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**A transferable water quality ladder for conveying
use and ecological information within public
surveys**

by

Stephanie Hime, Ian J. Bateman, Paulette Posen and
Michael Hutchins

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A transferable water quality ladder for conveying use and ecological information within public surveys¹

Stephanie Hime^{2,3}, Ian J. Bateman², Paulette Posen² and Michael Hutchins⁴

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ABSTRACT

This paper seeks to address the problem of conveying changes in open-water quality to the public within surveys such as within stated preference valuation studies. This is achieved through the creation of a new water quality ladder designed to address both the use value issues which have dominated previous such ladders and ecological change and associated non-use values. The assumptions underpinning the ladder allow its levels to be related to nationally available data concerning measures of water quality so as to enhance the transferability of values derived from using such a scale. This paper also seeks to compare the new water quality ladder to the standards for different ecological states outlined by the UK Technical Advisory Group (UKTAG) for meeting the commitments of the Water Framework Directive and shows how such an analysis can be an aid to applying a benefits transfer.

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1. INTRODUCTION

In contrast to the improvement of potable water supplies, which typically produces private good benefits captured by the household, changes to the quality of open access waters (such as rivers and lakes) typically generate public goods benefits. As such, analysts wishing to value such changes are forced to rely upon non-market valuation techniques. These can be broadly subdivided into two groups. Revealed preference methods rely upon assumptions of weak complementarity to infer values from observed behaviour (Champ et al, 2003). For example, the travel cost approach observes the trade-off between monetisable travel distance/time and the attributes and qualities of recreational sites to infer the value of the latter. The alternative, stated preference methods, attempt to directly elicit values by asking choice or direct valuation questions posed to respondents via survey interviews (Bateman et al., 2002). While both revealed and stated preference methods have a long history of applications, recent years have seen an increase in the use of the latter (Carson, 2007).

An issue for valuation research is to incorporate changes in the quality of non-market goods within analyses. Within the revealed preference travel cost literature such changes are captured by ensuring that data is gathered from surveys conducted at a variety of sites, each characterised by different levels of quality and accessibility. By pooling data across sites, changes in quality are captured and related to variation in visitation and hence values. However, stated preference studies face a double problem in capturing variation in the quality of a good. First, many surveys are conducted away from the sites in question and therefore the existing levels of quality need to be conveyed in a clear and understandable manner to respondents, some of which will not have visited these sites. Second, many stated preference studies concern changes in the provision of goods to hypothetical future levels, some of which may not have been previously attained and hence have not been experienced by respondents irrespective of whether they are users or non-users of the sites in question.

Stated preference researchers attempt to address the issue of conveying change in the quality (or quantity) of a good through the use of explicit information statements. However, the design of such information raises a number of empirical challenges. First and foremost it must convey the nature of the proposed change in provision. To do so the information must initially define the current level of provision (e.g. the initial water quality) and the proposed future level of provision. This of itself can be complex where the technical information is challenging to the general public from whom the survey sample is drawn. Further complexity arises when natural variation means that in fact the current level of provision varies across locations (or indeed is liable to vary temporally due to ongoing changes). In such circumstances it is often common to find that the proposed final level of provision also varies spatially (and/or temporally).

A second and often overlooked issue is to ensure that the information provided to respondents is such that the analyst can relate initial and proposed future quality states to existing environmental data to allow the extrapolation of derived values to real world locations and policy changes. So, for example, water quality states need to be related to existing data covering the extent of the desired study area.

A number of approaches have been used to convey information regarding water quality. Perhaps the most well known of these is the Resources For the Future (RFF) water quality ladder (Vaughan, 1986; Mitchell and Carson, 1989; Carson and Mitchell, 1993). This is a use-based measure, describing open water quality in an ascending scale from having no uses, to being suitable for boating, fishing and then swimming. Variants of this approach have been used in a number of stated preference studies right up to the present day (Desvousges et al., 1987; Bateman et al., 2006a). While providing excellent service through the years, the categories used in the RFF are somewhat limited regarding the extent to which they convey the ecological changes (and associated use and, importantly, non-use benefits) implicit in movement up or down the ladder. Furthermore, the ladder focuses upon use categories which do not readily relate to national data on water quality

(which to date typically tend to focus upon water chemistry measures)⁵. This limits the transferability of results.

The limitations of the RFF water quality ladder have been thrown into sharper relief by the introduction of the EU Water Framework Directive (WFD; European Parliament, 2000). This represents a fundamental shift in the management of water quality in Europe. Amongst many fundamental changes to previous practice, the Directive requires the catchment based management of water, entailing a substantial expansion in the spatial scale of water management. A further innovation is the requirement that European waters⁶ be improved to a 'good ecological status' by 2015. Therefore benefits assessment, which is compulsory for all derogations from these requirements, needs to address both the problem of transferability and to convey change in ecological status.

This paper seeks to address the problem of conveying the quality of open-waters to samples drawn from the general public and to do so in a manner which ensures that resultant values can be transferred nationally (and possibly internationally⁷). This is achieved through creation of a new water quality ladder. This has been designed to address not only the use value issues which have dominated previous such ladders but also to capture ecological change (with its associated non-use benefits) and to relate all of this to nationally available data concerning measures of water quality so as to enhance the transferability of value measures derived from using such a scale.

The objectives of this paper are to document the design of a standardised water quality ladder for the UK and link this ladder to current available measures of water quality in the UK. The following sections of the paper go on to describe how illustrations of a generic river area were constructed with regard to the ecology expected at different levels of

⁵ As part of the implementation process for the Water Framework Directive, work is ongoing to agree an EU-wide measure of ecological status. This should be available toward the end of 2008. The water quality ladder described in the present paper should be readily adaptable to such a measure.

⁶ It should be noted, however, that in many instances water remediation would be technically difficult or disproportionately expensive. This might justify derogations, provided that the related conditions are met.

⁷ This ladder will be tested both within the UK and across Europe through a series of common format stated preference surveys carried out through the aegis of the Aquamoney consortium.

measured biological oxygen demand (BOD) and Ammonia. This section continues with a description of how an initial standardised water quality ladder for the UK was modified for use across water bodies within Europe and how the chemical measures of BOD and Ammonia relate to the new WFD categories described by the environment agency (UKTAG, 2008). The remaining sections of this paper are then used to describe how a benefits transfer can be applied with the use of a transferable value function in conjunction with environmental quality measures can within a case study area and beyond to value improvements in water quality.

WATER QUALITY LADDER DESIGN

River quality encompasses a series of attributes or properties that relate to the ecological / biological, aesthetic, geological, chemical and flow characteristics of particular stretches of river (Holmes et al., 1999). Current river water quality within England and Wales is assessed annually under Environment Agency general quality assessments (GQAs) through reference to standards concerning recommended levels of dissolved oxygen, biological oxygen demand, ammonia (total/un-ionised), pH, water hardness (CaCO_3), dissolved copper, and total zinc (Environment Agency, accessed November 2007a; UKTAG, 2008). The WFD requires a substantial shift in assessment practice with the focus moving toward outcomes, in the form of ecological status, rather than chemical composition. However, this poses practical and methodological problems. The UK, has long time series data concerning the chemical composition of open waters although, assessments of macro-invertebrates and aesthetic river features have been a part of river water quality assessment in the UK for over twenty years (Environment Agency web site, accessed February 2009). To date there are few if any systematic assessments of ecological status in its entirety, where ecological status includes all features of the river environment e.g. aquatic plants, macro-invertebrates, bank-side vegetation and algae. Indeed even the meaning of 'ecological status' is the subject of a pan-European debate to be concluded late in 2008. However, as an initial focus UKTAG has sought to determine the biological elements associated with high ecological quality in terms of macro-invertebrates and their links to measured levels of BOD and Ammonia (UKTAG, 2008).

Given the impossibility of generating a measure of 'good ecological quality' prior to its final definition, data collected as part of the general quality assessments (GQAs) of river water in the UK were used as the basis for the creation of our water quality ladder. The wealth of existing research concerning measures such as BOD and Ammonia makes them highly suitable for linkage to ecological processes such as eutrophication and impacts upon fish species.

In order to generate a standard water quality ladder based on highly visual ecological changes a generic river was defined to control for various aspects that effect river classifications including; the level of habitat modification, river flow rate, depth and width. Several river classification schemes exist for British rivers for example; documenting river plant communities (Holmes et al., 1999) or documenting the ecology of special conservation areas (Hatton-Ellis & Grieve, 2003). In addition to these, several Environment agency reports on fish habitat (Environment Agency, accessed November 2007b,c) and river restoration guides (the River Restoration Centre, 2007) give details of plants in and around British rivers. According to Hatton-Ellis & Grieve, (2003) the most prolific river type in the UK is described as meso-eutrophic, or an area in which macrophyte species start to be replaced by algae (Hilton et al., 2006). This classification and its associated overlap (although limited), with plant communities described in Holmes et al., (1999) was deemed to be the most appropriate for the development of the generic river within the water quality ladder. Several assumptions about the nature of the generic river were made including: the existence of some areas of high flow to encourage *Salmonoid* fish species (see below), some areas of depths < 1m and widths of up to 5m (Holmes et al., 1999; Halton-Ellis & Grieve, 2003). In addition river shape, flow⁸, shade, temperature and width were assumed to remain largely unchanged between ecological states in the generic water quality ladder. Furthermore, flush time i.e. the amount of time taken for the volume of river water in a particular river stretch to pass through that river stretch entirely (Hilton et al., 2006) was assumed to remain at a relatively slow rate of

⁸ Flow rates will change slightly as plants species change from macrophyte communities to algal based communities. This will in turn somewhat increase the level of sediment as a eutrophic state is reached as plants roots are no longer available to prevent erosion (Hilton et al., 2006).

0.05 ms⁻¹ (Hatton-Ellis & Grieve, 2003) in all categories of the water quality ladder. We also assumed that light was not limited by large amounts of shade on any part of the generic river in visualisations, and that fish communities could be defined within each ecological category. Man made modifications such as the addition of weirs, dams and canalisation were excluded from this assessment so as to make the resulting illustrations more generally applicable to water bodies.

Each ecological category within the water quality ladder was described by various attributes including fish life, aquatic vegetation, river bank vegetation, substrate composition and water clarity. The overall shape and depth of the generic river shown within the water quality ladder was specifically chosen to meet the requirements described in (the River Restoration Centre, 2007) to support fish life and particular macrophyte species.

The implementation of the WFD has implications for the levels of permissible pollutants e.g. nitrogen, phosphate, pesticides and sewage as these may affect measures of BOD (Johnstone & Markandya, 2006). In order to convey a highly visible ecological response to changes in water quality as a result of increases in nitrates and phosphates, we tried to consider ecological changes that would be highly visible to most members of the public. The process of eutrophication is a highly visible and highly documented phenomenon resulting in increases in algae and macrophyte biomass along with the decrease of macrophyte diversity, dissolved oxygen and water clarity with increases in pollution (Hilton et al., 2006). The process of eutrophication can be thought of in four key stages oligotrophic, described as having an aquatic plant community dominated by submerged macrophytes; meso-eutrophic, in which floating leaved macrophytes become dominant; eutrophic, where algae begin to out compete macrophyte communities and finally, hyper-eutrophic, where algae is dominant and macrophyte communities are replaced (Modified descriptions from Hilton et al., 2006) or BLUE, GREEN, YELLOW and RED respectively within the generic water quality ladder.

Models currently exist to predict the effect of increases in nutrient levels within lakes and the likely affects on specific species of phytoplankton (Reynolds & Elliot, 2001). However, such models do not exist for river systems thus precluding the definition of nutrient level limits to avoid hyper-eutrophic river states (Hilton et al., 2006). Due to the complex nature of river ecology along with the influence of geology, flow and shade characteristics, these limits are likely to vary significantly between different rivers (Hilton et al., 2006). Therefore the exact point at which one ecological state will move to another cannot be guaranteed at any one level of pollution or nutrient concentration occurring in different water bodies. Moreover in a river with a particularly high flow and flush rate, increases in nutrient pollutants may not result in a transition from one particular stage of eutrophication to the next (Hilton et al., 2006).

To enable the water quality ladder to reflect the changes in water quality which implementation of the WFD might deliver, it was necessary to tie each ecological state to measurable chemical limits e.g. the expected BOD in each category of the generic water quality ladder. This relation allows for a translation from current, nationally available, measures of water quality (available from the Environment Agency) to corresponding improvements in ecological status.

Following on from these assumptions it is possible to link values of BOD from those recommended by the Environment Agency GQA of water chemistry (Environment Agency, accessed November 2007a) and coarse fish habitats as described by (Environment Agency, accessed November 2007b,c) to each category of ecological quality. Additionally, the limits of Ammonia concentration could be tied to each water quality level by following guidance within the Freshwater Fish Directive (European Parliament, 2006).

2.1 Aquatic Plants

Several sources of information were used to determine the aquatic plants that might be present within the generic water quality ladder. Hatton-Ellis and Grieve (2003) detail the

plant species that are most likely to be present within our generic river along with guidance as to how each community responds with increasingly eutrophic conditions. The species composition expected at high quality sites was derived from JNCC (2005) which details the level of species dominance expected within the total cover of aquatic plant species along with those particularly vulnerable to change in river conditions (JNCC, 2005). Cross checking the species in Hatton-Ellis and Grieve (2003) and Holmes et al. (1999) allowed us to determine which aquatic plants would be most dominant at different stages of eutrophication (see Table 1 for a description of aquatic plant communities within each ecological state). A further check with the aquatic species found in lake groups most common within England and Wales also showed considerable overlaps (Duigan et al., 2007). Further information from the (Environment Agency, accessed November 2007b) was used to determine where each species would be situated within the rivers and its optimum depth range. Explanations of aquatic plant composition within (JNCC, 2005; Hatton-Ellis & Grieve, 2003) allowed the specification of the percentage cover of all aquatic plant elements at different stages of eutrophication, these percentages are shown in the left hand side of each column detailing the aquatic plants in Table 1.

Table 1: A description of aquatic plant community composition and species frequency for each category of ecological quality: Percentage values in the top row show the expected river vegetation cover expected for each category.

Highest quality		Lowest quality	
BLUE 50%		GREEN 60%	
BLUE 50%		YELLOW 70%	
BLUE 50%		RED 85%	
No algae; Water plants (described below); Good clarity		Greater amount of aquatic plants taking up more of the open space; Slight increase in water turbidity	
Less aquatic plants with increases in algae; Further increase in turbidity and green hue to the water, Small number of algal mats		Large degree of siltation; Turbid water with a brown hue; Algal mat covering the substrate	
[60]	<i>Rhynchoetium riparoides</i> (20); <u><i>Myriophyllum alterniflorum</i></u> (20); <i>Leptodictyum</i> (<i>Amblystegium</i>) <i>fluviatile</i> (10); <i>Fontinalis antipyretica</i> (10)	[50]	<i>Apium nodiflorum</i> (20); <i>Leptodictyum</i> (<i>Amblystegium</i>) <i>riparium</i> (20); <i>Potamogeton crispus</i> (10)
[25]	<u><i>Ranunculus penicillatus</i></u> ssp. <i>Pseudoalutans</i> (4); <i>Pellia endiviifolia</i> (2); <i>Apium nodiflorum</i> (3); <u><i>Callitriche hamulata</i></u> (10); <i>Leptodictyum</i> (<i>Amblystegium</i>) <i>riparium</i> (3); <i>Rorippa nasturtium-aquaticum</i> (3)	[35]	<i>Rhynchoetium riparoides</i> (15); <u><i>Myriophyllum alterniflorum</i></u> (10); <i>Leptodictyum</i> (<i>Amblystegium</i>) <i>fluviatile</i> (5); <i>Fontinalis antipyretica</i> (5)
[15]	<u><i>Callitriche platycarpa</i></u> (5); <u><i>Callitriche stagnalis</i></u> (2); <i>Potamogeton crispus</i> (2); <i>Potamogeton natans</i> (6)	[15]	<u><i>Callitriche hamulata</i></u> (2); <u><i>Callitriche stagnalis</i></u> (8); <i>Potamogeton crispus</i> (5)

Aquatic plant frequency and species from (Hatton-Ellis & Grieve, 2003; JNCC, 2005; Holmes et al., 1999). Underlined species show plants that occur in up to 0.5m depth, **bold** type species names shows plants that occur in up to 0.5-1.25m depth; defined within (Environment Agency, accessed November 2007b). Numbers in square brackets show the % breakdown of the total vegetation cover associated with plant groups, while numbers in brackets show the % breakdown to the plant species level.

2.2 Bank-side Vegetation

The bank-side vegetation displayed in the water quality ladder focused on four common elements: *Phragmites australis*; *Rorippa nasturtium*; *Glyceria sp.*; *Salix sp.* and common grass or turf. The most prominent change shown between states within the water quality ladder is the absence of *Rorippa nasturtium* and *Glyceria sp.* in lower categories (see Table 2). These species were described in (Environment Agency, accessed November 2007b; Holmes et al., 1999) as common elements found on river sides with the exception of the *Salix sp.*, which was added to shade part of the river and thus encourage the spawning of brown trout with higher quality categories (Wild Trout Trust, accessed 2007; Environment Agency, accessed November 2007b).

Table 2: A description of common river bank vegetation and species frequency for each category of ecological quality

Highest quality		Lowest quality	
BLUE	GREEN	YELLOW	RED
<i>Phragmites australis</i> (Reed) Where Found: form beds on river banks.* ⁺	<i>Phragmites australis</i> (Reed)	<i>Phragmites australis</i> (Reed)	
<i>Rorippa nasturtium</i> (Water cress) Where Found: shallow flowing water, half- immersed.*	<i>Rorippa nasturtium</i> (Water cress)		
<i>Glyceria sp.</i> (Sweet-grass) Where Found: By shallow water.*	<i>Glyceria sp.</i> (Sweet-grass)		
<i>Salix sp.</i> (Willow) Where found: on land. Turf	<i>Salix sp.</i> (Willow) Turf	<i>Salix sp.</i> (Willow) Turf (increase in prominence)	<i>Salix sp.</i> (Willow) Turf (increase in prominence)

Bank-side vegetation ordered by frequency (Holmes et al., 1999). *Information modified from (Environment Agency, 2007b), ⁺Information from Arkive (2007a) and MarLIN (2007).

2.3 Fish Species

The habitat information of a number of common UK fish species was analysed from the ecological profiles available from (Arkive, accessed May 2007b), and (Environment Agency, accessed November 2007d). *Salmonoid* species e.g. Brown trout and species of Salmon; Vendace and Minnow were all considered indicators of high water quality as they are present in well oxygenated waters with shale / pebble substrate (Arkive, accessed May 2007b; Eklöv et al., 1999) and as such were classified within the highest river quality categories. However, *Cyprinid* species or coarse fish such as Carp which are able to withstand slightly lower levels of water quality (Environment Agency, accessed November 2007c) and tend to be located in areas that are sandier /muddy Environments for fish, (EA report, accessed November 2007) were classified as present across a wider range of water quality categories (European Parliament, 2006; Environment Agency, accessed November 2007c), (see Table 3). Using the GQA's and the FWFD we were able to match the tolerance limits of certain fish species to the measured limits of BOD and Ammonia concentration expected at each category of the generic water quality ladder (see Table 3). Finally, details of the substrates and flow rates associated with each category were also described within the guides for fisheries biology (Environment Agency, accessed November 2007d) and included within Table 3.

Table 3: Describes fish species, chemical limits and substrate types to be displayed within each ecological category of the generic water quality ladder.

Highest quality —————→ **Lowest quality**

BLUE	GREEN	YELLOW	RED
Fish species			
Mostly game fish with some coarse fish	Virtually no game fish more coarse fish	Virtually no game fish, less coarse fish	No fish
Brown trout (mid) central area fastest flow	-	-	-
Minnow (high)	-	-	-
Vendace (mid)	-	-	-
Barbel (mid)	-	-	-
Chub (mid)	-	-	-
-	Bream	Bream	-
-	Common Carp (mid) mid-water	Common Carp (low) Whole area – not edges (silt)	-
-	Perch (less) mid-water	-	-
-	Roach (mid) mid-water	Roach (high) Whole area – not edges (silt)	-
-	Rudd (mid) mid-water	Rudd (low) Whole area – not edges (silt)	-
Pike (v. low)	Pike (v. low) mid-water	Pike (v. low) Whole area – not edges (silt)	-
-	-	Stickle Back (mid) edges as small fish, not where too much silt	-
Chemical limits			
BOD Limit** < 4mgI ⁻¹ Cat: A & B Ammonia < 0.6 mgNI ⁻¹ Freshwater fish directive limit game BOD Limit = 3 mgI ⁻¹	BOD Limit < 6mgI ⁻¹ Cat: C Ammonia < 1.3 mgNI ⁻¹	BOD Limit < 8mgI ⁻¹ Cat: D Ammonia < 2.5 mgNI ⁻¹ Freshwater fish directive limit BOD Limit = 6	BOD > 8mgI ⁻¹ Cat: E & F Ammonia > 2.5mgNI ⁻¹
Flow rate overall = 0.05 ms-1 (Hatton-Ellis & Grieve, 2003)			
Patches of faster flow	Patches of lower flow	Low flow	No fast patches
Substrate types			
Gravel / pebble substrate, boulders no algae	Small gravel / sand, boulders and algae	Mud	Mud

**These BOD limitations (Environment Agency, accessed November 2007a; UKTAG, 2008) cross checked with the Fresh Water Fish Directive (European Parliament, 2006) and (Environment Agency, 2007c). Each of these BOD categories also has a corresponding ammonia concentration. The frequency of fish species to be shown in each illustration was given a relative value as the number of fish that could physically be shown within each illustration.

2.4 Icons

In addition to the visualisations developed for each category within the water quality ladder icons relating to the different recreational activities that can be carried out at rivers with certain categories of water quality i.e. fish species expected and whether or not common birds were present were also added. Two additional icons relating to use i.e. swimming and boating were included and are affected most at extreme levels of eutrophication due to increases in algae (Pearson et al., 2001).

2.5 Other Factors

Macro invertebrates and mammals were not overtly displayed within the generic water quality ladder illustrations. In the case of macro invertebrates these could not be seen at the scale of illustration chosen. Separate information regarding such invertebrates could be presented. However, this was not included within the present specification of the water quality ladder as this would be inconsistent with the single viewpoint approach adopted to date. Of course mammals could be seen if present. However, the inclusion of rare yet iconic species e.g. otters, might unduly influence respondents into voting for particular ecological states despite the chances of viewing such species in the wild being relatively low; this was also the case for invasive species which might unduly influence values disproportionately.

2. 6 Comparison Of The Generic Water Quality Ladder With Current Guidance On Good Ecological Status

At the time the generic water quality ladder was being developed within the UEA ChREAM project (Bateman et al., 2006b), UKTAG were defining standards with regard to the high, good, moderate and poor ecological status of river water within the UK. The Table 4 shows the difference between the chemical measures of BOD and Ammonia concentrations used within the generic water quality ladder from highest to lowest quality

and how these translate to the limits defined by UKTAG (2008). For the most part the limits described in both sets of documentation are similar and thus future valuation work considering the improvement of river water using the generic water quality ladder will be directly applicable to the ecological states defined by the UKTAG group.

Table 4: Description of river type within UKTAG document relates to two river types lowland high alkalinity and highland low alkalinity. The first type is applicable for the generic water quality ladder.

BOD			
<u>Generic water quality ladder</u>			
BLUE	GREEN	YELLOW	RED
<4	4-6	6-9	>9
<u>UKTAG</u>			
HIGH	GOOD	MODERATE	POOR
<4	5	6.5	9
Ammonia			
<u>Generic water quality ladder</u>			
BLUE	GREEN	YELLOW	RED
<0.6	0.6-1.3	1.3-2.5	>2.5
<u>UKTAG</u>			
HIGH	GOOD	MODERATE	POOR
0.3	0.6	1.1	2.5

(Values taken from UKTAG, 2008)

3 Generation Of Water Quality Ladder Illustrations And There Use Within

Valuation Studies

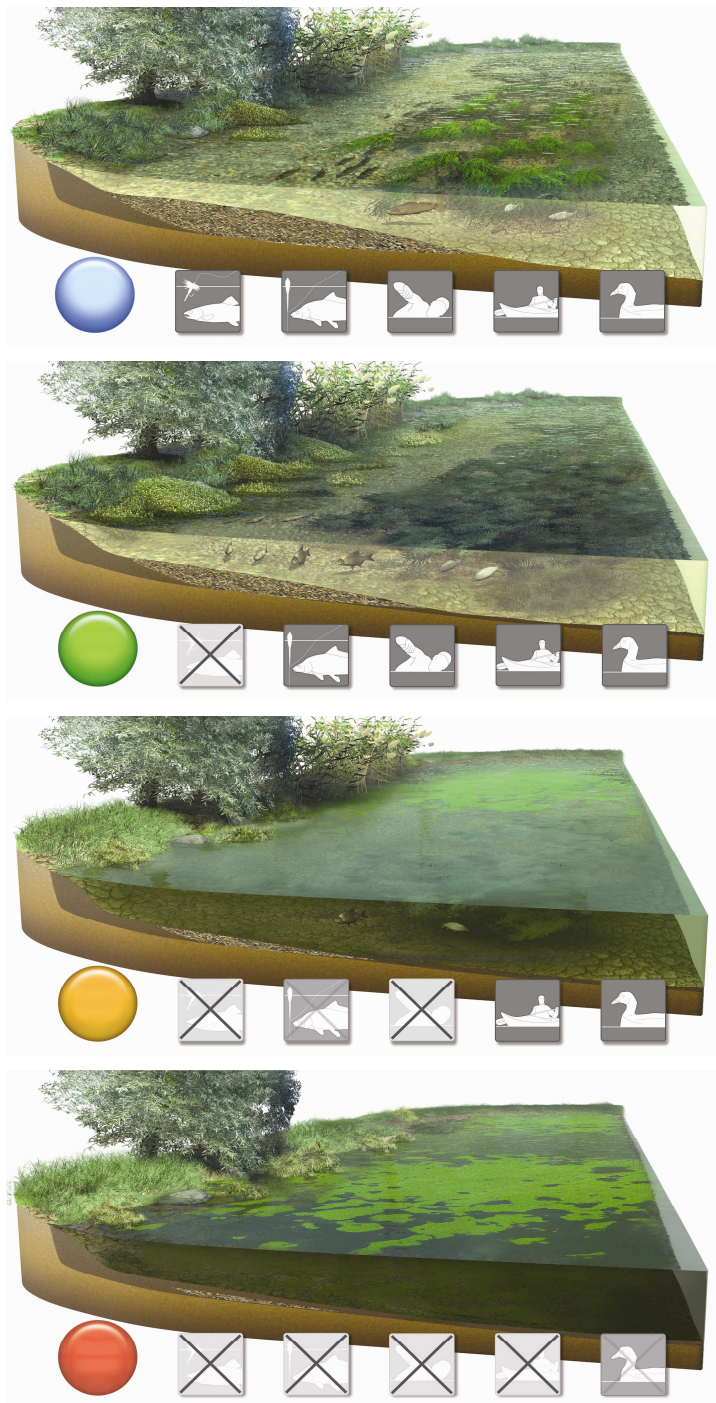
In addition to the above sections on the design of the water quality ladder each plant and fish specie had to be evaluated in terms of their suitability for a computer generated visualisation. As for macro invertebrates certain aquatic plant and fish species were not illustrated due to their small scale e.g. minnow. In addition, some aquatic species e.g. *Potamogeton crispus* comprised too smaller part of the overall vegetation cover to be

included within the generic water quality ladder. The modelling software Cinema 4D (version 10) (MAXON, 2008) was used to generate each illustration. After the development of an initial generic water quality ladder describing British rivers, the illustrations were modified to refer to a generic water body to allow the use of the ladder across both lakes and rivers across Europe (see Figure 1a-d). Table 5 describes the actual species shown within each different water quality level including details of all of the fish and aquatic plant species and bank side vegetation.

Table 4: Species pictured within each quality level illustration

Highest quality	→			Lowest quality
Aquatic plants				
<i>Ranunculus penicillatus</i>	<i>Ranunculus penicillatus</i>	<i>Potamogeton</i>	<i>Algae</i>	
<i>Myriophyllum</i>	<i>Myriophyllum alterniflorum</i>	<i>crispus</i>		
<i>alterniflorum</i>	<i>Potamogeton crispus</i>	<i>Algae</i>		
<i>Rorippa nasturtium-aquaticum</i>	<i>Rorippa nasturtium-aquaticum</i>			
Fish species				
Trout	Bream	Bream		
Chubb	Roach	Carp		
	Carp			
Bank side vegetation				
<i>Rorippa nasturtium-aquaticum</i>	<i>Rorippa nasturtium-aquaticum</i>			
Sweet-grasses, <i>Glyceria sp.</i>	Sweet-grasses, <i>Glyceria sp.</i>	Common reed		
Common reed (<i>Phragmites australis</i>)	Common reed (<i>Phragmites australis</i>)	(<i>Phragmites australis</i>)		
Willow	Willow	Willow	Willow	

Figure 1: The generic water quality ladder



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4 Use of the Water Quality Ladder

It is best to demonstrate how current measures of water quality can be used in conjunction with the generic water quality ladder with a case study example. In order to display current water quality in terms of the categories defined within our generic water quality ladder several steps were taken. First, the mean current measures for BOD and Ammonia concentrations measured from 1986 to 1997 by the Environmental Agency were determined for each sampling station on three rivers Wharfe, Aire and Calder within the Humber region our case study area. These mean values were then converted into the appropriate generic water quality ladder measures i.e. where the mean BOD at sample point one was 5mg l^{-1} , this was converted to the generic water quality ladder value of GREEN (see Table 3). Finally, maps of the case study area were developed (see Figures 2 and 3) to show the mean generic water quality level of each sampling point taking into account either BOD (Figure 2) or Ammonia concentration (Figure 3).

Figure 2: The mean generic water quality category derived from BOD measures taken at Environmental Agency sampling points from 1986-1997, for Environment Agency sampling points on three rivers; the Wharfe, Aire and Calder.

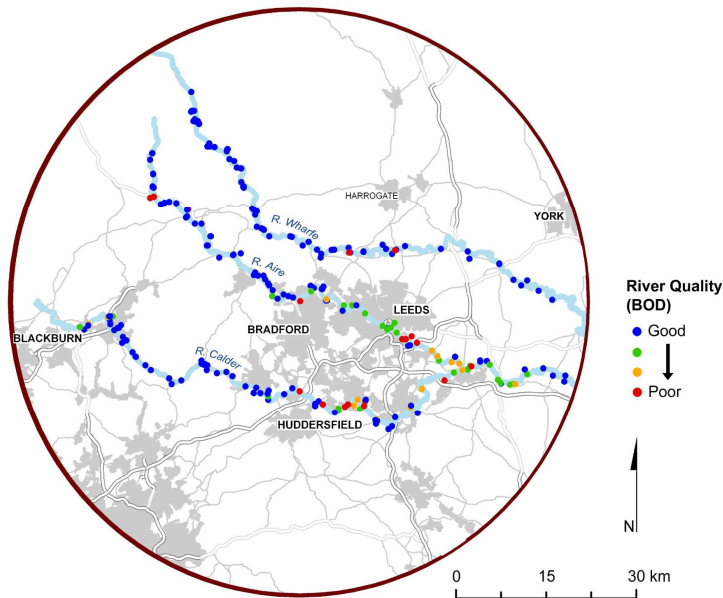
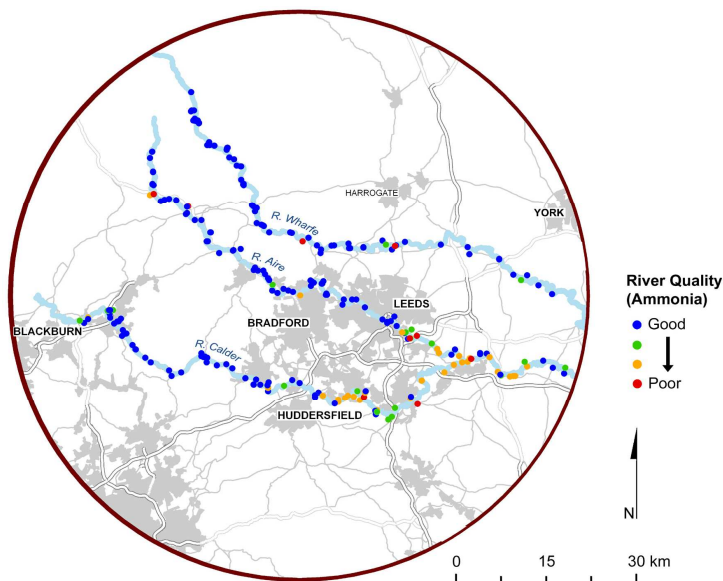
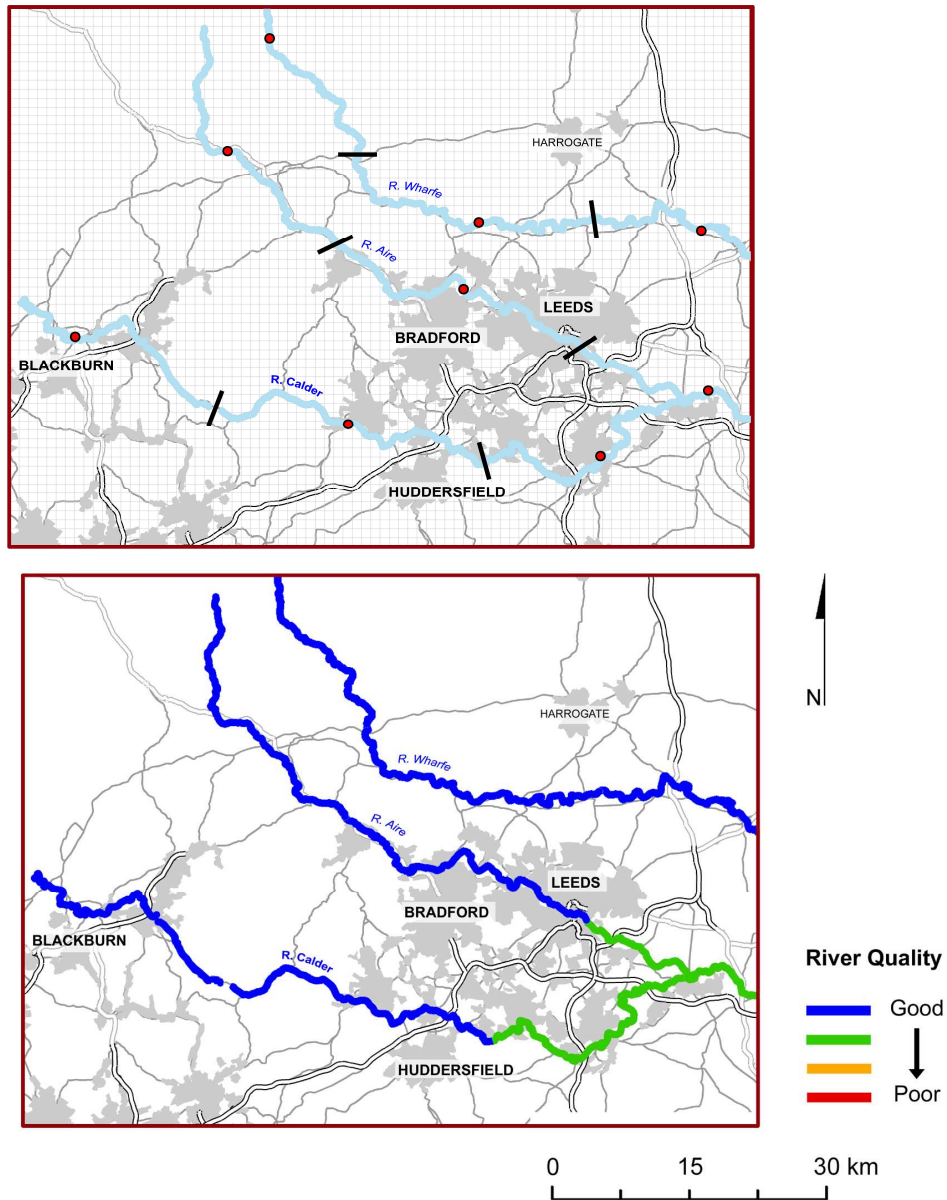


Figure 3: The mean generic water quality category derived from Ammonia concentrations taken at Environmental Agency sampling points from 1986-1997, for Environment Agency sampling points on three rivers; the Wharfe, Aire and Calder.



In order to demonstrate how a benefits transfer approach might be applied in further valuation work these maps had to be simplified. To achieve this simplification the three case study rivers were split into nine stretches (see Figure 3a and 3b). Each stretch was coloured to represent the water quality category within the generic water quality ladder. The appropriate category was calculated by determining the mean category for each stretch from the values of either BOD or Ammonia for each sample point within a each stretch.

Figure 4: Current levels of generic water quality (derived from mean BOD measures and Ammonia concentrations taken at Environment Agency monitoring points from 1986-1997) for nine river stretches of the rivers Wharfe, Aire and Calder.



Figures 2, 3 and 4 describe the current status quo conditions of the Wharfe, Aire and Calder within the Humber region. However, maps depicting potential future levels of water quality arising from the implementation of the WFD or other water quality policies

can also be generated. Thus combinations of these maps allow non-market valuation researchers to visually display in map form changes in the provision of water quality under alternative scenarios with reference to the generic water quality ladder. A comparison of current levels of water quality i.e. the status quo and various alternative schemes provide a potentially useful informational input into stated preference studies using contingent valuation, choice experiment and allied methods (Bateman et al., 2002). The following section demonstrates how such a study may determine the WTP for the local population of the Humber region (see Figures 2, 3 and 4 for the case study area) and how WTP values are distributed across the region.

5. Economic Valuation and Value Transfer

A key application of the generic water quality ladder is its relation to existing data on the spatial distribution of water quality along river stretches so as to generate water quality maps. The national coverage available for chemical measure data provides the requisite basis for the generation of a generally applicable water quality ladder and the estimation of the effects of improvements in those measures for any area of the country (providing the necessary science basis for the spatial transfer of associated benefit values). In addition, a significant advantage of basing valuation exercises upon water quality maps is the explicit incorporation of location within the valuation exercise. Bateman et al. (2006c) have shown that the location of improvements in relation to the individual can highly significantly affect resultant values. Through the use of a map based approach (linked to the spatial analytical capabilities of a geographic information system, GIS) the data necessary to parameterise a spatially explicit value function can be generated. Thus, a transferable value function can be developed from first principles, starting with a structural model of utility.

5.1 Deriving the value transfer function

Ferrini et al., (2008) undertake such development for a spatially dispersed set of goods (rivers) showing how both use and non-use values can be isolated from such a model. They argue that the value of a given improvement can be calculated by examining the utility generated by the improved site divided by the utility from all other sites and an ‘outside good’ of all other recreation options. Although Ferrini et al., report a simulation of their model it is both complex and highly non-linear. A rough approximation to that model⁹ would lead to an empirical specification along the lines of Equation (1):

$$WTP_{ij} = \left(\frac{\beta_1 \Delta Qual_j * \beta_2 \Delta Quant_j * \beta_3 Ln D_{ij}}{\sum_{k=0}^K \beta_4 Qual_k * \beta_5 Quant_k * \beta_6 Ln D_{ik}} \right) * \beta_7 S_i \quad (1)$$

Where:

WTP_{ij}	= the willingness to pay of individual i for a specified improvement in water quality at site j
$\Delta Qual_j$	= a measure of the nature of the water quality change at the improvement site j (e.g. from ‘yellow’ to ‘blue’ quality)
$\Delta Quant_j$	= a measure of the quantity of water quality change at the improvement site j (e.g. X km of improved river stretch)
D_{ij}	= the distance (km) from individual i ’s home to the nearest part of the improved site
$Qual_k$	= a measure of the water quality at substitute site k
$Quant_k$	= a measure of the quantity of that water quality at substitute site k
D_{ik}	= the distance (km) from individual i ’s home to the nearest part of the substitute site
S_i	= a matrix of socioeconomic and demographic characteristics of the individual (e.g. income, etc.).

We generate values for the explanatory variables in Equation (1) by considering a real world area; that around the River Aire in Yorkshire, our case study area shown in Figures

⁹ As Ferrini et al., argue, utility is likely to be a function of the site j utility pre and post the improvement and the utility of all other options pre-and post the improvement.

2, 3 & 4. This Figure specifies values for $\Delta Qual_j$ and $\Delta Quant_j$ for two scenarios; an improvement from ‘yellow’ to ‘blue’ quality (as defined in the water quality ladder shown in Figure 1) for a shorter or longer stretch of the river. The quality of all other rivers stays constant throughout (thus defining $\Delta Qual_k$ and $\Delta Quant_k$). A regular grid (based on the Ordnance Survey grid) was used to define nearly 5000 possible outset locations (home addresses) across the area. A GIS was then used to calculate travel distances from a each outset location to each access point both on the Aire and the potential substitute Rivers of the Wharfe and Calder¹⁰. This gives values for the D_{ij} and D_{ik} variables in Equation (1). Finally, for illustrative simplicity a single socioeconomic variable is considered, household income, with values for each grid square being taken from ONS Census ward data (thus defining S_i)

Value transfer exercises would normally obtain values for the various beta coefficients shown in Equation (1) through reviews of the literature. However, there are very few studies which have explicitly incorporated issues to do with the spatial location of sites into their analyses and, to our knowledge, none which have considered the spatial configuration of substitutes¹¹. Therefore for illustrative purposes we adopt the approach of Ferrini et al. , and use plausible beta values which are in line with the following, theory driven, expectations:

- $\beta_1 > 0$ (i.e. WTP rises as the quality improvement increases);
- $1 > \beta_2 > 0$ (i.e. WTP increases but at a declining rate as the quantity of improved river increases);
- $\beta_3 < 0$ (i.e. the further the improved site for the individual’s home the lower will be their WTP for that improvement);
- $\beta_4 < 0$ (i.e. the higher the quality at substitutes the lower the WTP for improvement at site j);

¹⁰ In future analyses further potential substitutes will be assessed along the lines suggested by Jones et al., (2002).

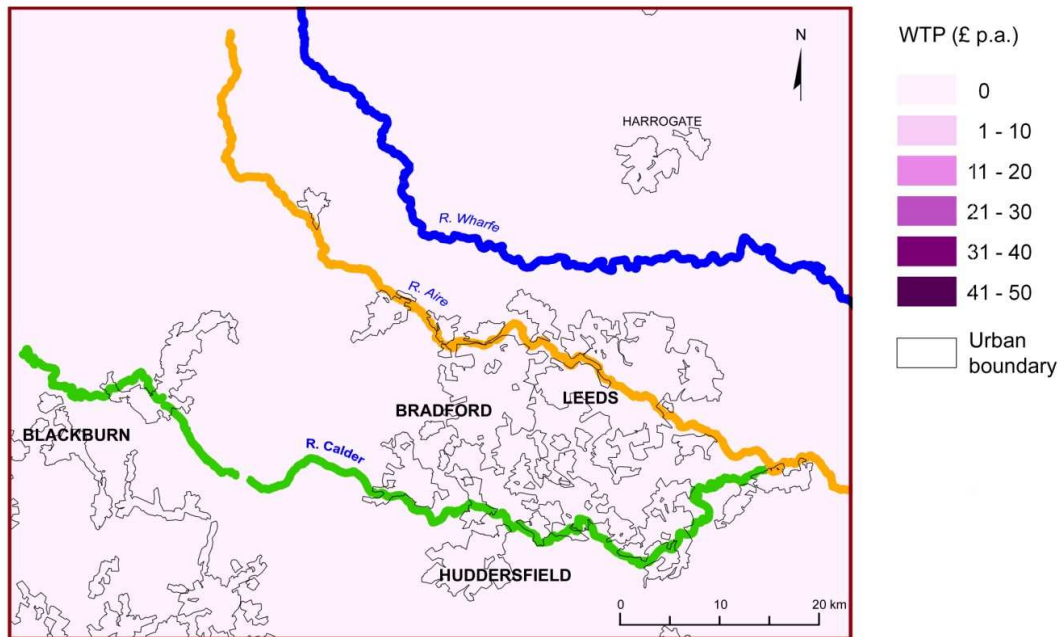
¹¹ The ChREAM project (Bateman et al., 2006c) has recently completed a survey of over two thousand households in the study area considered here. This study includes all of the locational attributes of improvement and substitute sites. Results from this analysis will be available during 2009.

- $0 > \beta_5 > -1$ (i.e. the greater the quantity of high quality substitute river stretches so the lower the WTP for the improved site);
- $\beta_6 > 0$ (i.e. the more distant the substitutes the higher the WTP for the improvement site);
- $\beta_7 > 0$ (this positive sign applies for factors which increase WTP such as higher incomes. The sign will change for those factors which constrain WTP).

5.2 Deriving monetary values for improvement in water quality and benefits transfer

Using the coefficients described in the above section we can work through an example illustrating how the WTP for a change in water quality can be calculated within our case study area. To illustrate a wide variety of possible circumstances, rather than using the rather undifferentiated situation shown in Figure 4 we adopt a more diverse baseline situation as illustrated in Figure 5.

Figure 5: Baseline water quality (set for illustrative purposes only)



By transferring a value function such as that set out previously we can obtain the pattern of predicted WTP per household per annum for an improvement in water quality. Figure 6 illustrates results for such an improvement shown as the blue section of the River Aire. The pattern of values reflects all of the factors captured in our value function at Equation (1). There is a clear distance decay effect with values falling as distance from the improvement increases. There is also a clear substitution effect as values near to the high quality river Wharfe fall rapidly and are zero for virtually all areas north of that river. In contrast the substitution effect of the lower quality River Calder is much weaker and positive values are recorded even some way south of this river. Finally there is a very clear effect of socioeconomic drivers with WTP being substantially higher in the wealthier areas to the north of Leeds and Bradford than in the poorer inner city areas of both cities.

Figure 6: Map of estimated mean willingness to pay (per household, per annum), for a single stretch improvement in water quality

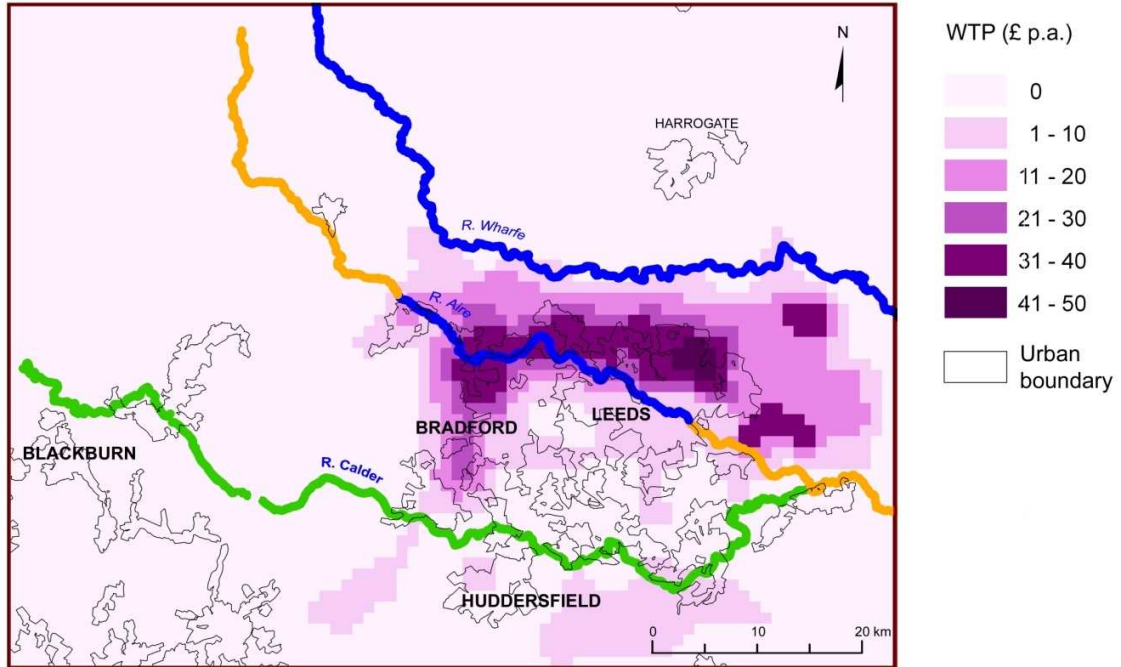
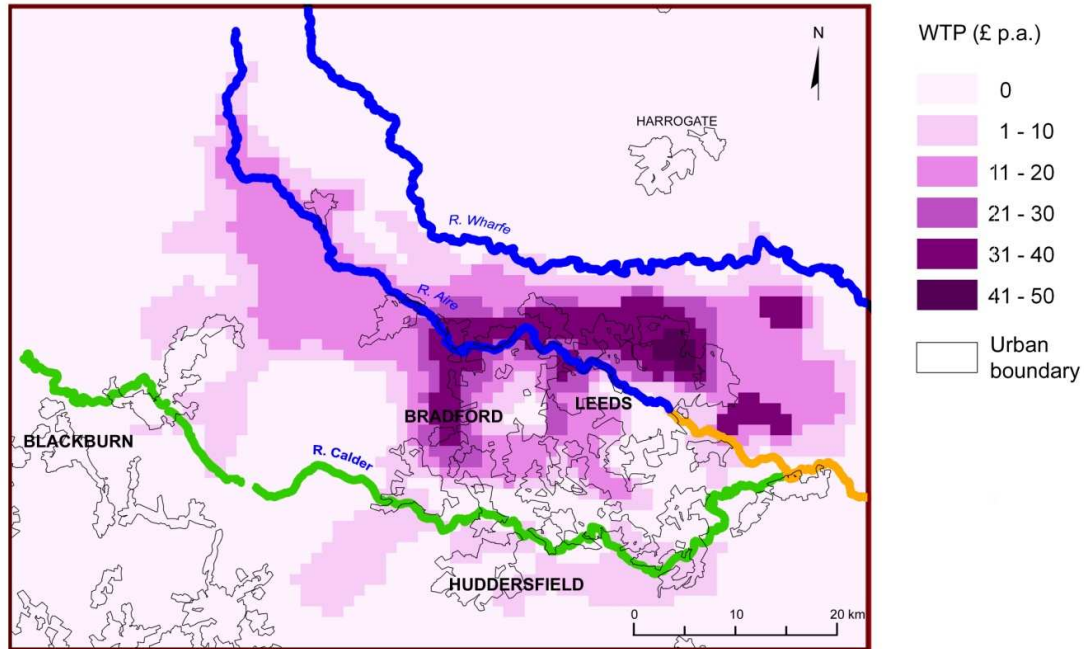


Figure 7 illustrates predicted WTP per household per annum for a longer stretch of improvement in water quality as shown as the extended blue section of the River Aire. Again patterns reflect the factors captured in our value function at Equation (1), the main difference from Figure 6 being the longer area of non-zero values now generated although modest income levels in the upper Aire mean that we do not see the high values generated by the initial project.

Figure 7: Map of estimated mean willingness to pay (per household, per annum), for a larger improvement in water quality



These maps illustrate the potential for value function transfers to spatially target improvements so as to make the most efficient use of limited resources. They also allow the decision maker to consider the re-distributional aspects of environmental improvements (although it should be highlighted that simultaneous maximisation of environmental and distributional goals would only occur by chance and there is always a trade-off when considering multiple goals). For example, the above figure shows that in this case benefits would be higher from creating improvements in high density urban areas than in remote rural areas. Thus the ‘transferable value function’ described in section 5.1 can be applied to the full length of any given improvement or set of improvements from which a value maps can be generated. However, several simplifying assumptions were made. First, the distances used were straight line distances to each river stretch and only a single substitute for each quality level was considered. Second, the model was accurate to 1km^2 only as this was the scale of the data available. Lastly, the

rivers shown within the example case study were large rivers only, in reality smaller tributaries maybe present within an area, and whether these can be included within such an exercise will depend on whether water quality data is readily available.

6. Discussion and Conclusion

This paper sought to demonstrate how current measures of river water quality such as biological oxygen demand could be incorporated into a generic water quality ladder based on highly visual ecological components e.g. aquatic plants and fish species. The construction and use of this water quality ladder will ensure that respondents in different areas actually value the same good i.e. water quality, in the same way without the fear of misinterpretation. In addition, the linking of the water quality ladder to measures of biological oxygen demand allows those seeking the value of a specific improvement in a single area the ability to transfer their results to other rivers in different areas where the appropriate chemical data is available. Finally, this paper also shows the potential for benefits transfer where the 'good' to be valued is described in such a way as to make it applicable to more than a single area. However, the approach shown here relies on a simplified model and as such additional analysis in terms of sensitivity and uncertainty should be included within future work.

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