD.

There were a number of water management levels at which BT could be undertaken for the purpose of valuing the achievement of GES under the WFD within Ireland. Pre-WFD, water bodies in Ireland were managed by 'hydro areas'. With the introduction of the WFD, the main WMU was the RBD. To conduct a BT at this level would lead to very coarse rankings. An alternative is that benefit analysis is undertaken at the basic unit of water management known as the 'water body'. There are over 5,500 of these water bodies within Ireland so to undertake a BT study for each of these would be highly intensive in terms of information and time. Therefore, in an effort to balance the coarseness of the seven RBDs against the intensity of conducting BT for 5,500 water bodies, it was decided to focus on WMUs, of which there are 151 within the country. These are used by those in charge of implementing the WFD to develop sub-RBD plans for achieving GES. Based on the scale of the WMUs and their use, it was decided to undertake BT at this level (see Figure 7.1. for a map of WMUs in Ireland).

Using GIS, information on electoral divisions (EDs) was overlaid on the WMUs: EDs are used by the Central Statistics Office (CSO) as the basic political jurisdiction unit within the state. Population statistics from the Census of Population are available down to this level. This therefore allowed an estimation of the size of the relevant population for each WMU. Figure 7.1 also shows the EDs associated with the WMUs. It should be noted that using the WMU to define the population may lead to an underestimation of the benefits of each WMU as only the mean BT estimates are aggregated by the population within each WMU. However, it may be the case that persons within one WMU may value water bodies in other WMUs that are not accounted for in this model. Further, for WMUs near the border with Northern Ireland, only EDs within the Republic were used, so this will lead to an underestimation of values in these WMUs relative to WMUs wholly within the Republic. Therefore, while the benefit estimates derived here allow a comparison of values across the different WMUs, using the benefit estimates generated here directly in a CBA could result in much higher cost/benefit ratios than may be the case. The methodology outlined in Appendix A should be followed for estimating benefits for use within CBA.

Figure 7.1. Map of electoral divisions (EDs) sorted by overlapping water management units (WMUs).

While overlapping the WMUs onto the EDs allows for the estimation of the relevant population residing in the catchment, this still does not account for tourists coming to the region. Therefore, this model also tries to account for foreign tourists coming to Ireland and visiting the different catchments by calculating what is termed 'tourist-resident equivalent' numbers for tourists. The data for foreign tourists in Ireland is available at county level for five different tourist markets (Fáilte Ireland, 2011) (see Table 7.1). However, the total numbers visiting each county was greater than total numbers visiting Ireland, so the figures at county level were reweighed to match the total visitor numbers for each market.

The tourist-resident equivalent for each county is calculated as follows (Eq. 7.1):

$$R_{eq} = \sum\left(Tourists_{Mkt_i} \times \frac{Tourist\ Days_{Mkt_i}}{365} \times \frac{DTE_{Mkt_i}}{\text{DIE}}\right) \qquad \text{(Eq. 7.1)}$$

where R_{eq} is the tourist resident equivalent, $Tourists_{Mkt_i}$ is the number of tourist per county from tourist market segment mkt_i (in Ireland it is divided into five market segments, Northern Ireland, Britain, Mainland Europe, North America and Other Areas), *Tourist Days* is the number of days spent in the country, *DTE* is the daily expenditure of a tourist and *DIE* is the daily expenditure of the average Irish person. The total local resident equivalent was then added to the local resident population. Once the resident equivalents have been calculated for each county the numbers were distributed over the EDs in that county by the number of housing units. This weights the tourism population towards EDs with perhaps more holiday homes than if residential population was used in the weighting process. Figure 7.2 shows a map of the tourist equivalents for each ED.

The WFD measures water status on a five point scale – (i) bad, (ii) poor, (iii) moderate, (iv) good and (v) high. The marginal WTP or consumer surplus of a change in water status has been shown to be non-linear (Martin-Ortega and Berbel, 2010). This means that people are willing to pay more for a change in water status from 'poor' to 'moderate' or from 'moderate' to 'good' than for a change from 'bad' to 'poor' or from 'good' to 'high'. As each WMU has a number of water bodies at various statuses it was decided to use a weighted WTP for a change in water status. The weight was determined by the percentage of surface water bodies at each status in the WMU. Each percentage was then multiplied by a WTP to change from that status to 'at least good status'. Although some water bodies may eventually achieve high status, the aim of the WFD is to preserve water

Table 7.1. Tourist data from Fáilte Ireland.

Tourist market	Estimate holiday makers (2009)	Daily expenditure ratio (€)	Average stay (days)
Northern Ireland	345,750	1.67	3.5
Britain	1,407,710	1.49	4.4
Mainland Europe	1,994,770	2.59	4.7
North America	1,111,820	2.94	5
Rest of World	291,880	3.10	5

bodies at 'good' and 'high' status and it was this target of achieving at least good status which was used within this ranking exercise. Therefore, if an WMU had all its water bodies at 'good' or 'high' status, then the value of change would be zero as it already has attained 'at least good status'.

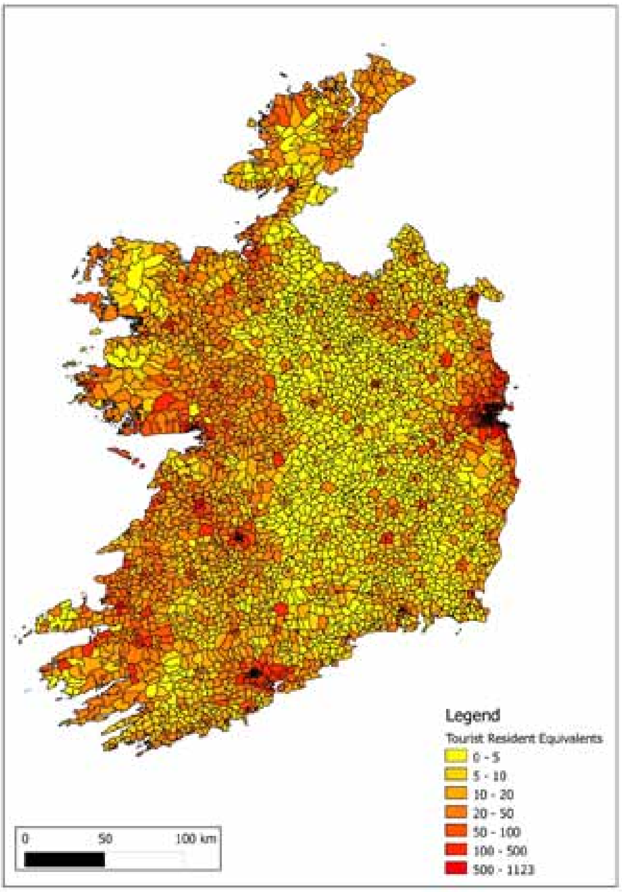

Figure 7.2. Map of Ireland showing electoral division tourist-equivalent numbers.

This does not mean that the value of that resource is zero. If it could be shown that without intervention the water body would decline to a moderate or poor status then the WTP to achieve 'at least good status' could be used as a lower bound estimate of the benefits. The reason this would be a lower bound is that willingness to accept (WTA) compensation is the measure that should be used for a decline in water status. In theory, WTA should be equal to WTP but in practice it has been shown that empirically WTA often exceeds WTP (Hanley et al., 2007a). Moreover, water bodies have values besides those associated with a change in water-quality status. For example, people may still get value from walking along the river bank no matter what the status of the water body, whereas for game fishers (e.g. salmon or trout), this resource may not be available at the water body if it is not a certain status. It should also be kept in mind however that the implementation of the WFD could actually lower the value for some water users, such as coarse fishers, as the type of fish they try to catch prefer murky conditions.

7.1 Unit Benefit Transfer to estimate the Value of achieving Good Ecological Status across 151 Water Management Units

Within this unit BT application, three policy scenarios were examined:

- A large change in water status – a change from 'bad' to 'at least good';
- A medium change in water status – from 'poor' to 'at least good';
- A small change in water status – from 'moderate' to 'at least good'.

The database of water-quality related estimates derived from the literature in this project is available to download at: http://www.nuigalway.ie/semru/bt.html This database contains over 200 estimates from the literature. However, only five studies from the database were considered useful for estimating the change in water status (Bateman et al., 2009; Del Saz-Salazar et al., 2009; Georgiou et al., 2000; Hanley et al., 2006a; Martin-Ortega and Berbel, 2010). These were from similar study sites to Irish catchments with similar population characteristics. They all also measured the WTP of households for various changes in water status. The average values across the studies for the three scenarios used are shown in Table 7.2. The values were adjusted for exchange rates in the year of study, gross national income ratio purchasing power parity (GNI PPP) between the original country of the study and Ireland in the year of study and finally for inflation using the consumer price index (CPI).

As can be seen from Table 7.2, the values increase with the size of the change in water status. This matches the expectations, but because of the small number of value estimates used for large and small changes this does not hold for the 95% lower bound, and the marginal changes are not as expected from the results found by Martin-Ortega and Berbel (2010).

Table 7.2. Average benefit value used for changes in water status (household per year).

Change	Mean (€)	Marginal change (€)	Standard deviation (€)	N	95% Lower bound (€)	95% Upper bound (€)
Large change	66.46		45.59	2	3.28	129.63
Medium change	38.98	€27.48	20.81	5	20.73	57.22
Small change	31.77	€7.21	20.72	3	8.34	55.22

N is the number of estimates used from the 5 studies used in calculating the policy changes.

Using the mean values from Table 7.2 and weighting them by the percentage of water status in each WMU (this latter information was sourced from the relevant RBD Management Plans available to download at http://www.wfdireland.ie/documents.html) gives the average benefit value for that WMU of achieving at least good status. This can be expressed as follows (Eq. 7.3):

$$\textit{WMU Weighted WTP}_i = \sum(\textit{WTP}_j \times \%\textit{WaterStatus}_{jWMU_i}) \qquad \text{(Eq. 7.3)}$$

where *WMU-weighted* WTP_i is the weighted WTP for the WMU_i, WTP_j is the WTP for change in from water status j to 'at least good status' (e.g. large, medium or small) and $\%WaterStatus_{jWMU_i}$ is the percentage of water bodies at status j within WMU i.

To get the value of the change in water status for this ranking exercise, the WMU-weighted WTP is multiplied by the number of persons or the number of households (which is the case in this exercise). Tourist resident equivalents are divided by 2.1 (average number of persons above 15 in each household according to the catchment Census ED data) to get a tourist-equivalent household. For each WMU, the number of households and tourist-resident equivalent households are summed, and the aggregated amount multiplied by the WMU-weighted WTP for a change in water status. This is shown below in Eq. 7.4:

$$\textit{WMU}_i\ \textit{Value} = \left(hh_{WMU_i} + \frac{R_{eqWMU_i}}{2.1}\right) \times \textit{WMU}_i\ \textit{Weighted WTP} \qquad \text{Eq. 7.4}$$

The results of this exercise are shown in Figure 7.3. The values are driven by the population size of WMUs and the average water status of WMUs, which are found to be negatively correlated. Two WMUs show a value of zero euros: Tempelrainy WMU which is a small coastal WMU in the Eastern RBD with one water body and an overall status of 'good' and Derry WMU, a hilly agricultural WMU in South Eastern RBD, with 16 water bodies with 94% at good status and 6% at high status. This zero value is because all water bodies within these WMUs were already at a level of at least GES.

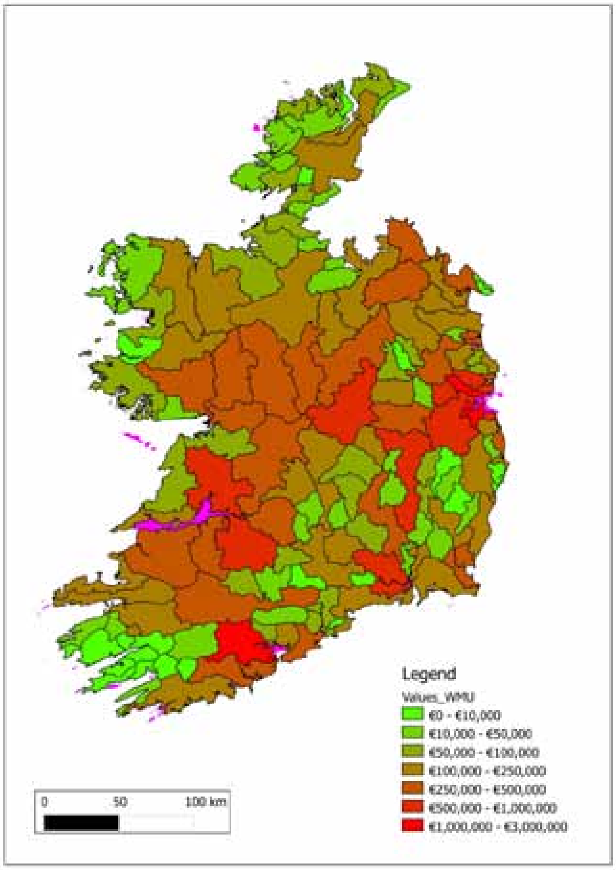

Figure 7.3. Map of Ireland showing value of the benefits per year of implementation the Water Framework Directive.

Note that purple areas are not covered by WMUs.

The ten lowest valued WMUs (other than the two with zero euro value) are shown in Table 7.3 and they tend to be located in the south-western and western parts or in upland areas of Ireland and are of relatively small size.

Table 7.3. Value of achieving good ecological status (GES) for the 10 lowest ranked water management units (WMUs) in Ireland (ignoring 2 with €0 value).

River Basin District (RBD)	WMU	Value (€)
South Western RBD	Sheen	388
South Western RBD	Glengarrif	1,296
South Eastern RBD	Tar	2,610
North Western RBD	Eske	2,887
South Western RBD	Upper Bandon	3,395
South Western RBD	Licky	3,885
Western RBD	Kilary Harbour	4,440
South Western RBD	Blackwater Kerry	4,583
North Western RBD	Cooley Peninsula	5,322
South Eastern RBD	Clodiagh WD	5,579

Examining the higher-valued WMUs shows that they are nearly all in the eastern part of the country, particularly surrounding Dublin city (apart from the Lower Lee/ Owenboy WMU which is located in Cork city and its suburbs) (Table 7.4). This is easily explained: the high numbers of households in the urban areas generate high aggregate values. Also, the higher-valued WMUs have a tendency to be physically large in area, and thus have a higher probability of having a larger population relative to smaller WMUs.

Table 7.4. Value of achieving good ecological status (GES) for the 10 highest ranked water management units (WMUs) in Ireland.

RBD	WMU	Value (€)
Eastern River Basin District	Tolka	2,800,352
South Western River Basin District	Lower Lee/ Owenboy	1,459,072
Eastern River Basin District	Cammock	1,416,838
Eastern River Basin District	Dodder	1,175,405
Eastern River Basin District	Shanganagh	792,670
Eastern River Basin District	Santry Mayne Sluice	774,354
Eastern River Basin District	Ryewater	744,621
Eastern River Basin District	Liffey	733,657
Eastern River Basin District	Barrow Main	691,336
Shannon River Basin District	Brosna	634,282

It should be noted that these values do not include the area within Dublin city centre which accounts for 128,000 households. If these households were classed together as an WMU, based on water status in nearby WMUs, achieving GES could be valued between €2.3 and 4.8 million per year. Another matter with this ranking exercise is that it assumes that people value water status based on river water bodies and does not take into account lakes, transitional waters, coastal water bodies or ground waters. Having said that, river water bodies interact and influence all of these, so it could be argued that the status measures used in the river catchments are a fair measure of the water status of those other types of water body in the WMUs.

7.2 Adjusted Unit Transfer for Boyne Catchment

In the second approach to applying BT to estimate the value of achieving GES on Irish water bodies an adjusted unit transfer was undertaken for the Boyne river catchment. The catchment, which is composed of 9 WMUs and 128 water bodies, covers large parts of Co. Meath and smaller parts of Cos Cavan, Kildare, Louth, Offaly and Westmeath. There are 129 EDs containing 57,999 households within the catchment and a total population of 172,239 persons (CSO, 2006). The Boyne river is the main river within the catchment, rising near Edenderry in Co. Offaly and draining the south-western area of the catchment. At its convergence with the Blackwater river (which drains the north-west of the catchment) lies Navan town. The town of Drogheda lies at the mouth of the Boyne estuary where the entire catchment drains into the Irish Sea.

The water status of the water bodies within the Boyne catchment is shown in Fig. 7.4. As can be seen, most of the bodies within the catchment are at moderate or poor status with some tributaries showing good status. Two sections of the main Boyne channel are shown at good status but there are no river water bodies at high status within the catchment. Agriculture is the predominant land use: 91% of the catchment is occupied by arable lands or pasture. Because the population density increases as one moves from west to east in the catchment, it could be assumed that the pressures in the western part of the catchment are predominately diffuse while those in the eastern part may be more point source dominated, such as industry or urban wastewater discharges.

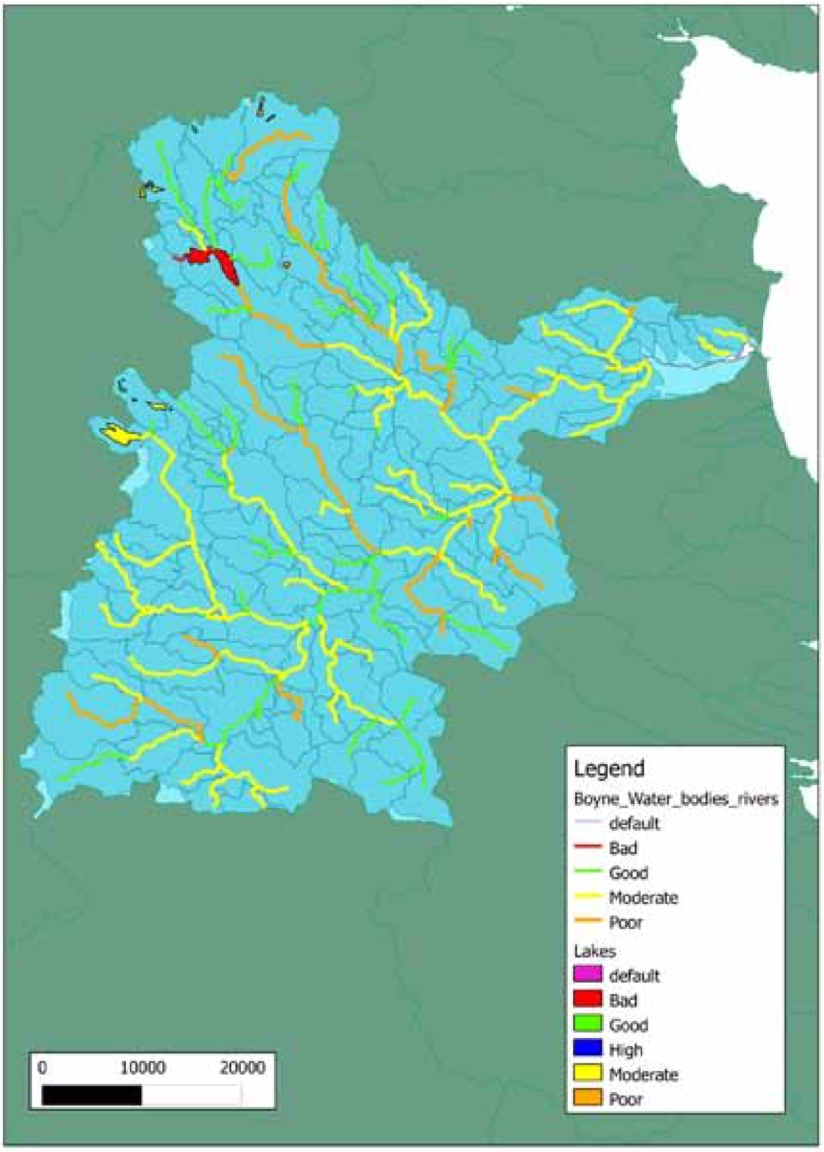

Figure 7.4. Status of water bodies within the Boyne catchment.

Based on data from EPA (http://gis.epa.ie/DataDownload.aspx).

The Three Rivers Project[3] has previously demonstrated that the Boyne river, along with the Suir and the Liffey rivers, are regarded as 'valuable, national and regional resources having major importance in terms of natural and cultural heritage, tourism, recreation and water abstraction for public and industrial uses' (M.C. O'Sullivan Consulting Engineers [MCOS] 2002, p.9). In addition, following the Three Rivers Project, the Boyne was one of the rivers in which the national decline in water quality was deemed to be reflected. Therefore, the river can be considered as a representative water body of Ireland where moderate improvements in water quality are likely to be needed to meet GES.

3 The Three Rivers Project was a government initiative, supported by the European Union Cohesion Fund, which started before WFD came into force and which had as objective to develop catchment based water quality monitoring and management systems for the Boyne, Liffey and Suir river catchments (MCOS, 2002)

Generally, Irish catchments can be thought to be similar to UK catchments and it has been suggested that estimates from UK studies may be the best suited for transfer to Irish water bodies in the absence of suitable Irish studies (GEC, 2008). With this in mind, only peer-reviewed studies conducted on similar catchments in the UK were used in the adjusted unit BT presented here. The non-market benefits of meeting GES in this BT exercise were therefore based on the following studies:

- Green and Tunstall (1991) – 12 UK river corridors – CV – WTP per person, Mean €19.60, SD €15.15, population (n) = 173. Chosen for improvement in water quality in number of rivers but details on rivers not available.
- Haney et al. (2003) – River Miriam in the southern UK – CV – WTP per household, Mean €19.27, n = 650. Chosen for improvement in river ecology from an improvement in flow regime.
- Bateman et al. (2006c) – River Tame – CV – Value for a medium change in water status – WTP per household, Mean €32.02, n = 675. Topic examined: changes in water quality. Note that this is for a mainly urban river, which is mainly poor status and the study involved large, medium and small changes in water quality.
- Hanley et al. (2006a) – River Clyde and River Wear – CE – Estimate for changes in river ecology from fair to good – WTP per person, Mean €31.89, SD €4.79, n = 420. Topic examined: changes in river ecology.
- Bateman et al. (2009) – River Aire – CV – Improving water-quality status from moderate to good – WTP per person, Mean €24.60, n = 434. Topic examined: changes in river water-quality status.
- Spash et al. (2009) – Tummel Catchment – CV – WTP per person, Mean €8.39, n = 719, Topic examined: increase in biodiversity through changing river flow.

The value estimates from the study sites have been adjusted for exchange rates in the year of study, GNI PPP between country of study and Ireland for year of study and inflation in Ireland for year of study (using CPI). In addition, the average transfer estimates are also adjusted for distance decay (i.e. estimates are adjusted based on the distance away from the river that the individuals in the catchment live). Bateman et al. (2006b) have examined the concept of distance decay, where the WTP of a person is on average a function of their distance from the site. Usually this takes the form of a non-linear decreasing WTP as distance increases. Therefore, at some point WTP should on average decline to zero. Bateman et al. (2006b) termed this the 'extent of the economic jurisdiction'.

The economic jurisdiction for water bodies of local, regional and nationally importance was based on a study by Bateman et al. (2006b) (see Table 7.5). The Boyne catchment was classed as a regionally important water body because of its size, because the Rivers Boyne and Blackwater and the Boyne Estuary are protected sites (Natura 2000 Sites under the Habitats Directive) and because the Boyne is of cultural and historical significance (the Battle of the Boyne took place here in 1690). Based on the information in Table 7.5, and using GIS, a buffer zone of 30km was placed around those water bodies with 'moderate' status and a buffer zone of 40km was placed around those water bodies with 'poor' status. The 40km zone overlapped the 30km zone but this may not be the case in all BT exercises. Where zones do not overlap they should be joined together to provide the total economic jurisdiction. The results of this can be seen in Figure 7.5.

Table 7.5. Economic jurisdiction size for change in surface water status.

	Size of change in ecological status		
Importance of water body	**Small change (e.g. poor →moderate, good → high)**	**Medium change (e.g. bad →moderate, poor→ good)**	**Large change (e.g. poor →high, bad → good)**
Local Importance	20km	25km	30km
Regional Importance	30km	40km	60km
National Importance	60km	150km	Whole nation

Economic jurisdiction is based on Bateman et al. (2006b).

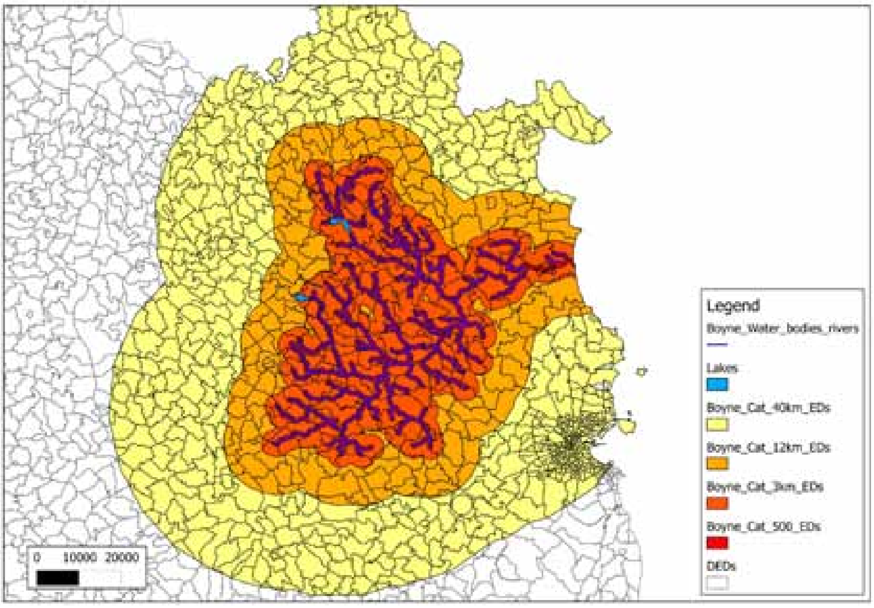

Figure 7.5. Buffer zones for the Boyne catchment and underlying electoral divisions.

Width of buffer zones based on Table 7.5. EDs are show outlined in black.

The result of these buffer zones indicated that there are 1,216 EDs within 40km of any water body within the Boyne catchment. Within this economic jurisdiction there are 633,360 households and 1,430,364 persons over the age of 15. Using the tourist resident equivalent formula outlined previously, the number of tourist resident equivalents within the 40km economic jurisdiction was estimated to be 51,285.

Table 7.6 shows the number of households and persons over 15 and foreign tourist equivalents at various distances from the water bodies within the Boyne catchment. To further explore the use of the distance decay effect on individuals' WTP for a water-quality change at a site, percentages generated from Hanley et al. (2003) (Table 7.7) allow an adjustment of the mean WTP values based on the distance that water body users and non-users in the catchment population are from the river. 'Users' are those that utilise the water resource directly (e.g. swimming, boating and fishing) or indirectly (e.g. walking along the riverside, picnicking and photography). Many surveys divide users and non-users by asking people have they visited the site for any activity (Georgiou et al., 2000); WTP is assumed to be the non-use value for those who haven't visited the site.

Hanley et al. (2003) used four buffer zones and the equivalent ones for the Boyne catchment are shown in Figure 7.5. It should be noted that the outermost zone (12–40km zone) in Figure 7.5 overlaps parts of Northern Ireland. Population and tourist data were however based only on EDs within the Republic of Ireland – therefore, by excluding the relevant population in the zone from Northern Ireland there will be an underestimation of the value of reaching 'at least good ecological status' for this catchment.

Table 7.6. Distance decay zones and associated populations.

Distance from water body	Households	Person over 15	Tourist resident equivalents (individuals)
0–0.5km	19,066	42,322	556
500m–3km	41,305	92,587	1,246
3km–12km	119,758	52,577	1,810
12km–40km	520,410	1,175,697	47,673
0km–40km	633,360	1,430,364	51,285

Table 7.7. Distance decay effect on individuals' willingness to pay.

			Distance decay factor	
	% Users	% Non-users	% Mean WTP of users	% Mean WTP of non-users
0–0.5km	100	0	310.0	
0.5–3km	70	30	240	230
3–12km	26	74	75	65
12–130km	2	98	30	30

Generated from figures reported in Hanley et al., 2003

For the adjusted BT approach, the concept of distance decay was applied by dividing the economic jurisdiction into zones as outlined above. The values for achieving at least good status were estimated by weighting the mean of the six estimates generated from the literature review multiplied by the relevant households or population figures in each zone by the distance decay adjustment outlined in Table 7.7. The population figures used were also adjusted to include tourist resident equivalents. The overall value for achieving 'at least good ecological status' in the Boyne catchment using the adjusted BT approach was estimated to be €13,600,000 with a 95% confidence interval between €7,100,000 and €20,200,000. The value per zone is also broken down by use and non-use value in Table 7.8 and Figure 7.6.

Table 7.8. Breakdown of transferred benefit value between zones and use/non-use in the Boyne catchment.

Zone (km)	Total value (€)
0–0.5	2,200,000
0.5–3	3,800,000
3–12	1,400,000
12–40	6,200,000
Total	13,600,000

Note not all figures add exactly due to rounding up. All figures are for achieving 'at least good status' per annum

The total non-use value is found to be larger than the use value, which is often the case for the valuation of water quality (Bateman et al., 2006b). Figure 7.6 also shows

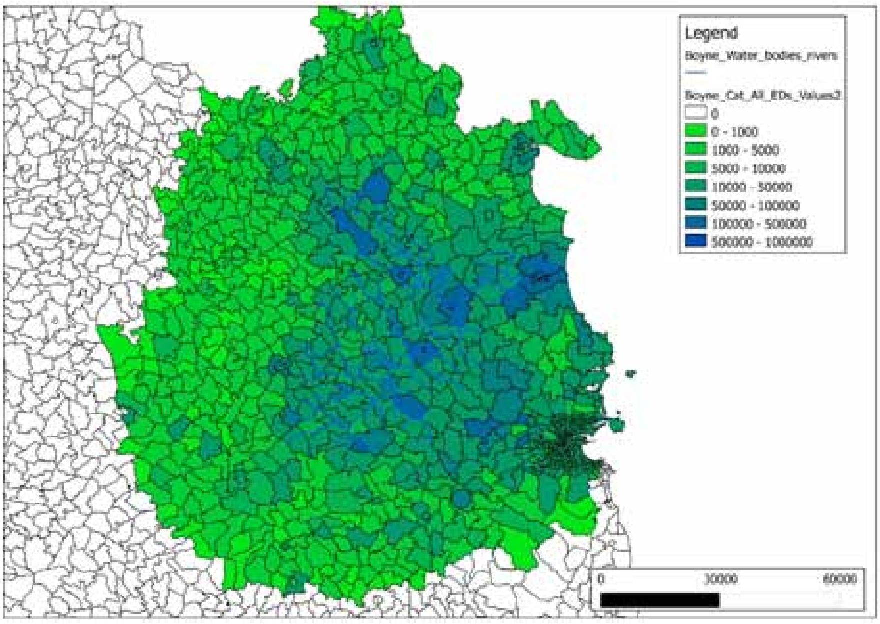

Figure 7.6. Estimated value of achieving 'at least good status' for the Boyne catchment per electoral division per annum. Values shown are in 2010 €s.

that generally EDs with the highest estimated WTP are located within the Boyne catchment and mainly in the eastern part of the catchment, corresponding with the higher populations.

A successful indication of the overall worth of the adjusted unit transfer approach adopted in this case study would be indicated by whether or not the transferred values are similar to equivalent primary estimates for the policy site on the basis of some statistical criteria. As outlined earlier, transfer errors are of great concern in the BT literature, as this indicator is of primary importance in providing confidence in the final valuation of the policy site (Colombo and Hanley, 2008). While one of the main reasons for using BT to measure the value of achieving GES in Irish water bodies is the lack of primary estimates for the WMUs in Ireland, the adjusted unit transfer approach can be tested against a recent study that was conducted in the Boyne catchment (Stithou, 2011). Stithou (2011) used a CV study to estimate the value of achieving GES in the Boyne catchment and this is used to estimate the transfer error in this case. Table 7.9 shows the value estimated for the Boyne catchment by Stithou (2011). The transfer errors are also shown in Table 7.9.

The results show that the BT exercise underestimated the actual value by nearly a third (29%). While this level of error may seem high, this value actually compares favourably with many transfer errors found in the literature. Indeed, Rosenberger and Stanley (2006) reviewed transfer errors in the environmental economics literature and found unit transfer BT error rates varying between 8 and 577%. Lindhjem and Navrud (2009) note that the reliability of international meta-analytic transfers, even with homogeneous valuation methods, similar cultural and institutional conditions across countries, and a meta-analysis with large explanatory power could still result in large transfer errors. Finally, it should be noted that the empirical evidence from the literature has found that international BT is as valid as intra-country transfer (Ready and Navrud, 2006).

7.3 A Function Benefit Transfer for the Boyne River Catchment

The previous two approaches simply used the WTP estimates from one or more study sites and applied their average value to the policy site. The WTP values in this unit transfer approach may be adjusted for one or more factors (e.g. adjustments for differences in income between study and policy sites and for differences in price levels over time or between sites) before the values are transferred between the sites. However, instead of transferring individual benefit estimates, as was done in the two transfer exercises presented above, an entire benefit function can also be transferred. This next step in the complexity of BT is referred to as the 'function transfer' approach. This may involve using the original WTP function from the study site and using input values from the policy site to generate the mean WTP. Meta-analysis is a more complex form of value function transfer: this uses a value function estimated from multiple study results together with information on parameter values for the policy site, to estimate policy site values (Wilson and Liu, 2008). In what follows, the original function from a CE study and use input values from water management experts in relation to the river features from a number of catchments are used to generate the mean WTP for each.

This 'function transfer' approach is conceptually more appealing because more information is transferred. The whole benefit relationship is in effect transferred from the study site(s) to the policy site. The underlying function used as the transfer function may once again be estimated using either revealed preference approaches – such as the TC method and HP – or stated preferences

Table 7.9. Estimated value of achieving 'at least good status' per annum using Stithou (2011) values and the benefit transfer (BT) transfer error.

Scenario	Stithou (2011) (€)	BT exercise (€)	Transfer error (%)
Aggregate WTP to achieve GES	19,100,000	13,600,000	-29

WTP = willingness to pay; GES = good ecological status approaches – such as the CV and CE methods. The following exercise employs a CE model that was previously estimated by Stithou et al. (2011) to estimate the value of achieving GES in the Boyne catchment. The results of this model and a description of each variable in the model are reproduced from Stithou et al. (2011) in Appendix B. In the CE framework, environmental goods are valued in terms of their attributes, by applying probabilistic models to choices between different bundles of these attributes. Individuals will choose to 'consume' the bundle of attributes presented in a choice card that gives them the highest utility. Respondents are asked to provide answers to a sequence of such choice cards. The alternatives/bundles are constructed according to experimental design theory which makes it possible to explore how an individual makes trades-offs in terms of a set of attributes whose levels differ across the choice options on the choice cards.

Using the coefficients for water features in the Stithou et al. model (see Table B2) in a BT transfer the value of achieving GES in a number of water bodies across Ireland is estimated. Experts involved in the implementation of the WFD across Ireland were asked to rate the rivers in their river basin districts using the attributes and levels from the Stithou et al. (2011) CE model. The attributes and levels for which coefficient estimates were available and which are in the model are described in Table 7.10.

The representative component of utility in the model for individual i and for choice alternative j is given as Eq. 7.5:

$$V_{ij} = \beta_0 + \beta_m M_{ij} + \beta_p P_{ij} + \beta_s S_{ij} \qquad \text{(Eq. 7.5)}$$

where β_0 is the constant, β_m is the vector of coefficients attached to the river quality attributes M that follows the normal distribution (β_m ~N(μ,σ²)), β_p the price vector, and β_s the vector of coefficients related to the individual's socioeconomic characteristics S.

The experts interviewed in the RBDs were asked to consider the rivers in their respective catchments and indicate what percentage of each of the river's catchments could be considered to fall under the different levels of each of the attributes M. On completion of these interviews, the percentages of the water bodies representing each level of each attribute were obtained, as shown in Table 7.11.

In order to obtain a measure of economic value represented by the compensating surplus (CS) associated with improvements that result from the achievement of GES in each of the river catchments in Table 7.11, the compensating variation log-sum formula, described by Hanemann (1984) for determining the expected welfare loss (or gain) associated with the policy scenarios, was used (Eq. 7.6):

Table 7.10. Attributes and levels used in Stithou et al. 2011 choice experiment model.

Attribute	Description	Attribute levels
River life: fish, insects, 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 (scrubs, trees) and animals (mammals and birds)	1. Visible erosion that needs repairs 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) (ALL)
Cost	Annual household taxation for 10 years	€0, 5, 10, 20, 40, 80

Table 7.11. Percentage of attribute levels for 15 water bodies ranked by experts in river basin districts (RBDs).

River	River life*			Condition of river banks		Water appearance			Recreational activities		
	Poor	Moderate	Good	Erosion	Natural looking	No improvement	Some improvement	A lot of improvement	No fishing and swimming	No swimming	ALL
Blackwater	10	37	53	10	90	90	5	5	5	5	90
Lee	17	16	67	5	95	95	0	5	2.5	2.5	95
Bandon	26	23	51	95	5	95	5	0	0	0	100
Roughty	0	13	87	100	0	95	5	0	2.5	2.5	95
Laune	18	29	53	95	5	90	5	5	5	5	90
Barrow	19	43	38	0.01	99.9	80	15	5	0	0	100
Nore	23	34	43	0.01	99.9	80	15	5	0	0	100
Suir	21	31	48	20	80	80	15	5	10	10	80
Slaney	10	41	49	0.01	99.9	80	15	5	0	0	100
Owenavorragh	31	38	31	0.01	99.9	80	15	5	0	0	100
Colligan Mahon	5	29	66	0.01	99.9	80	15	5	0	0	100
Ballyteigue Bannow	19	28	53	0.01	99.9	80	15	5	0	0	100
Boyne	26	52	22	60	40	85	10	5	5	5	90
Liffey	13	16	71	10	90	85	10	5	5	25	65
Avoca	13	30	57	20	80	85	10	5	10	20	70

*River life level attribute rankings are not based on expert opinion but on the information on water body status from the Water Management Unit Reports.

$$CV = \frac{1}{\hat{a}_p}[\ln(\sum_{j=1}^{J} \exp(V_j^1) - \ln(\sum_{j=1}^{J} \exp(V_j^0))] \qquad \text{(Eq. 7.6)}$$

where β_p is once again the coefficient of the cost attribute defined as the marginal utility of income, and V_j^0 and V_j^1 represent the deterministic part of the indirect utility function before and after the policy change. The deterministic part of the indirect utility function after the policy change (the achievement of GES) is calculated using the levels for each attribute shown in Table 7.11 and the model attribute coefficents in Table B2. The results of the CV estimation for each of the 15 water bodies as a result of transferring the original Boyne study function are shown in Table 7.12. Of course, transferring the Boyne function to these other sites assumes that the populations in these other catchments have the same preference values as the sample population in the original study.

Table 7.12. Attribute levels associated with achieving good ecological status (GES) for all water bodies.

Attribute	Levels associated with achieving GES
River life: fish, insects, plants	Good
Condition of river banks	Natural-looking banks
Water appearance	A lot of improvement
Recreational activities	Walking, boating, fishing, swimming, all possible

In the original study Stithou et al. (2011) estimate a compensating surplus value associated with the Boyne catchment achieving GES (per household/year) of €32.7 with a 95% confidence interval between €-55.26 and €114.68. Based on the expert opinion of the river attributes for the Boyne catchment and using the log-sum formulae a BT mean value of €51.73 was estimated. This represents a mean transfer error of 58.2%. While this may seem high, it should be noted that the mean BT estimate still falls within the confidence interval of the mean value associated with achieving GES for the Boyne from the original study.

Aggregating over the relevant household population, and once more using the distance decay population totals and decay factors outlined in Table 7.7, an aggregate (non-market) welfare impact of €20,883,240 from achieving GES in the Boyne catchment was estimated. This aggregate function transfer estimate is 54% larger than the aggregate estimate calculated for the Boyne catchment using the unit transfer approach (Table 7.13).

Table 7.13. Mean compensating surplus estimated from benefit transfer (BT) function transfer

River	Mean compensating surplus *(€/household/year)*
Blackwater	33.14
Lee	35.79
Bandon	67.99
Roughty	65.93
Laune	65.64
Barrow	27.90
Nore	29.99
Suir	40.60
Slaney	25.87
Owenavorragh	31.26
Colligan Mahon	26.27
Ballyteigue Bannow	29.72
Boyne	51.73
Liffey	40.36
Avoca	41.20

8 Discussion and Conclusions

This study examined a number of alternative BT techniques that may be used to calculate the benefit value of Irish water bodies achieving GES as specified under the WFD. In particular, the use of the simple unit transfer approach and the function transfer approach were examined. Unit value transfer with income adjustment, and unit value transfer where distance decay in WTP were accounted for, were adopted as the unit transfer methodologies. The function transfer approach employed transferred the results of a CE model conducted in the Boyne catchment to a number of other catchments in Ireland using the opinions of experts in the different RBDs to quantify the parameter values for each catchment that were used as input values in the model. Comparing the results of these BT approaches to the results in the primary studies in the Boyne by Stithou (2011) and Stithou et al. (2011), transfer errors of 29% and 58% for the distance decay unit transfer and the function transfer approaches respectively were found. In this case, it would appear that the unit transfer approach provides more reliable estimates. Other researchers (e.g. Navrud, 2007) have also found that the unit transfer method can produce lower transfer errors than the more complex procedures of value function transfers. This may be caused by the low explanatory power of WTP functions within stated preference studies; and, according to Narvus (2007), methodological choice, rather than the characteristics of the site and the affected populations, has a large explanatory power in determining WTP.

An obvious question is therefore which method is to be preferred in terms of use to quantify the value of achieving GES across different water bodies in Ireland. While the general consensus in the literature is that function transfer methodologies are to be preferred as they allow the researcher to use more information when transferring values from study to policy site the choice between transfer approaches needs to be looked at on a case-by-case basis. However, it should be noted that researchers such as Brouwer (2000) and Rosenberger and Stanley (2006) have found that the unit transfer approach can provide a lower range of transfer errors. Similar to these other studies, it was also found that the simple unit transfer approach gave lower transfer errors. It should be kept in mind that, in the case of the current study, this was based only on a comparison of this study's BT estimates to one primary study.

Overall, results from the current research's validity tests show that the uncertainty in value transfers can be quite large although it can be argued that the transfer errors estimated for the BT estimates for the Boyne catchment are not overly large when one compares them to estimates elsewhere in the literature. Given the magnitude of the transfer errors from this BT analysis, we would argue that any BT estimates produced in order to quantify the benefit value of a water body achieving GES should be used only to compare the relative values across water bodies or where the demand for accuracy is relatively low. The use of BT estimates for making decisions in relation to disproportional costs at single sites is not recommended. In these limited cases where policy-makers feel that the costs of achieving GES may be higher than the aggregate benefits from such a policy intervention, then a primary survey should if at all possible be carried out. If this is not possible, and some form of BT estimation must be employed, then the lower-bound estimate from the BT WTP confidence interval should be used to be as conservatively as possible in terms of quantifying the net benefits.

Where BT can play an important role in water policy formulation in Ireland is in terms of deciding where available funds and resources should be deployed in an RBD or water management area. Comparing BT estimates across water bodies would allow policy-makers in different RBDs to assess which river might receive the highest or the least amount of benefits from any policy intervention aimed at achieving GES. This would allow policy-makers to target scarce resources at those water bodies that will yield the highest net benefits. Of course, consideration would also have to be given to the costs of the implementation of such a policy and how those costs might vary for different water bodies. While primary valuation tools remain the first-best option for estimating the value of achieving GES in a given water body BT can still play an important role in facilitating management planning and in identifying target water management areas for optimum resource allocation and conservation.

It should also be kept in mind that the use of valuation studies within a cost-benefit framework is but one tool that can be used to determine how to allocate resources between the – sometimes conflicting – social, economic and environmental needs of the population. It is also very difficult to incorporate the many issues of specific relevance to the water body being valued into a BT analysis. Therefore, BT should not be used as the sole determinate whether or not to undertake a project to ensure that a water body achieves 'at least GES' or the timing of such a project. Turner et al. (2000) suggested that environmental valuation studies, whether primary or BT, should be used in conjunction with other studies (i.e. integrated modelling, environmental impact statements, physical planning, stakeholder analysis, and/or multi-criteria evaluation) in deciding how to manage such resources. This approach should also be applied to decisions regarding achievement of 'at least good ecological status' in Irish water bodies.

Finally, we would suggest as an avenue for further research a primary valuation exercise to estimate the full range of benefits of water to Irish society. The Stithou et al. (2011) study focused on just one water body but a similar CE study related to water features and reflecting the preferences of the general Irish public (rather than the preferences of those in a single catchment) would enable those responsible for setting water management policy to prioritise funding for features and water bodies that are most highly valued among the general population of Ireland. Such a study would also act as a further validation check of the BT results presented in this report.

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Abbreviations and Acronyms

BT	Benefit transfer
CBA	Cost–benefit analysis
CE	Choice experiments
CSO	Central Statistics Office
CPI	Consumer Price Index
CR	Contingent ranking
CV	Contingent valuation
DPSIR	Drivers, Pressures, State, Impact, Response
EDs	Electoral divisions
EU	European Union
GIS	Geographic information system
GES	Good ecological status
GGS	Good groundwater status
GSWS	Good surface water status
GNI PP	Gross national income ratio purchasing power parity
HP	Hedonic pricing
MA	Meta-analysis
MS	Member state
PPP	Purchasing power parity
RBD	River basin district
SAC	Special Area of Conservation
SPA	Special Protection Area
TC	Travel cost
WFD	Water Framework Directive
WMU	Water management unit
WTA	Willingness to accept
WTP	Willingness to pay

Appendix A. Guidelines for using Benefit Transfer for Irish Water Bodies

This guide sets out how to undertake an adjusted unit-value BT of the non-market benefits resulting from a change to GES in a surface water body (similar to that undertaken for the Boyne catchment in Section 7.2 of this report). It is assumed that the changes are positive with regard to the GES (i.e. a move from bad status towards high status) as most valuation studies determine WTP for improvements in ecosystem goods and services.

For a deterioration in an ecosystem good or service, WTA should be used. In theory WTP should be equal to WTA: however, in practice this has been shown not to be the case (Hanley et al., 2007b).

The steps involved in undertaking the BT are as follows:

1. Define study area and change in surface water status;
2. Identify the benefits arising from such a change;
3. Identify number of beneficiaries;
4. Undertake a literature review to identify relevant values;
5. Adjust values for to allow for income differences, exchange rate conversions and inflation;
6. Apply relevant values of beneficiaries allowing for distance decay;
7. Aggregate the values to give a total non-market economic value on an annual basis.

Once the total non-market economic value is calculated, the value can be integrated into a CBA. In what follows we go into each step in more detail.

Step 1. Define study area and change in surface water status

Study area definition is an important first step of undertaking a BT. The extent of the study area of users and non-users can affect the final aggregated benefits value. Examining users, the change in GES may have upstream effects as well as the normally considered downstream effects. Typically, the study area definition for users will be limited in a physical sense to the area near the banks of the river/lake where change in ecological status is expected to occur or have an impact.

The scale of the change in GES will determine both the value of the benefits and their spatial distribution. To estimate the change in status a particular project may cause, expert advice should be obtained (e.g. from ecologists and environmental scientists). Cognisance should also be taken of data available on the Irish WFD site [http://www.wfdireland.ie/maps.html]. Ceteris paribus, a large improvement in ecological status should invoke greater benefits and have a greater spatial effect than a small improvement. In this guidance document, we base changes in the surface water status on the rating scale used in the WFD, namely high, good, moderate, poor and bad. A change from 'moderate' to 'good' is an example of a small change, whereas a change from 'bad' to 'good' may be considered a large change. Table A1 sets out the definition of changes.

Table A1. Scale of change relative to level change in surface water status.

Level change	Scale of change
1 Level (e.g. poor → moderate, good → high)	Small change
2 Levels (e.g. bad → moderate, poor → good)	Medium change
3 Levels (e.g. poor →high, bad → good)	Large change

Use of GIS can also be incorporated into the study area definition. Publically available GIS data related to rivers, lakes and the WFD is available on the EPA GIS website [http://gis.epa.ie] and publically available GIS data on population and households is available on the CSO website [http://census.cso.ie/censusasp/saps/boundaries/census2006_boundaries.htm].

For both users and non-users the study area is dependent on the scale of the change and the importance of the site. Bateman et al. (2006b) have shown that the 'economic jurisdiction' is often smaller than the political jurisdiction and the larger the improvement the larger the 'economic jurisdiction' (Loomis, 2000 refer to a political jurisdiction as an area concerning some administrative area while

Table A.2. Economic jurisdiction determination based on water body importance and change in water status.

	Radius impacted by size of change in ecological status		
Importance of water body	**Small change**	**Medium change**	**Large change**
Local importance	20km	25km	30km
Regional importance	30km	40km	60km
National importance	60km	150km	Whole nation

an economic jurisdiction incorporates all those who hold economic values regarding a project). Based on similar advice offered by the UK Environment Agency (EA, 2003) the following methodology may be applied to defining the boundary of the relevant non-use population (Table A2). Users are defined as those that use the water resource directly (e.g. swimming, boating, fishing) or indirectly (e.g. walking along the riverside, picnicking, and photography). Non-users are those who have not used the environmental resources for a specified time period (in many valuation studies this period is often specified as the previous 12 months). Many surveys divide users and non-users by asking people have they visited the site for any activity (Georgiou et al., 2000); WTP is assumed to be the non-use value for those who haven't visited the site.

Local importance economic jurisdiction for a given water body is based on Bateman et al. (2006b) and the regional and national importance economic jurisdiction is based on EA (2003). The water bodies in Table A6 at the end of this Appendix are deemed to be of national and regional importance. These are based on Fáilte Ireland Report on Determination of Waters of National Tourism Significance and Associated Water Quality Status (Fáilte Ireland, 2009).

Step 2. Identify the benefits arising from such a change

Once the study area and scale of the change are identified, then the next step is to identify the benefits arising from the change in water status. The use of expert opinion is advised to assess what benefits will arise from the change in water status. The change may be as a result of a change in flow rather than just a change in water quality. Benefits may arise from a change in the consumer surplus due to the change in water status or for allowing new activities to be undertaken or more people to undertake the usual activities. Benefits may be divided between users and non-users. Examples to consider are presented in Table A3 (note that option value or quasi-option values are not included).

Step 3. Identify number of beneficiaries

Beneficiaries can be divided between users and non-users. Two methods are suggested for estimating the total number of individuals benefiting from the change in ecological status at a site. The first is to determine the activities undertaken within the water body and match these with the percentage of the population that are deemed to participate in these activities from Table A4.

Table A.3. Activities which may benefit from a change in water status

Activity	Description of activity
Informal recreation	Walking, picnicking, horse riding, etc. alongside the water body
Angling	Both game and coarse angling (coarse angling may suffer from an improvement in water status)
In-stream recreation	Boating, kayaking, swimming, etc.
Landscape and amenity	Landscape views and amenity value including property prices
Heritage and archaeology	Impact on man-made structures of historical value
Commercial outputs	Effect on inputs into commercial activities including agriculture activities
Non-use values	Value estimated from those who do use the water body but attach some value due to existence, altruistic or bequest value

These figures are based on a water-based recreational activities survey carried out by the ESRI in 2003 (ESRI, 2003). Based on the population within the economic jurisdiction the total number of potential users can be estimated by multiplying the relevant population by the percentage undertaking the activity of interest. The second method that could be employed to estimate the total number of individuals benefiting from the change in ecological status is onsite surveys. These could be used to estimate numbers of users (bankside walkers, photographers, etc.). The non-user population can be estimated by subtracting the estimated number of users from the population of the economic jurisdiction.

The second method to estimate the numbers of users and non-users is to follow the methodology used in Section 7.2 and use the percentages generated from Hanley et al. (2003) for each buffer zone (reproduced below in Table A5) to estimate the number of users and non-users in each buffer zone.

Table A.4. Percentage of users for water-based activities.

Activity	Percentage of population participating in activity
ANGLING	
Freshwater angling for coarse fish	2.20
Freshwater angling for game fish	2.70
Sea angling from the shore	2.50
Sea angling from boat	1.80
Any type of angling	7.20
COASTAL & INLAND BOATING	
Sailing at sea	2.00
Boating at sea in row boats, canoes, etc.	1.10
Boating in power boats, etc. at sea	0.80
Cruising/boating on inland waterways	1.40
Any type of boating or sailing	4.70
WATERSPORTS	
Water skiing, jet skiing	0.60
Surfing, Sail boarding	0.60
Scuba diving, snorkeling	0.30
Other sea sports	0.20
SEASIDE/RESORT TRIPS	
Swimming in the sea	11.70
Whale/dolphin watching	0.30
Bird watching in coastal areas	0.40
Visiting nature reserves, etc. in coastal areas	1.40
Other trips to the beach or seaside	37.60
Other trips to the islands	1.10
Any of the above water-based leisure activities	49.00

ESRI (2003)

Table A.5. Distance decay effect on individuals' willingness to pay (WTP).

			Distance decay factor	
	% Users	% Non-users	% Mean WTP of users	% Mean WTP of non-users
0–0.5km	100	0	310.0	
0.5–3km	70	30	240	230
3–12km	26	74	75	65
12–130km	2	98	30	30

Generated from figures reported in Hanley et al., 2003.

Tourists also use or are aware of water bodies within Ireland. Therefore, some effort should be made to account for tourists within the study area of the relevant catchment. The approach taken by the authors of this report is to convert tourists to resident equivalent households. Data is available at a county level for the numbers of tourists and at a regional level for the five foreign tourist markets in Ireland; Northern Ireland, Britain, Mainland Europe, North America and Other (Fáilte Ireland, 2011).

Using this data, a framework was developed that converted foreign tourists to resident equivalents based on time spent in the country and expenditure as a ratio to the average Irish daily equivalent. The tourist numbers are distributed across EDs based on total housing rather than population or occupied houses as this weights the EDs with holiday homes. The equation used is as follows (Eq. A.1):

$$R_{eq} = \sum\left(Tourists_{Mkt_i} \times \frac{Tourist\ Days_{Mkt_i}}{365} \times \frac{DTE_{Mkt_i}}{\mathrm{DIE}}\right) \qquad \text{(Eq. A.1)}$$

Where R_{eq} is resident equivalent, $Tourists_{Mkt_i}$ is the number of tourists from market *i*, $Tourist\ Days_{Mkt_i}$ is the average number of days that the tourist spends in Ireland, DTE_{Mkt_i} is the average daily expenditure by tourist from market *i* and *DIE* is the average daily expenditure by an Irish person. Combining these resident equivalent figures to the relevant population of users and non-users gives the total number of potential beneficiaries in the jurisdiction.

Step 4. Undertake a literature review to identify relevant values

Based on the expected benefits a literature search should be undertaken to identify relevant values of non-users and users. Both market prices and non-market prices should be used where available. Where no values are available, the final report should note this. A database of water valuations from across Europe is available at: http://www.aquamoney.org/sites/results.html but newer valuations may be available. Values used should be defensible and based on peer-reviewed articles where possible. It may be acceptable to use non-peer reviewed articles if no value is available and if the study is defensible. A database should then be created into which all the valuation estimates could be stored and analysed. The database of water-quality-related estimates developed in this project is available to download at: http://www.nuigalway.ie/semru/bt.html

Step 5. Adjust values for income differences, exchange rate conversions and inflation

The next step in the process involves standardising the benefit values in the estimate database. This is achieved by first converting the estimates from the literature to Euro (€) values using the appropriate nominal exchange rate and then by adjusting for inflation based on the Irish Consumer Price Index (CSO, 2010) to covert to prices in the relevant period of study. In the majority of cases the mean estimate values were used where reported or where unreported, and confidence intervals were given instead, and the lower bound value used to give more conservative value estimates. Transfer values should be adjusted for exchange rate differences in the reference year of the study, differences in income (purchase power parity adjusted) between countries or between study and policy area where available, should also be adjusted based on the rates for the year of the study. The values should then be adjusted for inflation so all values are equivalent for the year of interest for analysis at the policy site.

Usually this takes the following form: (Eq. A.2)

$$Value_{Adj} = Value_{Org} \times fx_{it} \times \frac{income_{i_t}}{income_{Ire_t}} \times CPI_{Ire_t} \quad (2)$$

(Eq. A.2)

where:

$Value_{Org}$ is the original value amount

$Value_{Adj}$ is the adjusted value amount

fx_{it} is the exchange rate for currency in country *i* during the year (*t*) of the original study

$income_{i_t}$ is the income for currency in country *i* during the year (*t*) of the original study values should be adjusted for purchasing power parity (PPP)

CPI_{Ire_t} is the adjustment for inflation from year of study in Ireland.

Exchange rate data, inflation figures and income data are available on the following sites:

Inflation figures:

http://www.cso.ie/en/statistics/prices/consumerpriceindex/

Exchange rates:

http://www.cso.ie/px/pxeirestat/Statire/SelectVarVal/Define.asp?Maintable=FIM02&Planguage=0

Income data:

http://data.worldbank.org/

http://epp.eurostat.ec.europa.eu/portal/page/portal/statistics/themes

Step 6. Apply relevant values to beneficiaries allowing for distance decay

Values are usually higher closer to the study site for both users and non-users. Hanley et al. (2003) also found that values for users decay faster than non-users. This distance decay can be modelled using a zonal method of analysis as employed by Hanley et al. (2003) and Bateman et al. (2006b).

Based on the results from Hanley et al. (2003), estimates of the ratio of users and non-users as a function of distance (distance measured by buffer zones) are given above in Table A5. The percentage change in value or WTP relative to the survey mean value or WTP for each zonal distance away from the environmental good is also estimated and presented in Table A5. These results show a rapid decrease both in the number of users and the WTP as the distance increases.

Using the distance decay factors (the percentages) in Table A5 in combination with the pre-defined economic jurisdiction based on Table A2, a number of buffer zones can be constructed around the water body being valued. For each buffer zone, *i*, the number of households can be obtained by summing the multiple of the area of the EDs within the buffer zone by the population density of each ED. This is more formally defined as Eq. A.3:

$$Users\_Value\_Bufferzone_i = \sum HH\_Den_{DEDi} \times \frac{Area_D - Ed_{Bi}}{Area\_Ed_i} \times Mean\_WTP_{HH} \times DD\_Adj_{UsersB_i}$$

$$Non\text{-}Users\_Value\_Bufferzone_i = \sum HH\_Den_{DEDi} \times \frac{Area\_Ed_{Bi}}{Area\_Ed_i} \times Mean\_WTP_{HH} \times DD\_Adj_{Non\_UsersB_i}$$

(Eq. A.3)

where:

$Users\_Value\_Bufferzone_i$ is the total value of a change in water status for users within bufferzone *i*

$Non\text{-}Users\_Value\_Bufferzone_i$ is the total value of a change in water status for non-users within bufferzone *i*

HH_Den is the density of households within a ED. This may include resident equivalent of foreign tourists

Area_ED is the area of a ED or portion of a ED

B_i is bufferzone *i*

$Mean\_WTP_{HH}$ is the mean willingness to pay per household per year

DD_Adj is the distance decay adjustment based on Table A5.

While one of the main reasons for using BT in the first instance is the lack of primary estimates for the policy sites, it is a worthwhile exercise at this stage in the process to test the transfer estimates derived against any existing estimates for any of the water bodies you are looking at. The transfer error for environmental good k (in this case GES value), is calculated as:

$$TransferError_k = \frac{(TransferredEstimate_k - PolicySiteEstimate_k)}{PolicySiteEstimate_k} \times 100$$

Of course, if no primary study has been conducted at any of the water policy sites the practitioners are interested in then it will not be possible to calculate this transfer error. In that case insuring that the study and policy sites used in the BT application are as close as possible in terms of physical and population characteristics will be even more relevant if policy makers are to have faith in the validity of the estimates derived.

Step 7. Aggregate the values to give a total non-market economic value on an annual basis

Having calculated the transfer error and assuming the practitioner accepts the resulting level of error, the final step in the unit transfer process involves aggregating the mean BT values per person across the relevant population. The values should be aggregated by adding together the values of the users and non-users in each of the buffer zones to give a total value resulting from the change in ecological status. This will give a value of the change in ecological status for the water body of interest per year. The value of the benefits could then be used within a CBA to establish if the benefits from a policy intervention aimed at achieving good ecological status in a water body would outweigh the costs.

Table A.6. Waters of national and regional importance – based on Fáilte Ireland Report on Determination of Waters of National Tourism Significance and Associated Water Quality Status (Fáilte Ireland, 2009).

Name of waterbody	County	Activities available
Lough Oughter	Cavan	Coarse fishing, watersports, events, walking, kayaking
Cliffs of Moher (waters around) and Liscannor Bay including Lahinch Beach	Clare	Sea fishing, boat trips, wildlife watching, extreme surfing, jet skiing
Mullaghmore Turlough (Burren National Park)	Clare	Nature study, landscape enjoyment.
Loop Head (waters around) and Kilkee Bay and Beach	Clare	Swimming, watersports, wildlife watching, sailing, surfing, diving.
Shannon Estuary	Clare, Limerick, Kerry	Sea fishing, boat trips, wildlife watching, swimming, sailing, watersports, cruising
Lough Derg	Clare, Tipperary, Galway	Coarse fishing, game fishing, sailing, watersports, cruising, walking, heritage, bird watching
Glandore Harbour and Union Hall	Cork	Sailing, watersports, festival, summer school, cruising, rowing
Kinsale Harbour	Cork	Sailing, cruising, sea fishing, diving, watersports
River Blackwater (Munster)	Cork	Coarse fishing, game fishing
Youghal Bay and Estuary	Cork	Sea fishing, swimming, maritime heritage, boat trips
Ballycotton Bay	Cork	Deep sea, shore and coastal fishing, walking. Watersports, swimming
Clonakilty Bay	Cork	Deep sea, shore and coastal fishing
Courtmacsherry Bay	Cork	Sea fishing, shore fishing, kayaking, dinghy sailing, adventure activities
Bandon river and estuary	Cork	Sea fishing, shore fishing, game fishing
Barleycove	Cork	Surfing, sea fishing, swimming
Cork Harbour	Cork	Sea fishing, sailing, watersports, boat trips, events, cruise ships, ferries
Beara Peninsula (waters around)	Cork	Sea fishing, diving, watersports, kayaking
Bantry Bay	Cork	Sea fishing, coastal cruising, watersports, diving, shore fishing
Baltimore Harbour, Clear and Sherkin Island, Schull and Roaringwater Bay	Cork	Deep sea fishing, shore fishing, watersports, cruising, walking, island visits, diving
River Bandon	Cork	Game fishing

Name of waterbody	County	Activities available
Lough Hyne and Tragumna Bay	Cork	Canoeing/kayaking
Sheephaven Bay	Donegal	Sea fishing, watersports, swimming, jetskiing, shore angling, kite sports
Arainn Mor (waters around) and Burtonport	Donegal	Sea fishing, island trip, watersports, cruising, sailing, shore fishing
Toraigh (Tory Island) (waters around) inc Magheroarty, Inisbofinne	Donegal	Island ferry, swimming, surfing, diving, walking, bird watching
Lough Swilly	Donegal	Sea fishing, sailing, shore fishing, watersports, swimming
Culdaff Bay	Donegal	Sea fishing, shore fishing, swimming, watersports
Lough Beagh (Glenveagh National Park)	Donegal	Game fishing, walking, wildlife watching
Lough Foyle	Donegal	Walking, sailing. Sea fishing, watersports, kayaking, maritime museum visits, ferry
Donegal Bay	Donegal, Sligo	Sea fishing, watersports, diving, surfing, boat trips, beach horse riding, shore fishing
Dublin Bay	Dublin	Sailing, cruising, diving, watersports
River Liffey (tidal section)	Dublin	Cruise ships, events, sailing, trip boats, ferry
Malahide Estuary	Dublin	Swimming, marina, sailing club, waterside walk, sea fishing
Dun Loaghaire, Scotsmans Bay and Seapoint	Dublin	Sailing, cruising, watersports
Portmarnock, Howth, Carrigeen Bay	Dublin	Swimming, walking, sailing, events, kayak, kitesports
River Liffey	Dublin, Kildare, Wicklow	Game fishing, canoeing, events, walking
Aran Islands and waters around Ceathra Rua	Galway	Sea fishing, diving, swimming, scenic touring, walking
Lough Corrib	Galway	Game fishing
Costello-Fermoyle (Casla)	Galway	Game fishing
Ballynahinch Fishery inc Derryclare and Inagh Lakes	Galway	Game fishing
Cleggan –Inisbofin	Galway	Sea fishing, island visits, cruising
Clifden Harbour	Galway	Kayaking, sailing, sea fishing, waterside walking, scenic touring
Galway Bay	Galway, Clare	Scenic touring, swimming, surfing, watersports, coastal cruising,
Dingle Bay	Kerry	Sea fishing, coastal cruising, diving, sailing, nature tourism
Lakes Of Killarney	Kerry	Scenic viewing, inland cruising, game fishing, canoeing, kayaking
Dingle Peninsula (waters around) Blasket Sound and the Blasket Islands	Kerry	Sea fishing, swimming, whale and dolphin watching, sailing, watersports, walking,
Portmagee, Skelligs and Valentia Island (waters around)	Kerry	Boat trips, sea fishing, sea kayaking, sailing, cruising
Kenmare Bay/River and Derrynane Bay	Kerry	Sea fishing, coastal cruising, sailing, watersports, diving, canoeing/kayaking
Tralee Bay	Kerry	Sea fishing, swimming, kitesports, shore fishing, riding, walking
Lough Currane	Kerry	Game fishing, waterside walking, waterway heritage visits
Cashen/Feale	Kerry	Game fishing, festivals, walking
Doulus Bay, Valentia Harbour and Valentia River	Kerry	Deep sea fishing, coastal fishing, sailing, watersports, cruising
Ballinskelligs Bay	Kerry	Shore fishing
Caragh Lake	Kerry	Game fishing, canoeing, sailing, wind surfing
River Laune and Flesk	Kerry	Game fishing, kayaking
River Barrow and Barrow	Kildare, Carlow, Kilkenny, Laois	Navigation
The Grand Canal	Laois, Offaly, Kildare, Dublin	Cruising, boat trips, coarse fishing, cycling, walking, canoeing

Name of waterbody	County	Activities available
Shannon Erne Waterway	Leitrim,Cavan	Coarse fishing, cruising, canoeing, walking
River Drowes	Leitrim, Donegal	Game fishing, canoeing
Lough Allen	Leitrim. Sligo, Roscommon	Cruising, watersports, events, fishing, walking
Lough Melvin	Leitrim/ Cavan	Game fishing, watersports, canoeing, sailing, events, boat hire
River Shannon (Lower-Killaloe to Limerick)	Limerick	Cruising, coarse angling, walking, kayaking
River Shannon (Upper – Carrick on Shannon to Lanesborough)	Longford, Roscommon	Cruising, coarse fishing, scenic, water activities
Lough Gowna	Longford, Cavan	Coarse fishing, kayaking, jet skiing
Lough Ree	Longford, Roscommon, Westmeath	Coarse fishing, cruising, watersports, sailing, walking, wildlife watching
River Inny	Longford, Westmeath	Coarse fishing, kayaking
The Royal Canal	Longford, Westmeath, Kildare, Meath, Dublin	Cruising, coarse fishing, water activities, walking, events
Boyne and tributaries	Louth, Meath, Kildare	Watersports/boat hire/river cruise/game fishing/scenic/heritage
Carlingford Lough	Louth/NI	Walking, sea fishing, sailing, watersports, scenic
River Moy	Mayo	Game fishing ‘salmon capital of ireland’
Clew Bay	Mayo	Sea fishing, sea trips, yachting, surfing
Loughs Mask and Carra	Mayo	Game fishing, scenic touring, Gaeltacht area
Loughs Conn and Cuillin	Mayo	Game fishing, walking
Achill Island (waters around)	Mayo	Sea fishing, sea trips, yachting, surfing
Delphi Fishery	Mayo	Game fishing, scenic route
Blacksod Bay	Mayo	Sea fishing
Broadhaven Bay	Mayo	Sea fishing
Killary Harbour	Mayo,Galway	Scenic touring, adventure/walking, sea fishing
Roonah, Inishturk, Clare Island (waters around)	Mayo/Galway	Sea fishing, sea trips, yachting, surfing
Killala Bay and the Moy Estuary	Mayo/Sligo	Sea fishing, swimming, watersports, kite sports, shore fishing, game fishing, thallasotherapy
Lough Sheelin	Meath, Westmeath, Cavan	Game fishing
Lough Muckno	Monaghan	Coarse fishing, canoeing, watersports
Lough Boora Lakes	Offaly	Coarse & pike fishing, wildlife watching, nature study, walking
Lough Key	Roscommon	Inland cruising, walking, scenic touring, swimming, kayaking, watersports, coarse fishing
River Suck & Tributaries	Roscommon, Galway	Mixed fishing, inland cruising, walking,
Rosses Point	Sligo	Swimming, sailing, cruising, sea fishing
Lough Gill	Sligo/Leitrim	Game fishing
River Suir	Tipperary	Game fishing, rowing, walking route, canoeing
River Nore	Tipperary, Laois, Kilkenny	Game fishing
Dungarvan Harbour	Waterford	Sea fishing, sailing, water sports
Tramore Beach	Waterford	Sea fishing, water sports,
Copper Coast (waters off)	Waterford	Scenic, visitor attraction/heritage
Blackwater River	Waterford	Game fishing, river trips, water sports

Name of waterbody	County	Activities available
Ardmore	Waterford	Sea fishing
Waterford Harbour and the tidal sections of the Suir and Barrow	Waterford, Wexford	Cruise ship visits, water sports, tall ships race
Lough Ennell	Westmeath	Coarse fishing, game fishing, watersports, family fun
Lough Owel	Westmeath	Fishing, scenic and water based activities
River Shannon (Mid)	Westmeath, Offaly	Coarse fishing, cruising, sailing, watersports, bird watching, heritage
Kilmore Quay and Saltee Islands	Wexford	Deep sea fishing, shore fishing, coastal fishing, water sports, scenic, bird watching, coastal cruising, island visit, boat trips
Courtown Harbour and Beaches	Wexford	Sailing, sea fishing, water sports
Hook Beaches	Wexford	Sea fishing, water sports,
Wexford Harbour, Wexford Bay and Rosslare Bay	Wexford	Sailing, sea fishing
Glendalough	Wicklow	Scenic, walking, historic
Blessington Lakes	Wicklow	Fishing, watersports, wildlife watching
Brittas Bay	Wicklow	Scenic, beach, water activities

Appendix B. Results of the Choice Experimental Model from Stithou et al. (2011)

Table B1. Definition of variables included in Stithou et al. (2011) choice experiment (CE) model.

Variable name	Description
River Life _G	River life (fish, insects, plants): good relative to poor
River Life _M	River life (fish, insects, plants): Moderate relative to poor
Appearance _A	Water appearance: a lot of improvement
Appearance _S	Water appearance: some improvement
Recreation _A	Recreational activities: walking, boating, fishing, swimming
Recreation _S	Recreational activities: walking, boating, fishing
River Banks	Condition of River Banks: Natural looking banks relative to Visible erosion that needs repairs
Cost	Household's annual tax payments for the next 10 years (€/year)
SQ	*Status quo* (No change alternative)
Age	Respondent's age scale 1 to 6, where 1=15 to 17 and 6=over 65
Hdegree	1 if education is higher than secondary school, 0 otherwise
Depnt	Number of dependants in the household
Fullempl	1 if respondent is full-time employed, 0 otherwise
Middlecl	1 if chief income earner belongs to middle class, 0 otherwise
NoIncome	1 if respondent reported her income, 0 otherwise
Waterpolicy	1 if respondent is aware of any specific water related policy taking place in Ireland at the moment or in the past, 0 otherwise
Nsconserned	1 if respondent is not sure thinking of him/herself as being concerned about the environment, 0 otherwise
Unsatisfqual	1 if respondent describes river's general environmental quality (water & surroundings) unsatisfactory, 0 otherwise
Instinct	1 if respondent chose by only following her instinct, 0 otherwise
Socialcon	1 if respondent chose according to what family/friends would expect/like her to choose, 0 otherwise
Cognitive	Total score of cognitive ability, measured on a 1 to 7 Likert scale, according to perceived degree of difficulty concentrating on the task, remembering the necessary information, thinking clearly and logically and choosing the best option. The smaller the score the higher the degree of difficulty.
Dist1km	1 if distance of respondent's townland is less than 1 km from closest tributary, 0 otherwise

Table B2. Stithou et al. (2011) choice experiment (CE) model results.

	Model results	
	est.	*t*-ratio
River Life _G	1.180	(2.890)***
River Life _M	1.754	(4.743)***
Appearance _A	1.649	(3.627)***
Appearance _S	0.671	(1.801)*
Recreation _A	1.000	(2.263)**
Recreation _S	0.250	(0.793)
River Banks	1.292	(3.518)***
Cost	-0.044	(-4.932)***
SQ	2.315	(1.177)

Continued over

Table B2. Stithou et al. (2011) choice experiment (CE) model results. *cont.*

	Model results	
	est.	***t*-ratio**
AgeSQ	0.070	(0.327)
HdegreeSQ	0.930	(1.523)
DepntSQ	-0.409	(-1.723)*
FullemplSQ	-1.699	(-2.900)***
MiddleclSQ	-1.438	(-2.581)**
NoIncomeSQ	1.526	(1.693)*
Dist1kmSQ	-2.355	(-3.295)***
WaterpolicySQ	-2.218	(-2.192)**
NsconsernedSQ	3.734	(2.733)***
UnsatisfqualSQ	-2.041	(-3.129)***
InstinctSQ	1.489	(2.514)**
SocialconSQ	1.922	(3. 140)***
CognitiveSQ	-0.128	(-2.367)**
River Life _G	1.580	(1.737)*
River Life _M	0.986	(2.115)**
Appearance _A	1.606	(3.469)***
Appearance _S	2.183	(4.148)***
Recreation _A	1.658	(2.553)**
Recreation _S	1.222	(2.563)***
River Banks	2.679	(5.068)***
LL	-577.386	
χ^2	638.163	
p^2	0.35	
BIC	654.498	
Correctly predicted	52%	
Observations	816	
# of respondents	204	

(*) indicates significant at 10%; (**) indicates significant at 5%; (***) indicates significant at 1%.

An Ghníomhaireacht um Chaomhnú Comhshaoil

Is í an Gníomhaireacht um Chaomhnú Comhshaoil (EPA) comhlachta reachtúil a chosnaíonn an comhshaol do mhuintir na tíre go léir. Rialaímid agus déanaimid maoirsiú ar ghníomhaíochtaí a d'fhéadfadh truailliú a chruthú murach sin. Cinntímid go bhfuil eolas cruinn ann ar threochtaí comhshaoil ionas go nglactar aon chéim is gá. Is iad na príomh-nithe a bhfuilimid gníomhach leo ná comhshaol na hÉireann a chosaint agus cinntiú go bhfuil forbairt inbhuanaithe.

Is comhlacht poiblí neamhspleách í an Ghníomhaireacht um Chaomhnú Comhshaoil (EPA) a bunaíodh i mí Iúil 1993 faoin Acht fán nGníomhaireacht um Chaomhnú Comhshaoil 1992. Ó thaobh an Rialtais, is í an Roinn Comhshaoil, Pobal agus Rialtais Áitiúil.

ÁR bhFREAGRACHTAÍ

CEADÚNÚ

Bíonn ceadúnais á n-eisiúint againn i gcomhair na nithe seo a leanas chun a chinntiú nach mbíonn astuithe uathu ag cur sláinte an phobail ná an comhshaol i mbaol:

- áiseanna dramhaíola (m.sh., líonadh talún, loisceoirí, stáisiúin aistrithe dramhaíola);
- gníomhaíochtaí tionsclaíocha ar scála mór (m.sh., déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta);
- diantalmhaíocht;
- úsáid faoi shrian agus scaoileadh smachtaithe Orgánach Géinathraithe (GMO);
- mór-áiseanna stórais peitreail;
- scardadh dramhuisce.

FEIDHMIÚ COMHSHAOIL NÁISIÚNTA

- Stiúradh os cionn 2,000 iniúchadh agus cigireacht de áiseanna a fuair ceadúnas ón nGníomhaireacht gach bliain.
- Maoirsiú freagrachtaí cosanta comhshaoil údarás áitiúla thar sé earnáil - aer, fuaim, dramhaíl, dramhuisce agus caighdeán uisce.
- Obair le húdaráis áitiúla agus leis na Gardaí chun stop a chur le gníomhaíocht mhídhleathach dramhaíola trí comhordú a dhéanamh ar líonra forfheidhmithe náisiúnta, díriú isteach ar chiontóirí, stiúradh fiosrúcháin agus maoirsiú leigheas na bhfadhbanna.
- An dlí a chur orthu siúd a bhriseann dlí comhshaoil agus a dhéanann dochar don chomhshaol mar thoradh ar a ngníomhaíochtaí.

MONATÓIREACHT, ANAILÍS AGUS TUAIRISCIÚ AR AN GCOMHSHAOL

- Monatóireacht ar chaighdeán aeir agus caighdeáin aibhneacha, locha, uiscí taoide agus uiscí talaimh; leibhéil agus sruth aibhneacha a thomhas.
- Tuairisciú neamhspleách chun cabhrú le rialtais náisiúnta agus áitiúla cinntí a dhéanamh.

RIALÚ ASTUITHE GÁIS CEAPTHA TEASA NA HÉIREANN

- Cainníochtú astuithe gáis ceaptha teasa na hÉireann i gcomhthéacs ár dtiomantas Kyoto.
- Cur i bhfeidhm na Treorach um Thrádáil Astuithe, a bhfuil baint aige le hos cionn 100 cuideachta atá ina mór-ghineadóirí dé-ocsaíd charbóin in Éirinn.

TAIGHDE AGUS FORBAIRT COMHSHAOIL

- Taighde ar shaincheisteanna comhshaoil a chomhordú (cosúil le caighdéan aeir agus uisce, athrú aeráide, bithéagsúlacht, teicneolaíochtaí comhshaoil).

MEASÚNÚ STRAITÉISEACH COMHSHAOIL

- Ag déanamh measúnú ar thionchar phleananna agus chláracha ar chomhshaol na hÉireann (cosúil le pleananna bainistíochta dramhaíola agus forbartha).

PLEANÁIL, OIDEACHAS AGUS TREOIR CHOMHSHAOIL

- Treoir a thabhairt don phobal agus do thionscal ar cheisteanna comhshaoil éagsúla (m.sh., iarratais ar cheadúnais, seachaint dramhaíola agus rialacháin chomhshaoil).
- Eolas níos fearr ar an gcomhshaol a scaipeadh (trí cláracha teilifíse comhshaoil agus pacáistí acmhainne do bhunscoileanna agus do mheánscoileanna).

BAINISTÍOCHT DRAMHAÍOLA FHORGHNÍOMHACH

- Cur chun cinn seachaint agus laghdú dramhaíola trí chomhordú An Chláir Náisiúnta um Chosc Dramhaíola, lena n-áirítear cur i bhfeidhm na dTionscnamh Freagrachta Táirgeoirí.
- Cur i bhfeidhm Rialachán ar nós na treoracha maidir le Trealamh Leictreach agus Leictreonach Caite agus le Srianadh Substaintí Guaiseacha agus substaintí a dhéanann ídiú ar an gcrios ózóin.
- Plean Náisiúnta Bainistíochta um Dramhaíl Ghuaiseach a fhorbairt chun dramhaíl ghuaiseach a sheachaint agus a bhainistiú.

STRUCHTÚR NA GNÍOMHAIREACHTA

Bunaíodh an Ghníomhaireacht i 1993 chun comhshaol na hÉireann a chosaint. Tá an eagraíocht á bhainistiú ag Bord lánaimseartha, ar a bhfuil Príomhstiúrthóir agus ceithre Stiúrthóir.

Tá obair na Gníomhaireachta ar siúl trí ceithre Oifig:

- An Oifig Aeráide, Ceadúnaithe agus Úsáide Acmhainní
- An Oifig um Fhorfheidhmiúchán Comhshaoil
- An Oifig um Measúnacht Comhshaoil
- An Oifig Cumarsáide agus Seirbhísí Corparáide

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag ball air agus tagann siad le chéile cúpla uair in aghaidh na bliana le plé a dhéanamh ar cheisteanna ar ábhar imní iad agus le comhairle a thabhairt don Bhord.

Science, Technology, Research and Innovation for the Environment (STRIVE) 2007-2013

The Science, Technology, Research and Innovation for the Environment (STRIVE) programme covers the period 2007 to 2013.

The programme comprises three key measures: Sustainable Development, Cleaner Production and Environmental Technologies, and A Healthy Environment; together with two supporting measures: EPA Environmental Research Centre (ERC) and Capacity & Capability Building. The seven principal thematic areas for the programme are Climate Change; Waste, Resource Management and Chemicals; Water Quality and the Aquatic Environment; Air Quality, Atmospheric Deposition and Noise; Impacts on Biodiversity; Soils and Land-use; and Socio-economic Considerations. In addition, other emerging issues will be addressed as the need arises.

The funding for the programme (approximately €100 million) comes from the Environmental Research Sub-Programme of the National Development Plan (NDP), the Inter-Departmental Committee for the Strategy for Science, Technology and Innovation (IDC-SSTI); and EPA core funding and co-funding by economic sectors.

The EPA has a statutory role to co-ordinate environmental research in Ireland and is organising and administering the STRIVE programme on behalf of the Department of the Environment, Heritage and Local Government.

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