Title:

Estimating the social benefit of environmental damage remediation based on value estimates for ecosystem services.

Authors: Craig Bullock & Robert O'Shea

In Press: Journal of Environmental Planning and Management 2016

Abstract

The European Environmental Liability Directive aims to ensure that damaged habitats are restored where possible, but allows for complementary remediation with replacement habitat where restoration is not possible or cannot be achieved within a reasonable time. It also allows for compensatory remediation of the resource based on an assessment of environmental values in cases where there are interim social losses. This paper concurs with the argument that physical restoration or remediation without consideration of social values can fail to be equivalent to the resource that has been lost. Using as a case study a river in Ireland, it demonstrates that estimating social value can be challenging in practice, noting also differences between the value of environmental gains and losses. The paper argues that estimates of final ecosystem service values, including wastewater treatment costs, can provide a measure of social value and makes a case for the collection of this data to inform decision making.

Keywords

Environmental liability

Remediation

Social values

Ecosystem services

Wastewater

1. Introduction

1.1 Environmental liability

The use of liability assessment following instances of physical damage or pollution of environmental resources has long been a feature of national legislation. For example, the US Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) has provided for the clean-up of hazardous waste sites since 1980 and requires resource damage assessment for this and similar instances of environmental injury. In Europe, the Environmental Liability Directive (ELD 2004/35/EC) now applies a common approach to assessment which aims to prevent and remedy environmental damage by holding those responsible liable for remediation.[1] However, while there are prescribed procedures for remediation, there remains the difficulty of how to achieve an equivalent level of habitat quality to that which existed before an incident and how to account for interim losses, including losses to social wellbeing.

1.2 Types of remediation

The ELD applies to the contamination of land, damage to water bodies, and damage to species and natural habitats protected by the EU Habitats Directive (92/43/EEC). To qualify as a significant risk, damage to land must present a threat to human health.[2] Damage to water must be such as to affect the quality status as defined by the Water Framework Directive (2000/60/EC) based on biological and chemical criteria relevant to ecosystems and health. Damage to protected species or natural habitats must be sufficient to undermine their "favourable conservation status". In this last respect, the ELD has a potentially wider scope than the Habitats Directive in that it addresses protected species and habitats both within and

[1] Strictly, the term remediation refers to the removal of a problem that is causing environmental damage. Recovery refers to the return to an original level of habitat or habitat services.
[2] European Community (2008) Environmental Liability Regulations

outside of designated Natura 2000 sites, for example where water pollution impacts on unidentified numbers of species.[3]

Damage as defined by the ELD presupposes that liability can be identified. Where this is possible, the ELD allows for three types of remediation:

- Primary remediation to restore a damaged resource or impaired service to its baseline condition.
- Complementary remediation when a site can not be fully restored using primary remediation and which involves intervention or improvements to habitat at another site which is physically or geographically linked in terms of species/habitats or human interactions.
- Compensatory remediation in cases where there are interim losses before ecological functions can be fully restored or replaced.

In the case of complementary and compensatory remediation, there is a need to provide *equivalency*, namely to scale remediation to compensate sufficiently for the original loss. In practice, the acceptability of measures has been informed by EU Member States' experience of implementing the Habitats Directive, often following legal debate on specific cases. However, while the Habitats Directive only tolerates environmental impacts in cases of imperative reasons of overriding public interest (IROPI), a distinction of the ELD is that it acknowledges the importance of both environmental functions and services to human beings. It notes that where restoration of services is not possible "alternative valuation techniques shall be used" to ensure that remediation is equivalent to the resource which has been lost. Potentially, this can include economic or monetary valuation methods to inform the level of remediation.

[3] Natura 2000 is a network of protected sites across Europe which was established under the Habitats Directive.

Of course, many impacts on environmental quality are occurring due to on-going loss of minor non-designated habitat, e.g. ponds or hedgerows, or because of diffuse pollution of unknown origin. As well as liability issues, measures of equivalency are also relevant to biodiversity offsets. Offsets are not confined to the type of damage that would be addressed by the ELD, but similarly seek to replace environmental losses with equivalent gains. Indeed, it has been argued that offsets have the potential to deliver new habitats additional to those protected by designation (Crowe and ten Kate, 2010; McManus and Duggan, 2011).

The purpose of this paper is to examine those occasions when it could be useful to apply economic valuation to estimate the social benefits and costs of remediation. The paper discusses the practicalities of doing so and some of the factors that would need to be considered. **Section 2** introduces the notion of providing equivalency of habitat and the inadequacy of like-for-like replacement of habitats without consideration of social values. **Section 3** discusses the practical difficulties of providing for equivalency, including of social values. **Section 4** discusses these issues in the context of a case study of the River Suir in Ireland and the information available. The paper closes with an assessment of the practicality of accounting for social values.

2. Equivalency

2.1 Resource equivalency

Primary remediation is often challenging. A case emerges for complementary or compensatory remediation because of the difficulty of restoring ecological functions within a meaningful time-frame. Equivalency measures become appropriate where the nature, scale or location of remediation differs from that where losses were incurred (Lipton et al., 2008). In this case, the ELD favours a hierarchy of resource-to-resource, service-to-service and value-to-value approaches (EPA, 2011a). The first can be achieved using resource equivalency analysis (REA) expressed in physical units such as bird or animal species and vegetation. A service-to-service approach seeks to provide equivalency in terms of ecological functions and direct-to-human services and is described under a heading of habitat equivalency analysis (HEA).[4] A value-to-value approach converts services into a monetary metric, although this is intended to inform biophysical remediation rather than to provide financial compensation except in cases where in-kind measures are not feasible.

Criteria exist to guide the provision of replacement habitat, e.g. REMEDE[5] in the EU and NOAA (2000, 2006) in the US. However, while HEA may be appropriate if two habitats are of a similar type or quality to permit a scaling of losses to gains, value-to-value approaches become useful where there are differences in service flows, particularly in the interim period until remediation is achieved.

2.2 Social values of remediation

Following the recommendations of the NOAA Panel in 1995 on the Exxon Valdez oil pollution incident in Alaska[6], willingness-to-pay values were accepted as one basis by which to assess environmental impacts (subject to stringent conditions). Revealed preference methods assume that if people voluntarily use a resource, then the value they attach to the resource is captured in terms of their willingness-to-pay the costs incurred. Stated preference methods employ surveys to ask people directly how much they are willing-to-pay using methodologies such as contingent valuation or choice experiments. In principle, stated preference can capture a higher representation of people's maximum willingness-to-pay, including for goods for which there is only passive use.

In practice, the challenges of estimating and agreeing values meant that compensatory restoration based on resource units or services has become the dominant approach (Dunford et

[4] REA is often used as a generic term to include HEA.
[5] REMEDE www.envliability.eu (Lipton et al, 2008).
[6] The National Oceanic and Atmospheric Administration convened a panel of economists (chaired by Nobel laureates Kenneth Arrow and Robert Solow).

al., 2004; Zafonte and Hampton, 2007). The arrangement can facilitate agreement between polluter and agency by confining the negotiation to biological resources (Dunford et al, *ibid*). A simple like-for-like equivalency may be sufficient where the social value of the services provided by an individual site are modest or for small, viable habitats whose ecological value is encapsulated in the national extent and condition of the habitat type to which they belong (Hill et al., 2005). Likewise, this approach may suffice if it is assumed that people derive utility from an environment in proportion to the supply of these services (Roach and Wade, 2006).

The use of HEA predates the burgeoning interest in ecosystem services that followed the Millennium Ecosystem Assessment (MEA, 2005) which defined the supporting, regulating, provisioning and cultural services that are provided by ecosystems and which supply a wide range of benefits to human beings.[7] The cost of replacing these services could be greater (or less) than the cost of a like-for-like restoration or replacement of habitat. Although one particular habitat may provide limited ecosystem services, another could be a significant element in a network of habitats or be of substantial importance in itself and responsible for numerous services of value to people. In these situations, it could be very difficult for complementary remediation based on habitat or species alone to restore the same level of services to that which existed before the site was damaged. If it thought that a habitat does have a significant social value, then remediation should be scaled to compensate society with sufficient physical units of a resource to ensure that the aggregate change in social wellbeing is zero (Jones and Pease, 2007).[8] Unless the social values of ecosystem services are explicitly considered, there is the danger that they will be ignored in just the same way that the ecosystems themselves were prior to liability legislation. Indeed, Cowling et al (2008) find that many biophysical assessments have given little regard to social aspects.

[7] The principles of the service-to-service approach were first set out by King & Adler (1991) and Unsworth & Bishop (1994).

[8] In economic terms, that the sum of individuals' (i) compensating variation (CV) is zero ($\Sigma_i CV_i = 0$). In this situation, the winners are potentially able to compensate the losers according to the Kalder-Hicks criterion.

In addition, Flores and Thacher (2002) argue that individuals are likely to hold heterogeneous, or varying, social values for both the damaged and replacement habitat. For example, even if a proposed replacement habitat is geographically linked into the same natural system as the habitat that was lost, it could be located somewhere that is distant or unfamiliar to some people. In such a case the replacement habitat would not provide the same range of socio-cultural values as the established site. Cultural values could also be associated with the original site's landscape setting or sense of place. Zafonte and Hampton (2007) describe a 'market effect' in which a site will be of higher value to local communities than to people living further away who might rather value the site in terms of its contribution to the total stock of the habitat. If the size of the local population relative to the non-local population is large, then standard physical remediation represented by HEA could under-compensate.

Furthermore, as it takes time for a habitat to recover or for a replacement habitat to be provided, standard HEA does allow for the loss of services in the interim period by prescribing criteria for the discounting of gains from this new habitat. This is typically expressed in terms of the acre years requiring that a larger spatial extent of replacement habitat compensates for the losses. However, even using a HEA approach, Strange et al (2002) demonstrate how the inclusion of different ecological functions can lead to markedly different amounts of compensatory area. Furthermore, the social benefits of the original habitat, for example, shellfish beds, may depend on complex supporting and regulating services that could take a very long time to restore. Even then it might not be possible to achieve a population that permits a sustainable harvest. Dunford et al (2004) also describe how new services might emerge during the process of restoration. Indeed, this could occur naturally as the new habitat develops, or from the necessities and sequencing of engineering work. For instance, a newly created water body would attract a succession of wildlife and flora as it evolves over time. This means that social values would change too.

Shaw and Wlodarz (2013) remark that it is difficult to define an appropriate discount rate to address social values given how these can change over time and vary between individuals. A positive discount rate implies that society values services today more than services in the future. However, empirical studies have shown that, not only does the rate of time preference vary between people, but it also tends to decline over time (Gollier, 2002; Weitzmann, 1998). Consequently, Shaw and Wlodarz postulate that long-term restoration projects may deserve lower discount rates than shorter ones. On the other hand, if people perceive that the risk of unsuccessful restoration is high, then this would be an argument for a higher discount rate.

3. Practical constraints on equivalency

3.1 Ecological knowledge

Boyd (2010) observes that successful equivalency involves a need to know, firstly how natural systems provide valuable goods, and secondly, what the value of these goods are. The first requires a biophysical assessment. A common problem in this regard is the absence of baseline data (Natura, 2005). This data is usually available at one level or another for protected sites, but is often patchy or non-existent for the wider environment. There may be evidence of the presence of certain well-known bird or animal species, but there is typically very little information on natural variations in species abundance, local ecological relationships, or predator-prey relationships.

Furthermore, too often we understand very little about the dynamics of the ecosystem (Maler et al., 2009). These include the processes that allow for ecological functions, how these functions provide valuable ecosystem services, and how these change in response to altered environmental conditions. Much ecological research has been conducted at specific locations or within narrow bands that might not guarantee external validity (Moore and

Robinson, 2004). Research is often undertaken in laboratories because of the need to control for the wide variety of factors that occur in wild conditions. A laboratory can be used to identify the risk thresholds associated with hazardous chemicals (Gala et al, 2009), but dose-response relationships have often not been demonstrated for marginal changes in natural ecosystem processes. The same effects will not necessarily reoccur at new locations due to differences in context, the assimilative capacity of the environment, or the varying complexity of natural systems. This includes the mix of species present and the role of keystone species versus redundancy (Mooney, 2002; Naeem et al., 2002). For example, for aquatic ecosystem services it can be hard to identify the impact of a particular river or wetland on water quality relative to terrestrial catchment management. Similarly, there will be physical, as well as biological, influences such as the role of weirs in introducing oxygen or of sunlight in killing harmful bacteria (Gray, 2004).

Additional challenges arise from temporal or spatial factors. Change can take a long time to work its way between species and species function (trophic) levels until the point at which it is revealed as an impact. Ecological damage can take years to play out with impacts on species higher up the food chain being realised long after a critical threshold has been surpassed or only after gradual reductions in a species' population (Findlay and Bourdages, 2000). Temporal and spatial characteristics combine to present a familiar problem for water pollution where the agent responsible for an impact can be very difficult to identify after a period of time has passed (Liu et al., 2005). Impacts may be realised far from where the original damage occurred with the problem of attribution being exacerbated by the mobility of species, including mobility at various stages in their life cycles. The classic example is of a fish population where an impact may occur in a spawning territory, but be realised in terms of adult fish or catches far downstream. The location of the spawning territory may be known, but the range of conditions that make for good fish spawning, survival and growth is typically poorly understood and subject to many factors (Harriman et al., 1990).

There are challenges too in estimating social values. In principle, the concept of ecosystem services does provide a useful means to communicate the effect of ecological impacts in terms of social wellbeing. Choice experiments can be used in which survey respondents trade-off a monetary value in return for the value of varying provision of attributes of a resource, for example different ecosystem services. Ideally, choice experiments should be supported by the availability of incremental data that describe the relationship between habitat losses and ecosystem services. If, though, there is no scientific evidence with which to demonstrate the incremental ecological change, it may still be possible to present change in terms of impacts on final goods or services (Boyd, 2010). The CICES classification of ecosystem services (Haines-Young and Potschin, 2013) does indeed focus on final services as a means to avoid double counting for environmental accounting purposes. As the public are understandably unfamiliar with many intermediate services, it is practical to identify changes in more familiar final provisioning and cultural ecosystem services while acknowledging the uncertain ecological causal links. Robust estimates are most likely when respondents can independently comprehend the link between the levels of environmental change and final ecosystem services such as the impact on valued wildlife species or recreational activities.

Nevertheless, evidence of the value of even final services can sometimes to be difficult to identify. For example, the benefits that a wetland provides in moderating run-off can be calculated in terms of the costs avoided due to downstream erosion or flooding. However, the cost of physical damage to property is difficult to estimate given the unique set of human factors and natural factors which influence the level of damage in each flood event.

3.2 The Reference Point

HEA assumes that society values damaged and restored resources equally. This assumption of perfect substitutability is highly unlikely (Flores and Thacher, 2002; Shaw and Wlodarz, 2013; Zafonte and Hampton, 2007). For example, Zafonte and Hampton (2007) identify a link with the amount of undamaged habitat. On the basis of historical trends and supply and demand, they argue that the marginal value of remaining habitat typically increases as the national area or quality of this habitat declines. The amount of compensation would then depend, not just on the recovery time, but the respective change in, or elasticity, of demand for the habitat.

Revealed preference can be used to estimate social value if people are aware of the change. Alternatively, stated preference methods can be used if people are informed of the implications of environmental change that could go unnoticed in terms of behaviour. For example, people might continue to swim in an increasingly polluted river until such time as warning notices are erected. Stated preference also has an edge in hypothetical scenarios where a threshold has not yet been crossed.

An accurate valuation of losses and gains is relevant to damage assessment and to the ELD which acts primarily as a deterrent to behaviour that might lead to environmental *losses*. Most stated preference surveys elicit willingness-to-pay estimates for a positive change or to prevent an adverse change. In principle, a willingness-to-accept format is more appropriate to estimate the value of losses and should differ from estimates of willingness-to-pay only to the extent of an income effect due to the different starting points. Hanemann (1991) judged this effect to be small for goods for which there are substitutes. However, there is a good deal of empirical evidence from psychology to show that people value losses more than they value gains (Kahneman and Tversky, 1979; Kahneman and Tversky, 1984). Loss aversion can arise in relation to a reference point that is often the status-quo supply of a resource.[9] It may also apply to a situation where legislation creates an entitlement to the protection of environment quality (as with the ELD) and to expectations of future outcomes or an ideal scenario (Koszegi and Rabin, 2006).

[9] See Samuelson & Zeckhauser, 1988; Kretsch & Wong, 2009 for further discussion of status-quo bias, endowment effects and reference points.

An individual's perception of the likelihood of gains or losses can depend on how close they perceive the environment to be to a threshold. It may happen that high social values of change are associated with particular thresholds at which people become aware of the change or to which they are especially attached (Schumacher and Zou, 2013). These could include tipping points at which natural systems change from one familiar state to another in response to stressors overcoming ecological resilience (Parsons et al., 2009). If the new state is less desirable, this could be because certain types of recreational activity, for example direct contact sports such as swimming, are no longer safe. Similarly, wildlife species of passive use value may no longer be present.

In the context of equivalency, complementary remediation may be valued less than the original loss. For instance, in a contingent valuation survey following damage caused to the Donada wetland in Spain by the catastrophic failure of an upstream tailing pond, Martin-Ortega et al (2011) found that respondents were willing to pay less to maintain complementary habitat than to avoid a reoccurrence of the original damage, at €4.75 and €9.59 per household per year respectively. This finding could indicate that respondents valued losses differently to gains. Although Martin-Ortega et al did find that frequent visitors to the new habitat were willing to pay a figure closer to the average value of the loss, it does highlight the benefit of using remediation to achieve more than a ratio of one-to-one replacement or *no net loss*. It suggests a need to provide a distinct environmental gain to account for the relatively higher value of losses as well as the likelihood of heterogeneous preferences and the presence of uncertainty. The principle of no net less is well-established in biodiversity offsets and has the potential to provide greater assurance that the aggregate stock of natural capital will be maintained (EC, 2007).[10] Noting also that many habitats are already degraded to one extent or another, it also offers opportunities to use remediation to strengthen the functionality of the natural environment (Helm, 2014).[11]

[10] See also the ten best practice principles of the Business and Biodiversity Offsets Programme www.bbop/forest-trends.org

[11] For example, the use of offsets to provide improved compensatory habitat or physical connectivity.

4. Case Study

4.1 The River Suir, Ireland

Every year around a dozen businesses in Ireland, mostly pig and poultry farms, are prosecuted for pollution related offences, but there have been no major incidents in recent years.[12] However, while we are not able to demonstrate the consequences of a significant recent incident, we can look at the information that would be available to inform value equivalency in the event of one. In this respect, the Suir is a rather typical river in southeast Ireland. The river is 184 kilometres long, but has a total channel length including tributaries of 530 kilometres. It is joined by the River Barrow shortly before emptying into the estuarine environment of Waterford Harbour.

4.2 Information on supporting and related ecosystem services

The Lower Suir forms part of a Special Area for Conservation (SAC) that has been designated for its alluvial and yew woodland. The SAC supports a number of protected species such as lambrey, salmon, pearl mussel and otter as well as good examples of vegetation types. However, while data is available for the SACs, baseline information is slight for other locations. There is no central database of wetlands in Ireland, although map-based information on wetland locations is being developed by Wetland Surveys Ireland.[13] The middle reaches of the river do include a handful of small wetlands, some of which (e.g. Cabragh Marshes) have been heavily modified by eutrophication due to past industrial

---

[12] Amongst the more significant incidents of recent years has been the dumping of hazardous waste at a landfill in County Wicklow and a fire at another landfill in County Kildare (SKM Enviros, 2010). Both incidents presented significant threats, but neither incident led to impacts beyond the site.

[13] www.wetlandsurveysireland.com

activity, but none of these play an important role in the ecological functions of the river. The wetlands therefore provide more instances of species-to-species habitat services for which a HEA could be sufficient than for other ecosystem services. Cultural ecosystem services, such as wildlife viewing, are of local rather than national value.

Baseline information on water quality in the main channel and for tributaries (EPA, 2011b, 2015) is available to help assess impacts. Water quality is classified as being of moderate WFD status, but is recorded as being unpolluted for 62% of sample stations as of 2010.[14] It is known that the aquatic ecology of rivers and wetlands, in particular the mix of plants, invertebrates and micro-organisms, performs an important regulating ecosystem service by assimilating nutrients and pollutants. However, there is no firm evidence for the Suir of the relative role played by these ecosystem services in combination with physical factors such as turbulent or open slow-moving stretches that may introduce necessary oxygen and light.[15] In addition, much of the physical interaction between wetlands and the river occurs in winter when the potentially beneficial regulating microbial activity is low.

Some of the species which contribute to and/or benefit from high water quality are themselves vulnerable to excess nutrients. Many species which perform a regulating service are sensitive to the eutrophication of surface waters by farm nutrients, domestic and industrial waste which is a major environmental challenge in Ireland (EPA, 2008). For instance, the freshwater pearl mussel filters the current for edible debris, but its distribution has already passed a threshold whereby it has suffered severely from the increase in nutrient levels in much of the Suir (Ross, 2006). Species of cultural ecosystem service value, such as eel and salmon, have also evolved in low-nutrient status conditions. All these species are vulnerable to elevated nutrient levels. Both the pearl mussel and eel are at risk of extinction in many Irish rivers including the Suir and may never recover (SRFB, 2008) should a significant pollution incident occur.

---

[14] Based on Environmental Protection Agency (Ireland) Q-values of 3-4 based on presence of key macroinvertebrates.

[15] See Sturt et al (2013) for a discussion of some factors.

For human beings, there are potential ecosystem service benefits in terms of water consumption as the amount of treatment necessary is reduced by the assimilative capacity of the aquatic ecosystem. However, a minimum level of treatment is still required for reasons of public health. Treatment costs at source are not high at an operating cost for a small/medium sized plant of around 8.69c per 1000 litres compared with per capita domestic water use averaging 160 litres per day.[16]

4.3 Information on provisioning and cultural ecosystem services

Now that commercial salmon draft netting and eel fishing have been prohibited due to diminished stocks, provisioning services on the Suir are restricted to the abstraction of water. Most of this abstraction is public. Private abstraction occurs mainly from groundwater. Water from the river and tributaries is consumed by farm animals, but amounts are small due to the typically good rainfall. Only one major factory abstracts water for industrial use. The value of water for different uses can be estimated using a pay-back approach (Moran and Dann, 2008; Renzetti and Dupont, 2003), but this method values water in terms of its contribution to final goods such that the value can appear higher for use in the chemical industry where quantity is the main factor, than in the food sector where water quality is more critical.

Estimates of final cultural ecosystem service values provide one of the main grounds on which to value water quality, but these are restricted to a few water-related activities. There is some kayaking, but recreational angling is the main activity. Salmon and trout stocks are below their potential, but have shown some recovery in response to improvements in water quality (IFI, 2011; O'Grady and Delanty, 2006). Inland Fisheries Ireland records sales of salmon rod licenses, but local permits to fish are required from twelve or more local angling associations at prices of between €15 and €30 per day or €75 and €150 per week. Assuming that the Suir accounts for half of the salmon angling effort in the South-East Region this suggests license revenue of €44,000 and permit income of €365,000 per year.

[16] Figures courtesy of Louth County Council.

Licenses are not required for brown trout, but a similar level of income from permits can be assumed given the importance of the river for this species. This angling effort could contribute to local expenditure of €5 million on the basis of a recent national survey of angling expenditure (IFI, 2013). Minimum social value could be estimated using a mixture of travel cost and expenditure data and would depend on the relative proportions of local and visiting anglers, the availability of substitute sites (including unharmed sections of the same river), and the importance of angling to the local economy. Utility values to anglers in excess of trip costs can be assumed.

Given information on fish stocks and angling activity, an assessment of the social costs of an incident could be made. Depending on the nature of an incident, it could take a long time before the consequences of the damage are realised. Kennedy and Crozier (1997) present sample economic values for cases where losses could extend to future angling seasons in the event of impacts on fish spawning and survival. Stock assessments have been conducted for the Suir (O'Grady and Delanty, 2006), but information on current local conditions and trout numbers may have to depend more on local knowledge than scientific counts. Salmon numbers vary considerably from year to year (IFI, 2011). Physical factors such as tributary drainage works, lack of shade or habitat diversity have a considerable influence as do migratory conditions in the Atlantic.

The outflow from the River Suir also impacts on water quality and final provisioning and cultural services in Waterford Harbour. This transitional environment is important for wading birds and wintering wildfowl, but is less attractive to birdwatchers than sites in neighbouring County Wexford. There is potential to estimate the value of sea angling in Waterford Harbour. There is also potential to place a value on provisioning services. The formerly important commercial harvest of cockle has been suspended due to low stocks, but the local Gigas oyster and blue mussel catch averages 1,100 tonnes per year valued at €5 million (BIM, 2008). The estuary is also thought to be a nursery for commercial fish species

(Ellis et al., 2012), although the importance of European estuaries as nursery grounds is little understood (Hinz et al., 2006) and can only be speculated for Waterford Harbour based on nearby sea catch data.[17] County Waterford has one of the south-east's three main fishing ports at Dunmore East and experiences the highest fishing effort for the Irish coast for demersal species (BIM, 2009).

The role of the inflowing freshwater to the water quality of the transitional and inshore waters depends on many factors that are poorly understood. These include ecosystem services such as presence of shellfish colonies, but also abiotic factors such as depth and the tidal environment. Excess nutrients still flow from the river. Biological Oxygen Demand (BOD) in Waterford Harbour appears to have reduced due to better agricultural practice, but dissolved inorganic nitrogen (DIN) remains at threshold levels in common with others estuaries in the region (EPA, 2007, 2010).

4.4 Other use and non-use utility values

Aside from angling, it can be assumed that local people also value the environment of the River Suir and Waterford Harbour for both use and non-use. However, while the river is visible from roads and bridges, only short stretches are accessible for passive recreation such as walking. Non-use values are likely to be significant, but there have been no surveys to demonstrate the scale or heterogeneity of values that the public attaches to the River Suir. The only primary study to date in Ireland has been for the Boyne (Stithou et al., 2011), a river in the east of Ireland. In this survey, water quality was represented by indicators of ecological status and survey-based economic valuation was used to estimate the welfare values associated with significant improvements in water quality.

[17] Species with significant estuary dependence include whelk, cockle (market value €174,000 for the ports of Dunmore East, Kilmore Quay & Dungannon). Species though to have some dependence: flounder, sole, sprat, plaice, cod, whiting, herring, razorshells (value €12m). Possible dependence: cod, blue mussel, clam, scallop, crab, lobster and cod (value €21m). Market value includes ecosystem services value, but also financial and human capital return.

Norton et al (2012) used both the data provided by Stithou et al, and an average of estimates from five other EU studies, to transfer value estimates to other rivers in Ireland. On this basis, the water quality of the Suir was valued at €40.60 compared with an estimate of €51.73 for the Boyne. However, while it is possible to adjust values for the catchment population and water status of a new location, it is difficult for 'benefit transfer' to determine the preferences of the population at this location (Rosenberger and Stanley, 2006). The Norton et al study commented on these limitations including the use of willingness-to-pay relative the alternative of willingness-to-accept. Indeed, this is a very pertinent factor where estimates of equivalency are needed. As discussed above, willingness-to-accept is more appropriate to the compensatory actions required by the ELD following environmental losses, but few applied studies have used this approach.

Figure 1 provides a list of the essential ecosystem services along with approximate values or levels of participation. For the purpose of determining equivalency, detailed investigation would be needed in some cases, but in many others this would need to be combined with ecological information on the broad nature of habitat services.

[Figure 1 here]

4.5 Wastewater costs

An alternative means of valuing the social benefit of existing water quality and of the avoided costs of deterioration is offered by the amount spent on wastewater treatment to maintain the quality, including the assimilative capacity, of receiving waters in line with the WFD (Hernandez-Sancho et al., 2010). Both capital and operating costs for wastewater tend to be high relative to the treatment of potable water at source. The wastewater plant for Waterford City is located on the Lower Suir. It serves a population equivalent of 143,500 [18] and has an estimated discharge of 250 million litres per year. The capital cost of the new plant

[18] Based on a population of 47,000 and commercial discharges.

was €38 million on its opening in 2011.[19] Operating costs differ considerably between plants depending on size and the amount of treatment required. Sewage loadings in Waterford have generally been around 10mg/l suggesting actual costs of €0.63 per kilogramme of sludge removed based on averages estimated for different sized plants in the UK (Oxera, 2006). On this basis, the treatment cost for BOD would be equivalent to between €1.1 million and €1.85 million per year given daily reported removal of up to 5,000kg compared with an aggregate willingness-to-pay based on the Norton et al estimate of €1.9 million. Additional costs apply to nitrate which is a limiting nutrient in the transitional environment of Waterford Harbour (EPA, 2010). Phosphorous removal would require expensive tertiary treatment and would involve both higher operating costs and significant capital expenditure. At present this treatment is essentially being provided by the natural ecosystem services of the transitional and inshore environment.

In principle, the cost of different levels of wastewater treatment reflects society's willingness to pay for this artificial service in order to maintain water quality for the purposes of water supply, habitat, biodiversity and amenity. In practice, criteria are set by the EU Directive, but ultimately depend on public support. If the costs of treatment were deemed to be "disproportionate" as defined by the WFD[20], then this could be an indication that the public's willingness-to-pay for this infrastructure has been exceeded. In this case, the Directive requires that a *derogation* be sought from the targets that have been set.

[19] http://www.environ.ie/en/Environment/Water/WaterServices/News/MainBody,26475,en.htm
[20] Although the WFD uses the term "disproportionate costs", it does not define what these costs are, although Commission guidance EC, 2009. WFD CIS Guidance Document No. 2 Exemptions to the Envrironmental Objectives does suggest that an effort be made to estimate the social and environmental costs and benefits.

5. Discussion

If a serious pollution or related incident were to affect the River Suir it would be challenging to arrive at estimates for equivalency values given that information on baseline ecological conditions is largely absent outside of designated sites. Transforming whatever information is available into estimates of social value would be further problematic. At best a mix of ecological information on habitat services and some of the more reliable economic information would need to be used together as suggested by Shaw and Wlodarz (2013) to inform a scenario of no net loss.

Angling is the social activity most at risk from an impact on water quality. There is only partial evidence of the relationship of catchable stocks at one location to spawning and migration conditions at another. However, the numbers of anglers and a minimal estimate of their willingness-to-pay could be collected using data on license and permit sales, possibly supplemented by data on travel costs or a stated preference survey.

Without doubt the river and the quality of its waters is valued as a final cultural ecosystem service by local people, both those who occasionally visit stretches of the river or those who value its existence. Unfortunately, only one primary survey of preferences for water quality has yet been carried out in Ireland. New additional studies would help to validate benefit transfer estimates and improve our knowledge of the values that different communities attach to rivers and their ecosystem services.

Ideally, for economic assessment, information is needed on the values attached to marginal changes in water quality and ecosystem services, but the implications of small changes can be challenging to communicate within stated preference surveys. An understanding of these linkages is often easier to achieve in small workshops or focus groups (Christie et al., 2012). Where information on the intermediate ecological processes is not available, a “water quality ladder” (Mitchell and Carson, 1981; Vaughan, 1986) can be used to illustrate the implications of changes in water quality for species and final goods such as recreation and human health. Hime et al (2009) have subsequently adapted the ladder to relate more closely to ecological quality sought by the WFD.

Samples of European valuation surveys indicate that people are predictably willing-to-pay most for large improvements in water quality from poor or good (Norton et al., 2012). In fact, very few Irish water bodies are in such a poor quality state and half (52%) of Irish rivers are of high or good quality WFD status (EPA, 2010).[21] Rather few surveys ask for willingness-to-accept values for losses, often for fear that this format will cause people to overstate their values for strategic reasons (Knez et al., 1985). However, non-linearity of the valuation function will also result form differences in the valuation of environment losses vis-a-vis gains. The empirical evidence is that people value environmental damage of the type addressed by the ELD more than gains. Remediation should therefore aim to exceed these value estimates and go beyond no net loss if they are to compensate adequately for environmental losses.

6. Conclusion

The European Environmental Liability Directive provides for complementary or compensatory remediation in cases where the restoration of an environment to its original condition is either not possible, or not possible within a short time-frame. The Directive favours the physical remediation in terms of resources or services, but does allow for an estimation of social values especially in the case of interim losses. In the United States in particular, various authors have examined the case for the wider inclusion of social values in damage assessment. A principal issue in these studies has been the likelihood of differences in social values for the damaged and remediated habitat. The distance between these respective values is likely to be further widened by differences in the values attached to environmental losses and gains.

[21] Results are for 2007-09 the latest for which national figures are available.

In many instances, therefore, remediation without consideration of social costs and benefits is incomplete. Ideally, a mix of information on habitat services and social values is needed for damage assessment based on the results of new primary ecological and economic studies. As a minimum, generic indicators that demonstrate the location, prevalence and scale of social values could be prepared ex-ante for resources such as rivers.[22]

This paper has discussed the importance of estimating social values to inform remediation, particularly as these relate to ecosystem services. It has also sought to demonstrate some of the practical problems that are likely to be encountered. These include our often poor understanding of the linkages between many ecological processes and ecosystem services. Taking the example of the River Suir in the south-east of Ireland, the paper shows that rather little is known of some of the processes that are relevant to water quality, fisheries and habitats. It explains how likely changes in final provisioning or cultural services can be discussed as an alternative to detail on intermediate processes, but that these services cannot always be presumed to be significant. In the case of the River Suir, direct amenity is largely restricted to angling. Information on passive and non-use values is limited to a single study of another river and subject to the imprecision of benefit transfer. Furthermore, most survey data does not include willingness-to-accept information that could best be used to estimate losses in social value. In the absence of much data, it is argued that the amount spent on wastewater treatment, as a measure of social value, is as good an indicator as any.

Indication of figures and tables

One figure

[22] In Ireland, the Environmental Protection Agency, is examining the case for a Beneficial Use Index, namely a spatial database of social and environmental values.

Acknowledgements

This paper has been written on the basis of research funded by the Environmental Protection Agency (Ireland)

References

BIM, 2008. Assessment Monitoring and Management of Dundalk Bay and Waterford Estuary Cockle Fisheries 2007. Bord Iascaigh Mara.
BIM, 2009. Atlas of the commercial fisheries around Ireland. Bord Iascaigh Mara, Oranmore, Co Galway.
Boyd, J., 2010. Lost Ecosystem Goods and Services as a Measure of Marine Oil Pollution Damages, in: Future, R.f.t. (Ed.), Discussion Paper 10-31, Washington DC.
Christie, M., Fazey, I., Cooper, R., Hyde, H., Kenter, J.O., 2012. An evaluation of monetary and non-monetary techniques for assessing the importance of biodiversity and ecosystem services to people in countries with developing economies. Ecological Economics, 69-80.
Crowe, M., ten Kate, K., 2010. Biodiversity Offsets: Policy Options for Government. BBOP.
Dunford, R.W., Ginn, T.C., Desvousges, W.H., 2004. The use of habitat equivalency analysis in natural resource damage assessments. Ecological Economics 48, 49-70.
EC, 2007. Guidance document on Article 6(4) of the 'Habitats Directive' 92/43/EEC.
EC, 2009. WFD CIS Guidance Document No. 2 Exemptions to the Envrironmental Objectives
Ellis, J.R., Milligan, S.P., Readdy, L., Taylor, N., Brown, M.J., 2012. Spawning and nursery grounds of selected fish species in UK waters, Cefas Science Series Technical Report No. 147. Centre for Environment, Fisheries and Aquaculture Science, Lowestoft, p. 56.
EPA, 2007. Water Quality in Ireland 2004-2006. Environmental Protection Agency.
EPA, 2008. Ireland's Environment. Environmental Protection Agency, Johnstown Castle, Wexford.
EPA, 2010. Water quality in Ireland 2007-2009, in: McGarrigle, M., Lucey, J., O'Cinneida, M.E. (Eds.). Environmental Protection Agency, Dublin.
EPA, 2011a. Environmental Liability Regulations. Guidance Document, in: Carty, G., Herlihy, S., Chadwick, P., Smith, D., Foley, B., Mortherway, K. (Eds.). Environmental Protection Agency, Johnston Castle, Wexford.
EPA, 2011b. Report on River Water Quality in Waterford City and County 2011, Integrated Water Quality Report 2011 - South East Ireland. Environmental Protection Agency, Johnstown Castle, Wexford, Ireland.
EPA, 2015. EPA Geo Portal. Water Framework Directive, 2015 ed. Environmental Protection Agency (Ireland).

Findlay, C.S.T., Bourdages, J., 2000. Response time of wetland biodiversity to road construction on adjacent lands. Conservation Biology 14, 86-94.
Flores, N.E., Thacher, J., 2002. Money, who needs it? Natural resource damage assessment. Contemporary Economic Policy, 171-178.
Gala et al, W., 2009. Ecological Risk Assessment and Natural Resource Damage Assessment: Synthesis of Assessment Procedures. Integrated Environmental Assessment and Management 5, 515-522.
Gollier, C., 2002. Discounting an uncertain future. Journal of Public Economics 85, 149-166.
Gray, N., 2004. Biology of Wastewater Treatment, 2nd ed. Imperial College Press.
Haines-Young, R., Potschin, M., 2013. Common International Classification of Ecosystem Services (CICES): Consultation on Version 4 Aug-Dec 2012, EEA Framework Contract No. EEA/IEA/09/003. European Environment Agency.
Hanemann, W.M., 1991. Willingness to pay, Willingness to Accept: How much can they differ? American Economic Review 81, 635-647.
Harriman, R.E., Gillespie, E., Morrison, B.R.S., 1990. Factors affecting fish survival in Scottish catchments. Cambridge University Press, Cambridge, UK.
Helm, D., 2014. Taking natural capital seriously. Oxford Review of Economic Policy 30, 109-125.
Hernandez-Sancho, F., Molinos-Senante, M., Sala-Garrido, R., 2010. Economic valuation of environmental benefits from wastewater treatment processes: An empirical approach for Spain. Science of the Total Environment, 953-957.
Hill, D., Fasham, M., Tucker, G., Shewry, M., Shaw, P., 2005. Handbook of biodiversity methods: Survey, evaluation and monitoring. Cambirdge University Press, Cambridge, UK.
Hime, S., Bateman, I.J., Posen, P., Hutchins, M., 2009. A transferable water quality ladder for conveying use and ecological information within public surveys in: 2009-01, C.W.P.E. (Ed.), University of East Anglia.
Hinz, H., Bergmann, M., Shucksmith, R., Kaise, M.J., Rogers, S.I., 2006. Habitat association of plaice, sole, and lemon sole in the English Channel. Journal of Marine Science 63, 912-927.
IFI, 2011. Wild Salmon and Sea Trout Statistics Report. Inland Fisheries Ireland, Dublin, p. 105.
IFI, 2013. Socio-Economic Study of Recreational Anglers. Inland Fisheries Ireland.
Jones, C., Pease, K., 2007. Restoration-based compensation measures in natural resource liability statutes. Contemporary Economic Policy 15, 111-122.
Kahneman, D., Tversky, A., 1979. Prospect Theory: An analysis of decision under risk". Econometrica 47, 263-292.
Kahneman, D., Tversky, A., 1984. Choices, values and frames. American Psychologist 39, 341-350.
Kennedy, G.J.A., Crozier, W.W., 1997. What is the value of a wild salmon smolt, Salmo salar L.? Fisheries Management and Ecology 4, 103-110.
Knez, P., Smith, V.L., Williams, A.W., 1985. Individual rationality, market rationality, and value estimation. American Economic Review 75, 397-402.
Lipton, J., LeJeune, K., Calewaert, J.-B., Ozdemiroglu, E., 2008. Toolkit for Performing Resource Equivalency Analysis to Asses and Scale Environmental Damage in the European Union, REMEDE. EC Sixth Framework.
Liu, G.D., Wu, W.L., Zhang, J., 2005. Regional differentiation of non-point source pollution of agriculture-derived nitrate nitorgen in groundwater in northern China. Agriculture, Ecosystems and the Environment, 211-220.

Martin-Ortega, J., Brouwer, R., Aiking, H., 2011. Application of a value-based equivalency method to assess environmental damage compensation under the European Environmental Liability Directive. Journal of Environmental Management 92, 1461-1470.
McManus, N., Duggan, O., 2011. Innovative Funding Methods for Landscape Conservation through Carbon and Biodiversity Credits. Heritage Council & Cosain.
MEA, 2005. Ecosystems and Human Well Being: Synthesis Report. Millennium Ecosystem Assessment Washington, USA.
Mitchell, R.C., Carson, R.T., 1981. An Experiment in Determining Willingness to Pay for National Water Quality Improvements. National Centre for Environmental Economics, Environmental Protection Agency (US)
Mooney, H.A., 2002. The debate on the role of biodiversity in ecosystem functioning, in: Loreau, M., Naeem, S., Inchausti, P. (Eds.), Biodiversity and Ecosystem Functioning. Oxford University Press.
Moore, R.P., Robinson, W.D., 2004. Artificial bird nests, external validity, and bias in ecological field studies. Ecology 85, 1562-1567.
Moran, D., Dann, S., 2008. The economic value of water: implications for implementing the WFD in Scotland. Journal of Environmental Management 87, 484-495.
Naeem, S., Loreau, M., Inchausti, P., 2002. Biodiversity and ecosystem functioning: the emergence of a synthetic ecological framework, in: Loreau, M., Naeem, S., Inchausti, P. (Eds.), Biodiversity and Ecosystem Functioning. Oxford University Press.
Natura, 2005. Local Biodiversity Areas: A pilot study on the identification and evaluation of local areas for wildlife and nature conservation. Natura Environmental Consultants and The Heritage Council, The Murrough, Wicklow.
NOAA, 2000, 2006. Habitat Equivalency Analysis: An Overview. National Oceanic and Atmospheric Administration, Washington, DC.
Norton, D., Hynes, S., Doherty, E., Buckley, C., Campbell, D., Stithou, M., 2012. Benefit Transfer for Irish Water, STRIVE Programme 2007-2013. Environmental Protection Agency.
O'Grady, M., Delanty, K., 2006. A Fishery Survey of the River Suir Catchment and Management Recommendations for this Resource. Central Fisheries Board for Southern Regional Fisheries Board.
Oxera, 2006. What is the cost of reducing ammonia, nitrates and BOD in sewage treatment works effluent? Prepared for Ofwat.
Parsons, M., Thoms, M., Capon, T., Capon, S., Reid, M., 2009. *R*esilience and Thresholds in River Ecosystems, Waterlines Report. National Water Commission, Canberra, Australia.
Renzetti, S., Dupont, D.P., 2003. The Value of Water in Manufacturing, in: CSERGE (Ed.), Working Paper ECM 03-03.
Roach, B., Wade, W.W., 2006. Policy evaluation of natural resource injuries using habitat equivalency analysis. Ecological Economics 58, 421-433.
Rosenberger, R.S., Stanley, T.D., 2006. Measurement, generalization and publication: Sources of error in benefit transfers and their management. Ecological Economics 60, 372-378.
Ross, E., 2006. Initiation of a monitoring program for the freshwater pearl mussel, Margaritifera margarifera (L.) in the Cloglagh River (Suir). National Parks and Wildlife Service, Dublin.

Schumacher, I., Zou, B., 2013. Threshold preferences and the environment, Cahier de recherche 2013-16. <hal-00850543>.
Shaw, W.D., Wlodarz, M., 2013. Ecosystems, ecological restoration and economics: Does habitat or resource equivalency analysis mean other economic methods are not needed? Ambio 42, 628-643.
SRFB, 2008. South Eastern River Basin District Eel Management Plan. Southern Regional Fisheries Board.
Stithou, M., Hynes, S., Hanley, N., Campbell, D., 2011. Estimating the Value of Achieving "Good Ecological Status" under the Water Framework Directive in the Boyne River Catchment: A Mixed Multinomial Logit Approach", SEMRU Working Paper Series No. 11. Department of Economics, National University of Ireland Galway.
Strange, E., Galbraith, H., Bickel, S., Mills, D., Beltman, D., Lipton, J., 2002. Determining ecological equivalence in service-to-service scaling of salt marsh restoration. Environmental Management 29, 290-300.
Vaughan, W.J., 1986. The RFF Water Quality Ladder, in: Mitchell, R.C., Carson, R.J. (Eds.), The use of Contingent Valuation data for benefit/cost analysis in water pollution control. Resources for the Future, Washington DC.
Weitzmann, M.L., 1998. Why the far distant future should be discounted at its lowest possible rate. Journal of Environmental Economics and Management 36, 201-208.
Zafonte, M., Hampton, S., 2007. Exploring welfare implications of resource equivalency in natural resource damage assessments. Ecological Economics, 134-145.

There are no appendices for this paper.