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Using secondary data to analyse socio-economic impacts of water management actions

Emma L Westling[1*], David N Lerner[1], Liz Sharp[2]

[1*] Corresponding author: Catchment Science Centre, Kroto Research Institute, University of Sheffield, North Campus, Broad Lane, Sheffield, S3 7HQ, U.K.

Telephone: 0114 2225734. Fax: +44 (0)114 222 5701.

Email: e.westling@sheffield.ac.uk

[2] Division of Archaeological, Geographical and Environmental Sciences, University of Bradford, Bradford, West Yorkshire, BD7 1DP, U.K.

Abstract

This paper provides an analysis of the socio-economic impacts of river restoration schemes, and is novel in considering how a wide range of socio-economic variables can be used to understand impacts on the entire resident population within an area. A control-impacted approach was applied to explore differences in socio-economic characteristics of areas within which a restoration scheme had been carried out compared to areas without such a scheme. The results show that significant differences exist between control and impacted areas for a range of socio-economic variables. However, due to constraints in the methods and the data available, there are currently limitations in the extent to which socio-economic impacts of river restoration schemes can be fully explored. Additional datasets that become available in the future may increase the ability to detect associations between improvements in the water environment and socio-economic benefits. However, whilst the secondary data used in this paper are potentially powerful, they should be used alongside other techniques for assessing the impacts of decisions as part of future frameworks to deliver sustainable water management.

Keywords: Sustainable water management, River restoration, Census data, The Index of Multiple Deprivation, Control-impacted analysis

1. Introduction

Sustainable and integrated approaches to water management are starting to gain recognition and acceptance among water managers as a route to more effective decisions (Galaz, 2007). Consequently, there has been a clear change in water policy, moving away from managing water in a fragmented way and towards more holistic approaches (Hooper, 2003; Steyaert and Olliver, 2007). An example of this change can be seen in the EC Water Framework Directive (WFD) which was transposed into UK law in 2000 (EC, 2000). The aims of the WFD include securing 'good' ecological and chemical status for all surface water bodies, and good chemical status for all groundwater bodies, by 2015. More interestingly, the holistic approach embodied by the WFD opens up new possibilities for future water management by requiring the water environment to be managed in an integrated way. Such a management approach should be in line with Meyer's (1997) definition of a healthy ecosystem as "sustainable and resilient, maintaining its ecological structure and function over time while continuing to meet societal needs and expectation". Hence, the costly and ambitious implementation of the WFD should aim to generate multiple environmental, social and economic benefits, and not only to achieve good ecological status (Wharton and Gilvear, 2006). These multiple benefits may include outcomes such as greater community well-being arising from a more amenable local river environment.

Despite increased pressure for sustainable water management, and new holistic policy approaches such as the WFD, environmental, economic and social impacts are currently not integrated in a way that will meet this demand (Pahl-Wostl, 2007). In particular, social impacts are often neglected (Hooper, 2003; Eden & Tunstall, 2006), and little consideration is given to determining whether social gains have resulted from water management decisions and actions (Hooper, 2003). To achieve sustainable water management and fulfil the objectives of the WFD, on-the-ground implementation must be aligned with higher-level aspirations. However, contemporary implementation of many aspects of water management continues to be opportunistic rather than strategic, with clearly stated objectives, monitoring and post project appraisals largely absent (Skinner and Bruce-Burgess 2005). Such opportunistic approaches might be less likely to prioritise social and economic components, and decisions potentially more likely to be driven predominantly by technical and ecological aspects of the water environment. The aspiration of the WFD to implement holistic decision making and actions could be a key driver in moving away from opportunistic and towards more strategic water management approaches.

Evidence of the social and economic benefits derived from water management actions would help to support the development of strategic approaches to their implementation, and would help to ensure that social and economic objectives were prioritised alongside environmental goals in sustainable water management. There is some emerging evidence to suggest that improvements in the water environment can result in a variety of social benefits, such as increased recreational use of the environment, increased aesthetic values, increased local pride and reduced stress levels (see Tapsell, 1995; Tunstall et. al, 2000; Jungwirth et al., 2002; EA, 2006; Gobster et al., 2007). These observations are often based on surveys (see for example EA, 2006), which, although valuable, are often time consuming and costly to carry out. Water management actions also have the potential to influence other areas of the socio-economic system, such as the demographic, income or education characteristics of the resident population. Demographic change due to a changing local environment has been subject to a range of studies. Smith and Phillips (2001) concluded that ‘green’ residential space was a key driver of in-migration to an area, and consequently caused socio-economic change in the characteristics of the resident population. Similar observations were made by Paguette and Domon (2003) who showed that the attractiveness of a landscape had strong associations with in-migration flows and changes in the composition of the rural community. Examples of such links can also be found in urban environments. For example, Sieg et al. (2004) studied the impact of improvements in air quality on land value and population change. They concluded that significant price increases could be detected in properties in communities with substantial air quality improvements, relative to communities with marginal improvements in air quality. Banzhaf and Walsh (2008) also found strong links between improvements in environmental quality and changes in local community demographics. Such research begins to suggest that improvements in water environments not only have the potential to improve amenity values for the resident population, but in the long term also have the potential to impact a range of socio-economic factors, such as demographics, both in rural and urban areas. If these wider socio-economic impacts are not understood in the context of environmental processes, then this is likely to limit the understanding of ecosystems and of ecosystem change in itself (Lazo et al., 1999 as cited in Habron et al., 2004; Eden & Tunstall, 2006). In contrast, if social dynamics are understood in the context of water management, this could highlight key decision-making points and define activities needed in order to successfully implement sustainable water management (Habron et al., 2004), and to deliver multiple benefits from such activities.

Whilst survey methodologies have been used to detect impacts such as increased amenity values or improved aesthetic quality of a river environment (Tapsell, 1995; Gobster and Westphal, 1998; Tunstall et. al, 2000,) other methodologies are potentially suitable for exploring long-term, large-scale effects, such as those resulting from demographic changes as described above. Secondary data, i.e. data already available but originally collected for other purposes, could potentially underpin such methodologies. These data are often collected over long time periods allowing more gradual change, such as that associated with in-migration, to be detected. They also cover a broad set of socio-economic variables and capture a large proportion of the resident population across national scales. Therefore, secondary data could be used to detect impacts, and also to compare these impacts, across a large number of water management activities.

The aim of this paper is to develop a methodology using secondary data that enables the social-economic impacts associated with water management actions to be explored. A further aim is to apply this methodology to one set of actions, namely river restoration. As a result of this work, the limitations and opportunities offered by secondary datasets will also be examined.

2. Methodology

2.1 Datasets

A wide range of socio-economic data are available in the UK, which is the case study area used in this paper. The primary body responsible for collecting, analysing and presenting socio-economic data is the Office for National Statistics (ONS) for England and Wales, the General Register Office for Scotland (GROS) and the Northern Ireland Statistics & Research Agency (NISRA) for Northern Ireland. The most complete and significant socio-economic dataset in the UK is derived from the UK Census, which counts all people and households within the UK every ten years. The data cover information about the population in terms of housing, health, employment, transport, and ethnic groups, and are provided at national, regional and local scale (ONS, 2008a). Other socio-economic data such as crime, employment and health statistics can be derived from various UK governmental departments and local authorities. In contrast to the Census, these other data are updated on a more frequent basis, often annually or every second year. However, they are often not available at the same spatial resolution as the UK Census data.

Since socio-economic data include a wide range of variables, the sources of the data are often fragmented, the data are collected at different temporal and spatial scales, and for different purposes. As a consequence, socio-economic data derived from different sources can be difficult to compare. To overcome this problem, attempts have been made to combine different socio-economic data from different sources into coherent datasets or indices and classifications. The two most complete and commonly used indices in the UK are the 2001 Census Output Area Classification (OAC), and the Index of Multiple Deprivation (IMD). The OAC and the IMD cover a wide range of socio-economic variables, and serve as the basis for exploring the socio-economic characteristics of a population in this paper.

2.1.1 The 2001 Census Output Area Classification

The OAC is the first freely available social classification covering the whole of the UK. The spatial resolution of the data used in the classification is based on Output Areas (OAs), which are the smallest geographical units for which 2001 Census data are available (Vickers et al., 2005). The OAs are built from several postcode areas and are designed to contain roughly equal numbers of people (ONS, 2008b). In the UK there are 223,060 OAs, and on average each OA contains 110 households and 264 people (Vickers et al., 2005). The OAC is based on five main categories: Demographic Structure; Household Composition; Housing; Socio-Economics; and Employment.

When initially developed, the aim of the classification was to use as few Census variables as possible that adequately represented these domains. All Key Statistics (94 variables), the first statistics to be released at OA level, were initially considered for use in the classification. Some variables were merged together and some were removed due to high correlation, which resulted in a final set of 41 variables that were used to produce the five categories described above (Vickers et al., 2005).

2.1.2 The Index of Multiple Deprivation

The Index of Multiple Deprivation (IMD) is available for England, Wales, Scotland and Northern Ireland. Even though some variability occurs across the indices in the different countries, in general they draw upon similar indicators. However, in this paper the IMD for England is used as an example, and will therefore be explained in more detail below.

The IMD is partially based on Census data, but uses a combination of Census data with further data derived from other sources such as the Inland Revenue, the Department of Health and the Department of Transport. The purpose of the IMD is to measure multiple deprivation at the small area level to identify the most disadvantaged areas in England (Noble et al., 2004). The index provides a total measure of deprivation, based on seven different domains which are summarised in Table 1. In addition to a total deprivation score, measures for each deprivation domain are also available. To create the total IMD score the deprivation domains were assigned different weights (Noble et al., 2004) as shown in Table 1.

Table 1. Summary of the seven domains constituting the Indices of Multiple Deprivation (IMD), and the weight used for each domain in calculating the final IMD score.

Indices of Multiple Deprivation (IMD)	
Domain	**Weight (%)**
Income deprivation	22.5
Employment deprivation	22.5
Health deprivation and disability	13.5
Education, Skills and Training	13.5
Barriers to housing and services	9.3
Crime	9.3
Living Environment deprivation	9.3

The IMD is based on data derived from Super Output Areas (SOAs), which are built from groups of the OAs described above (see Figure 1). There are approximately 4-6 OAs within each SOA, and they are designed to be consistent in population size. On average each SOA contains 1500 people (ONS, 2008c). The IMD is available in two forms. Firstly as a rank, which shows how an individual SOA compares to other SOAs in the country, and secondly as an absolute score (Noble et al., 2004).

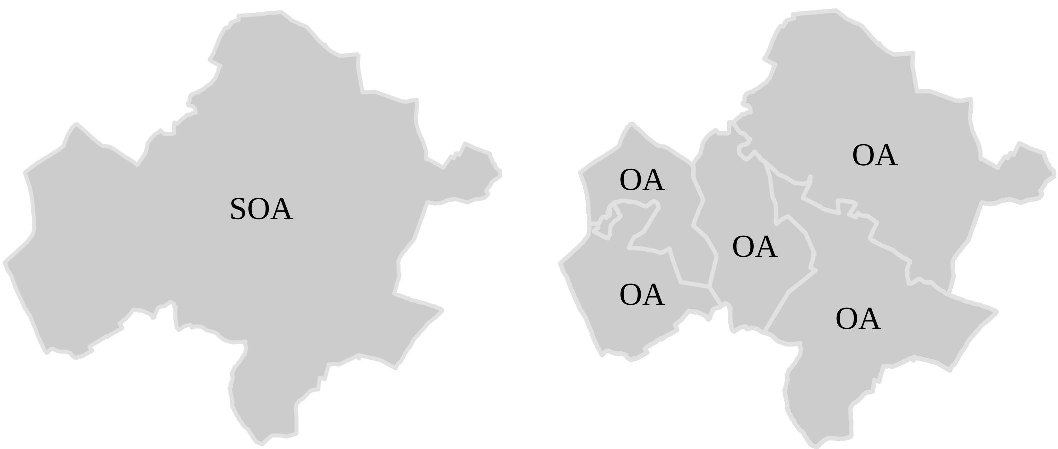

Figure 1. One SOA (a) and the same SOA built from five OAs (b).

2.2 Developing a methodology to investigate the socio-economic impacts of water management actions

Two commonly used approaches that can be applied to evaluate the impact of environmental management actions are the “before-after” approach, and the “control-impacted” approach (Osenberg and Schmitt, 1996). In a before-after approach the indicators, such as those related to socio-economic characteristics, are measured before and after the action of interest. The before scenario is used as a control against which the effects of the after scenario are compared. However, the limited timescale over which suitable socio-economic data are currently available in the UK does not generally allow an analysis of an area before and after the implementation of many water management actions. Some datasets have only been collected over relatively short periods of time, for example data for the IMD that are comparable over time are available for 2004 and 2007 only. Other data, such as that derived from the Census, have been collected over much longer periods of time, but the data released from each individual Census are not currently comparable.

Instead of comparing a set of indicators before and after an action, the control-impacted approach compares outcome indicators for an area within which an action has occurred, against outcome indicators in a control area without the action. Since the control-impacted approach compares areas with and without the management action at a specific point in time, the socio-economic datasets available in the UK are suitable for this type of analysis. The analyses in this paper are therefore based on the control-impacted approach. This approach is a common field assessment approach, and is widely used in monitoring activities (Osenberg and Schmitt, 1996).

2.2.1 Focus on river restoration schemes

Water management potentially includes a wide range of actions and decisions affecting the water environment. At one extreme, implementation of international regulation, such as the WFD, can be envisaged. At the other end of the extreme, water management can include local actions such as introducing a fish pass to a weir to allow easier passage of fish along a river. The difference in character and spatial and temporal scale between different water management actions will have significant implications for how suitable different secondary data are for analysing socio-economic impacts of particular actions. A specific dataset that is suitable for analysing the impacts of one action may not be useful for analysing the impacts of a different action. This paper will focus on one common type of water management action, and develop and apply a methodology to analyse the resulting socio-economic impacts.

The example that will be taken is river restoration, decisions about which are often taken at the regional or local level. River restoration is defined as return to a pre-disturbed state (Cairn, 1991 as cited in Wharton and Gilvear, 2006). So defined, river restoration is often unachievable in many parts of Europe as rivers have been substantially altered over many centuries. However, since river restoration is the most common term for activities involving some form of re-naturalisation of the river it will be used in this paper. River restoration is taken here to include a broad suite of activities taking place within a river or the associated floodplain, which seek to improve the environmental quality of the river. Such activities may include the introduction of secondary channels, fish passes on weirs, or the reconnection of rivers to their floodplains. The number of examples of river restoration schemes has increased substantially in the UK over the last ten years, and this increase is likely to continue into the future, not least because of the potential of river restoration to be employed as a management action to deliver the objectives of the WFD (England et al., 2007).

River restoration is a particularly relevant water management action to analyse since the schemes often claim to deliver multiple gains, including social and economic benefits alongside environmental improvement (Tunstall et al., 2000). However, the evidence to support such claims has not yet been thoroughly tested. This is primarily the result of the lack of post-project monitoring and appraisal associated with many river restoration schemes (Bernhardt et al., 2005), a feature that is certainly true for socio-economic impact analyses (Purcell et al., 2002). One objective of the analysis described in this paper was to evaluate whether evidence could be derived from secondary datasets to test the claims that socio-economic benefits result from river restoration schemes.

2.2.2 The Don as demonstration catchment

The analysis of socio-economic impacts of river restoration reported in this paper is based on eleven restoration schemes and associated control sites in the Don catchment in the north of England (see Figure 2 and Table 2). The Don catchment covers an area of approximately 1700 km^2 and has a diverse topography with the higher altitude, steep valleys of the Peak District in the west contrasting with the low-lying floodplains in the east. Most of the catchment area is densely populated with a total population in the catchment of approximately 1.5 million people. The main rivers in the catchment are River Don (114.1 km), River Dearne (51.9 km) and River Rother (50.8 km) (EA, 2003).

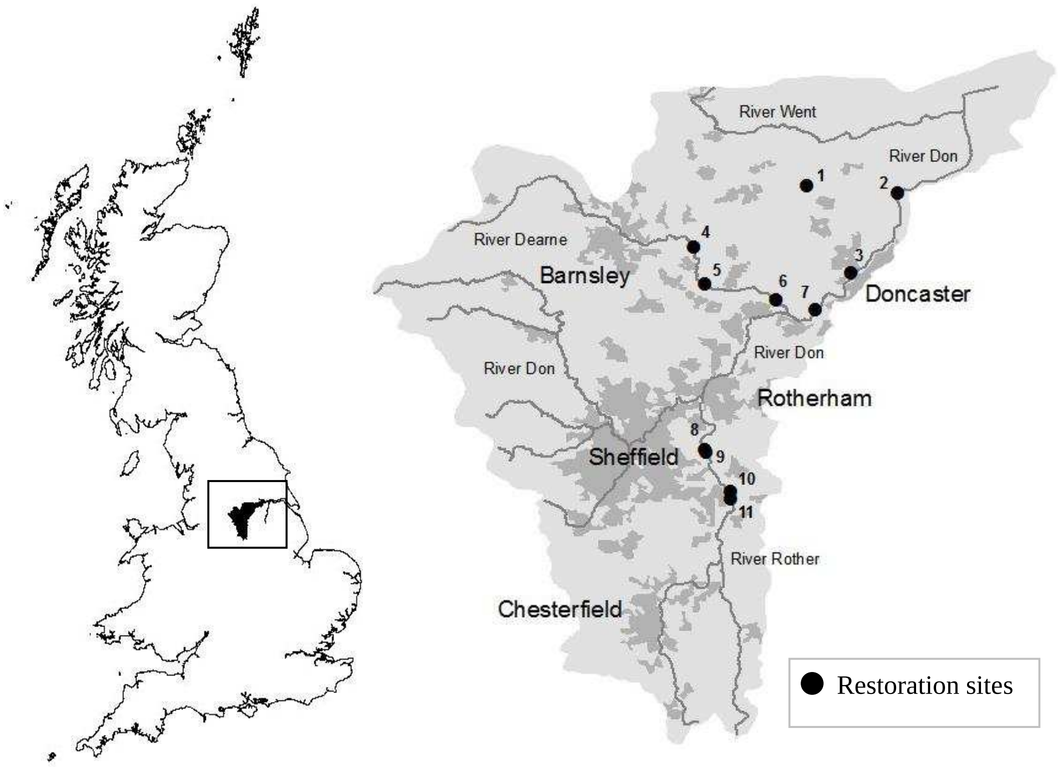

Figure 2. River restoration schemes in the Don Catchment. Note that the location of some sites is obscured by close proximity to others in Fig. 2.

2.2.3 Selection of river restoration schemes

Two approaches to selecting sites for analysing socio-economic impacts of river restoration schemes were considered. The first route was to include a smaller number of schemes that were very similar in character, whilst the second route was to include a larger number of schemes but covering a broader range of type of scheme. The latter route was chosen in this paper in order to include a representative sample of restoration schemes within the Don catchment. The eleven restoration schemes analysed in this paper cover a continuum from small scale projects, such as the introduction of a fish pass or remeandering of a stretch of the river, to larger scale wetland and nature reserve creation. However, the majority of the schemes analysed in this paper were carried out at the river reach scale, rather than at larger scales. It might be assumed that larger scale river restoration schemes such as a wetland creation could have a larger impact on socio-economic characteristics than smaller schemes. However, social impacts may still be expected even from schemes where the ‘physical’ modification to the river is relatively small (Tapsell, 1995). For example, the installation of a fish pass on a weir is designed to ‘restore’ a far larger area of the river than is affected by the physical structure itself. By enabling free passage of fish upstream and downstream, more extensive and sustainable fish populations are expected, which would add to the amenity value of the river. In addition, secondary effects such as increased bird and mammal life might be expected to follow, as these populations are often dependent on fish as an important food source. Such environmental improvements have been shown to be highly valued by local residents (e.g. Tunstall et al. 1999), and may result in social benefits being derived from relatively small river restoration schemes. The aim of this paper is not to compare socio-economic impacts between individual schemes of different size. Instead, a control-impacted approach is adopted, comparing an area where a river restoration scheme has been carried out to a control area. A brief description of each river restoration scheme is given in Table 2.

338 Table 2. Restoration Schemes in the Don catchment.

River Restoration Scheme	Description	Year Completed
1. River Skell	A section of the river was meandered to improve habitat diversity and aesthetic value[1]	2000
2. Broad Ings	A straight river channel was re-meandered and connected to its old bends. Two lakes were also created as part of the scheme. The site is now an important wildlife area.[2]	1992
3. Crimpsall Sluice	A rock chute fish pass was created to replace the sluice that needed updating. The aim was to allow the movement of fish over the obstruction.[3]	2000
4. Little Houghton pond creation	A new channel was created to link the backwater area to the main river to provide a spawning area for fish and to improve wildlife opportunities[4]	1999
5. The Old Moor	A wetland was created on old industrial land and a stretch of the river was re-meandered to increase the biodiversity value of the washland. Old Moor is now a Royal Society for the Protection of Birds (RSPB) nature reserve[5]	2002
6. River Dearne - Low flow channel	To maximise the fishery and wider environmental potential of the river an extensive, sinuous, low-flow channel was created within a much wider flood channel. [6]	1997
7. Sprotborough Flash Nature Reserve	Created by mining subsidence in 1924 and now managed by Yorkshire Wildlife Trust. The site includes a controlled washland. In 1997 the EA carried out works at the site to allow the water levels to be more sensitively managed.[7]	1997
8. River Rother, realignment – Orgreave	The river was diverted and re-meandered through a new channel[8]	1999

[1] The RRC (year unknown) "River Skell channel rehabilitation and education". Project: 200631
[2] Firth C. (2007) Personal communication
[3] The RRC (year unknown) "Crimpsall Rock Chute". Project: 200567
[4] The RRC (year unknown) "Little Houghton pond creation". Project: 200419
[5] Carmichael et al. (2006) "Delivering regeneration through environmental improvement". Environment Agency. Science Project Number: SC040051
[6] The RRC (online). Creating a sinuous low flow channel in an over-widened river. Available from http://therrc.co.uk/pdf/manual/MAN_3_6.pdf (accessed on 22 July 2008)
[7] The Wildlife Trusts (2008) "RESPONSE FROM THE WILDLIFE TRUSTS", The Wildlife Trusts No. 207238.
[8] The RRC (year unknown) "River Rother realignment – Orgreave". Project: 200541

9.	River Rother, rock chute – Orgreave	Construction of a rock chute fish pass on a recently recovered section of the river to allow free passage of fish[9]	1999
10.	River Rother rock weir and introducing fish stock	A rock weir was created to increase the flow velocity to remove deposits of contaminated sediments. Fish was reintroduced to the river and after two years the population was reproducing.[10]	1994
11.	Rother Valley Country Park	Four lakes on old coal mining areas were created to increase recreation opportunities, provide habitats for plants and animals and to create a flood storage system.[11]	1983

2.2.4 Criteria for identifying control sites in a control-impacted analysis

The control-impacted approach relies on the assumption that the only significant difference between the control and impacted site is the presence or absence of the river restoration activity. Hence, all other factors should be as similar as possible between the control and restoration sites (Kerr and Chung, 2001). Selecting suitable control sites is therefore crucial to a robust analysis. Note that the impacted sites described in this paper refer to river restoration sites, whilst the controls are sites without any restoration activity.

In order to meet the assumption that, as far as possible, the only difference between the control and impacted sites was the presence or absence of the river restoration scheme, a number of criteria for selecting the control sites were applied in the analysis. Firstly, the control site needed to have a river flowing within it that had not been affected by a river

[9] The RRC (year unknown) "River Rother - Orgreave Rock Chute". Project: 200566
[10] Firth C. (2007) Interview, 19/7/2007, Doncaster.
[11] Rotherham Metropolitan Borough Council (online). Available from http://www.rothervalley.f9.co.uk/ (accessed on 8 July 2008)

restoration activity. The preferred situation was that the control site included the same river as the impacted site. Secondly, the control sites needed to be a sufficient distance from the river restoration site to ensure that any influence of the restoration activity was eliminated from the resident population within the control site. Research suggests that greenways and recreation areas are mostly visited by nearby residents, often less than two kilometres away (Gobster and Westphal, 1998). Hence, a distance of two kilometres from the restoration site was chosen as a reasonable distance beyond which direct impacts on the resident population due to the restoration activity were assumed to be minor. The rivers within the control site and the impacted site also needed to be the same or similar in terms of their General Quality Assessment (GQA) scores for biology, chemistry, nitrate and phosphate. In addition, River Quality Objectives (RQO), and whether these were complied with, were used to give the most complete check of the control-impacted pairs possible with regards to chemical and biological data. To avoid comparing rural and urban areas, the control site and the impacted site needed to have the same or similar urban-rural characteristics. The Rural and Urban Area Classification 2004 was used to distinguish between rural, suburban and urban areas for this purpose. The classification is provided by the Office for National Statistics (ONS) and is based on differences in household density using clusters of postcode boundaries (Bibby and Shepherd, 2004). In addition, the broad physical characteristics of the river needed to be similar for the control site and the restoration site. For example, aerial photographs were used to visually ensure that comparisons were not made between large rivers and small streams. Finally, the closest site outside the two kilometre boundary that was able to fulfil all the criteria described above was chosen as the control.

2.2.5 Developing a methodology for comparing socio-economic indicators in control and impacted sites

Analysing socio-economic impacts of river restoration schemes requires a boundary within which the socio-economic characteristics of the population, and the impacts on those characteristics due to the restoration activity, can be assessed. Even though the spatial resolution of the datasets used for the analysis is relatively high, they do not necessarily serve as a sufficient base for the analyses. Figure 3 shows the location of a river and floodplain restoration activity that occurred in the Rother Valley Country Park near to Rotherham in the Don catchment, as well as the surrounding Super Output Area (SOA) boundaries. Simply using the SOA within which the restoration activity lies as a base for the analysis would give a potentially inaccurate result, by including residents who live a considerable distance (over 3 km) from the restoration activity. Conversely, residents living close to the site, more likely to be impacted by the improved water environment yet outside of the specific SOA, would be excluded in such approach. It is more appropriate to use distance from the restoration site to create a boundary for the analysis, rather than apply the spatial units at which the socio-economic data were originally released. A 1 km buffer was therefore created around each control and impacted site. The grid reference for each restoration scheme was used to create the centre point of the buffer. This assumes that the restoration scheme is a point, which is not true for all of the restoration sites. However, all restoration sites were kept as points in order to compare buffer areas that were uniform in size.

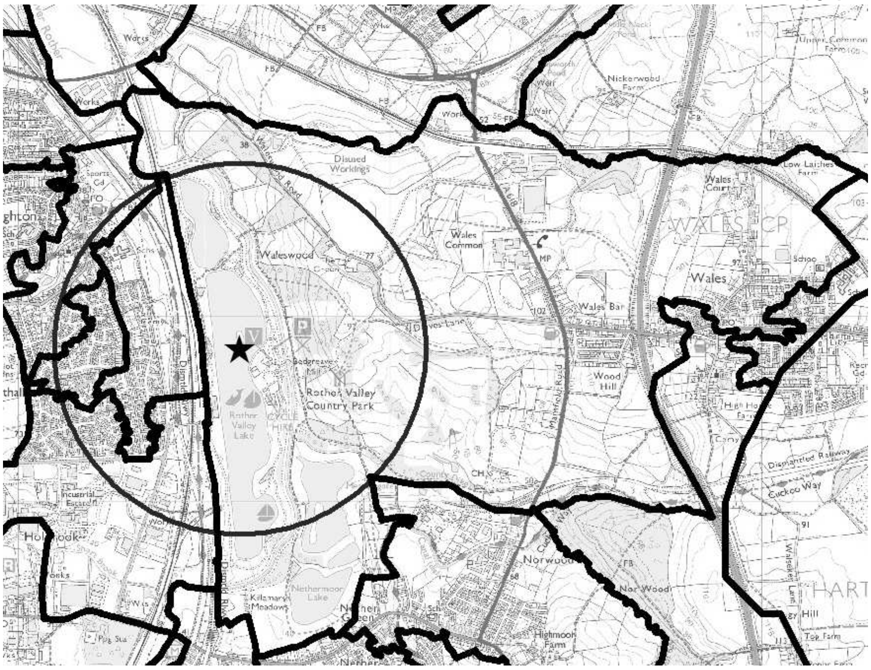

Figure 3. Example of restoration scheme (star) with a one kilometre buffer including multiple SOAs (black boundaries).

In calculating the socio-economic characteristics of the area within the buffer, a weighting could be applied to each individual SOA based on the proportion of the area of the SOA that falls within the 1 km buffer. However, applying this type of simple area weighting assumes that the resident population is evenly distributed within the SOAs, which is rarely the case. To address this problem, the location of the residents must be taken into account in the analysis as far as possible. Therefore, the proportion of the SOA's population, rather than the area of each SOA, inside of the buffer must be estimated. The proportion of the total SOA population within the buffer can then be used as a weight to apply to any socio-economic variable in the analysis. A methodology to obtain a more accurate estimate of the population within the SOA, by using the population data that is available at OA level, was developed. Using the proportion of each OA within the buffer to estimate the SOA population within the buffer still assumes that the population is distributed evenly across the OA. This remains a simplification, but the error associated with the estimate of the population within the buffer, and therefore the weighting factor, is reduced substantially compared to using other approaches. The approach was applied to IMD total and IMD domain data that are available at SOA level. A similar weighting approach has been developed separately by Huby et al (2007) to calculate voter turnout percentage for SOAs. For the analyses based on Census data, a second weighting was not necessary since the data is already reported at OA level. Hence, the proportion of the OA area within the 1 km buffer was used as a weighting factor.

Following Brunsdon et al. (2002), a weighted mean and weighted standard deviation value were calculated for each buffer based on the weighted scores for each individual SOA or OA within the buffer, using equations 1 and 2 below:

$$\bar{x} = \frac{\sum w_i \ x_i}{\sum w_i} \qquad (1)$$

where $\bar{x}$ = weighted mean, w_i = weight of the ith SOA or OA within the buffer, x_i = the score of the ith SOA or the OA within the buffer

$$sd_w = \sqrt{\sum (x_i - \bar{x})^2 w_i} \qquad (2)$$

where sd_w = weighted standard deviation, all other terms are as defined for equation 1.

The data were tested to ensure that they met the assumption of normal distributions using the Sharipo-Wilks test. The results of these analyses indicated that none of the data had distributions that were significantly different to the normal distribution at p = 0.05. Paired t-tests were then used to establish whether differences between the control and impacted sites were statistically significant.

The methodology described above uses data and cases from England as an example. However, the methodology is potentially transferable to other areas where socio-economic data at similar temporal and spatial scales are available.

3. Results

The datasets used in this paper allow us to examine the socio-economic impacts of river restoration using data at index level, domain level and variable level. The IMD provides a total deprivation score as well as a score for each individual domain. The OAC allows analysis of socio-economic impacts at individual variable level. This index-to-variable hierarchy maximises the potential to gain insight into the responses of complex socio-economic systems to river restoration, responses that may be hidden if only one hierarchal level of data is used.

3.1 Results of analyses at index level

For the analysis based on the IMD, the deprivation score rather than rank was used. The score provides an absolute measure of the state of individual SOAs rather than a relative measure as provided by the rank, and is suitable for the calculation of weighted means that are used in this analysis. Figure 4 illustrates the total deprivation score based on 2007 IMD data for the control and impacted sites. The scale on both axes shows the deprivation score, which is based on a range from 0-100, where 100 represents the most deprived score. The 1:1 line represents the situation under which the control and impacted sites have identical deprivation scores. Data points above the 1:1 line indicate that a control site is more deprived than the associated impacted site. The total deprivation scores across all eleven control and impacted sites suggest that in eight of eleven cases the control sites were more deprived than the impacted sites. These differences were statistically significant at $p = 0.05$. A similar pattern was seen for total deprivation scores based on 2004 IMD data, where seven of the eleven control sites were more deprived than the impacted sites. These differences were also statistically significant at $p = 0.05$.

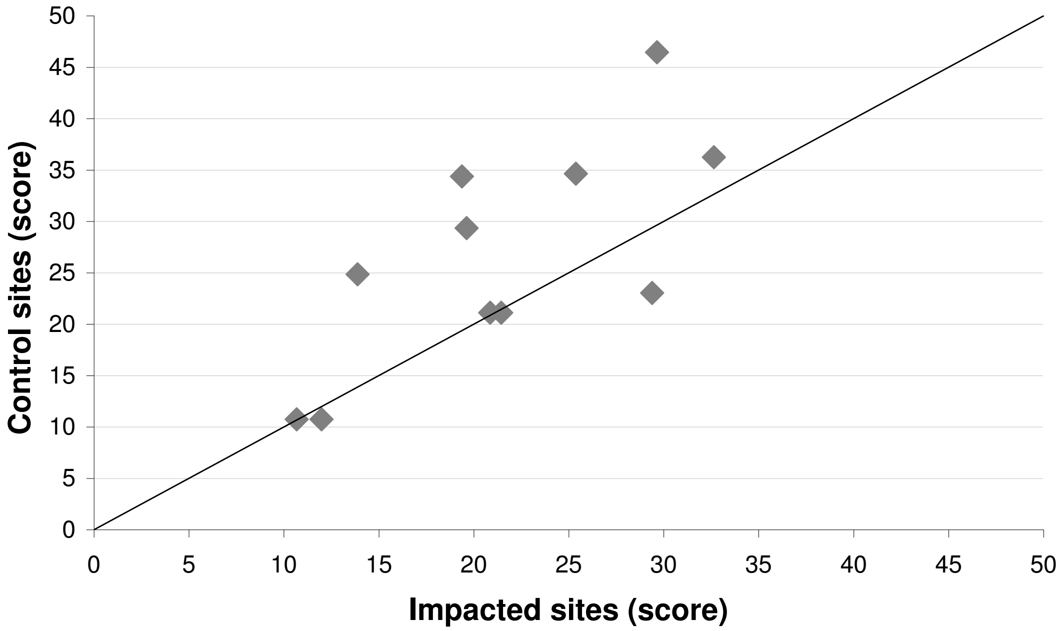

Figure 4. Total IMD score for 2007 for control and impacted sites.

3.2 Results of analyses at domain level

In addition to the total IMD, it is also possible to compare deprivation between the control and impacted sites using individual deprivation domains. Considering only the total score runs the risk of masking potentially important patterns of variability in deprivation at the level of individual domains. The data at domain level are based on the seven domains of deprivation described in Table 1. For each of these domains, higher scores are associated with more deprived SOAs. However, data for the individual domains are not provided on a standardised scale and they have different minimum and maximum values and ranges, making it impossible to directly compare deprivation across different domains for an individual SOA. Despite this, the domain level data allow for a more sophisticated analysis of different types of deprivation, particularly for comparison of individual domains across different SOAs (Noble et al., 2004).

Table 3 summarises the results of the domain-level analyses. The average value of C:I in Table 3 indicates the direction of the difference between the control and impacted pairs, considering all eleven sites together. Values exceeding one indicate that the control sites were more deprived than the impacted sites. Four of the seven domains show the same pattern as described above for the total IMD score, with impacted sites being less deprived that their associated control sites. For three of these four domains, namely Income, Employment and Education, these differences were also significant at $p = 0.05$. The same statistically significant patterns were also observed for these three domains when analysing IMD data from 2004. The four domains that showed impacted sites to be less deprived that their controls were also the domains receiving the highest weighting in the calculation of the total IMD data (Table 1), explaining why impacted sites were significantly less deprived than their associated controls in terms of total deprivation scores. Note that some of the average C:I values in Table 3 are relatively large, but the results of the t-tests indicate that the differences are not significant. This suggests that some individual C:I pairs differed substantially in their domain scores, but that consistent differences were not present for all eleven pairs. Similar patterns emerged from the analyses at variable level (see below).

Table 3. Deprivation domains indicating the direction of any differences between control and impacted sites (C:I), and significance at p¼0.05 (*¼ significant at p¼ 0.05, NS ¼not significant).

Domain	Significance	C:R
Income deprivation Domain	*	1.38
Employment deprivation Domain	*	1.27
Education, skills and training deprivation Domain	*	1.44
Health deprivation and disability Domain	NS	2.09
Barriers to Housing and Services Domain	NS	0.93
The Living Environment deprivation Domain	NS	1.04
Crime Domain	NS	0.89

3.3 Results of analyses at variable level

The choice of socio-economic variables included in the analysis in this paper is based on the list of variables identified in the original OAC (see Vickers et al., 2005). The OAC is different from the IMD in that it is based on a nominal rather than an ordinal scale, but it can be used to explore socio-economic differences and inequalities between the control and impacted sites. The chosen subset of the original OAC variables that was believed to be the most relevant for analysing socio-economic impacts of river restoration schemes, and the outcomes of the analyses, are summarised in Table 4.

Table 4. Census variables indicating the direction of differences between control and impacted sites (C:I), and significance at p¼ 0.05 (* ¼ significant, NS ¼not significant).

Variable	Significance	C:R
Demographic variables		
Resident population aged 0-18 (%)	*	1.19
Resident population aged 19-64 (%)	NS	0.98
Resident population aged 65+ (%)	*	0.92
Household Composition variables		
Residents 16+ not living in a couple and are separated/divorced (%)	*	1.16
Households with one person who is not a pensioner (%)	NS	1.06
Households which are single pensioner households (%)	NS	0.99
Lone parent households with dependent children (%)	*	1.79
Cohabiting or married couple households with no children (%)	*	0.93
Households comprising one family with non-dependent children (%)	NS	0.98
Housing variables		
Households resident in public sector rented accommodation (%)	NS	4.24
Households resident in private/other rented accommodation (%)	NS	1.32
All household spaces which are terraced (%)	NS	2.39
All household spaces which are detached (%)	*	0.82
Household spaces which are flats (%)	NS	5.16
Occupied household spaces without central heating (%)	NS	2.23
Average house size (rooms per household)	NS	0.99
Average number of people per room	*	1.08
Socio-Economic variables		
People aged between 16-74 with higher education qualification (%)	NS	0.89
People aged between 16-74 in routine or semi-routine jobs (%)	NS	1.13
Households with 2 or more cars (%)	NS	0.90
People who reported suffering from a limiting long term illness (%)	NS	1.02
Employment variables		
People aged 16-74 who are students (%)	NS	1.10
Economically active people aged 16-74 unemployed (%)	NS	1.42
Economically active people aged 16-74 working part time (%)	NS	1.01

The demographic variables were included because they potentially explain differences in other variables. For example households with no dependent children are more likely in areas where the percentage of the population aged 65 or over is high. The results of the analyses suggest that the age structure of the resident population differs slightly between the impacted sites and their associated control sites. The control sites had a higher percentage of the population aged 0-18, while the impacted sites had a higher percentage of the population aged 65 or over. However, significant differences were only observed for 5 of the 21 non-demographic OAC variables. For these 5 variables there was no consistent direction of difference, for three of the variables control sites had higher percentages than impacted sites, whilst for the remaining two variables this pattern was reversed.

3.4 Variability of socio-economic characteristics within impacted and control sites

Statistically significant differences were observed between impacted sites and their associated control sites at index, domain and variable levels. However, in analysing only the weighted mean data there is no consideration of the variability of socio-economic characteristics within the individual control and impacted buffers. The methodology developed in this paper also allows examination of this variability. Each individual buffer includes multiple geographical units (SOAs or OAs). Despite the fact that these spatial units are relatively close to each other, they can still differ substantially in socio-economic characteristics. To represent this variability, a weighted standard deviation was calculated for each 1 km buffer for every dataset. Figure 4 showed that a majority of the impacted sites were less deprived than their associated control sites in terms of their weighted mean total IMD score for 2007. In addition a paired t-test confirmed that these differences were statistically significant. Figure 5 shows the same total IMD dataset as Figure 4, but here one weighted standard deviation is displayed in addition to the weighted mean data. It is clear that the variability of the IMD total score within any individual buffer is relatively large. Similar observations were made for all other datasets analysed in this work. These findings suggest that whilst average differences may exist between control and impacted sites, there remains substantial variability in socio-economic characteristics even within the relatively small buffers used in this work. This indicates that any interpretation of the mean differences should be made with some care.

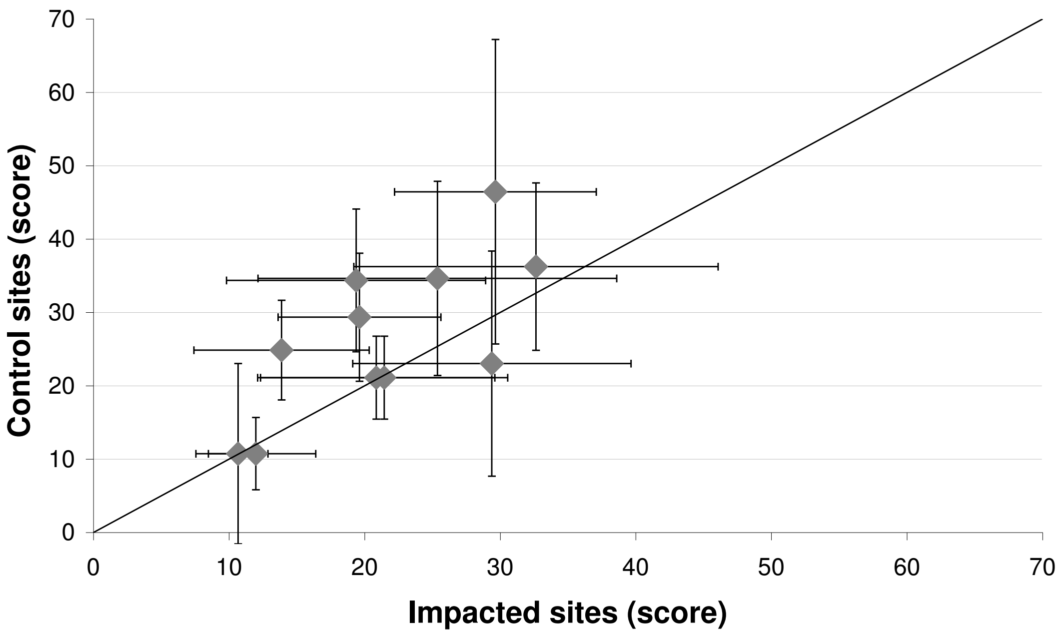

Figure 5. Total IMD score for 2007 for control and impacted sites with variability shown as ± one weighted standard deviation.

4. Discussion

River restoration schemes are often referred to as having the potential to generate multiple benefits, including social and economic gains alongside environmental improvement (see Tapsell, 1995; Tunstall et. al, 2000; EA, 2006; Gobster et al., 2007). However, evidence to support the claims of multiple benefits is largely lacking. The methodology and subsequent analyses presented in this paper provide one of the first attempts to examine the impacts of river restoration activities using a broad range of indicators relating to the socio-economic characteristics of the resident population. The results have shown that significant differences exist between paired control and impacted sites for a range of indicators at index, domain and variable level. For the significant differences observed in IMD, control sites were more deprived than the impacted sites, both for total deprivation and individual domains. For the nominal variables based on Census data it is not possible to identify if an area is 'better' or 'worse' in terms of socio-economic characteristics. However, they do give an indication of differences in socio-economic characteristics between control and impacted sites. The analyses of these Census variables indicate that some significant differences occur. However, there is no consistent direction of difference between restoration and control sites, and the majority of the variables do not show significant differences. In summary, the analyses in this paper highlight significant differences between control and impacted sites for a number of variables. However, conclusive evidence to support the claim that river restoration schemes result in significant impacts across all the variables analysed in this paper was not found.

The ‘mechanisms’ responsible for the significant differences that were observed are potentially related to perceptions about the attractiveness of the local environment. These perceptions have been shown to be an important factor causing in-migration and socio-economic change within an area (e.g. Smith and Phillips, 2001). For example, according to Carter (2001) the environment and quality of life issues are highly prioritised by what he refers to as a ‘new middle class’. Therefore improvements in the local water environment brought about by river restoration may be particularly attractive to these sectors of society, resulting in their relocation to areas in close proximity to restoration schemes, and as a consequence generating shifts in the socio-economic characteristics of the impacted areas. However, to assess these mechanisms fully would require analyses at a different level, using techniques such as questionnaires, focus groups or in-depth interviews with individuals. This paper focuses on the development of a methodology to explore socio-economic impacts of water management activities using secondary data. Analyses of primary data, such as from interviews, and of how secondary and primary data could be combined, are beyond the scope of this paper, but should be the subject of future research.

Secondary datasets are powerful in that they allow for meta-analyses, covering a large number of examples of any particular water management action, and cover a broad range of socio-economic components. Despite this potential, such analyses are rare in the water management context. Socio-economic analyses have been included in decision-support systems for flood risk management (see Haynes et al., 2008), which often include an element of river restoration, but specific research covering the socio-economic impacts of improved water environments is currently lacking. One study in the UK analysed the social distribution of river water quality in England and Wales. The analyses concluded that rivers were less natural and had poorer chemical water quality in more deprived areas, but that there was apparently no relationship between aesthetics and deprivation (EA 2002). There are however examples from other environmental research where secondary data has been used to analyse change. Huby et al. (2006) explored associations between socio-economic components and biodiversity in rural England. According to their results, inclusion of socio-economic variables provides better understanding of the distribution of biodiversity. Socio-economic datasets have also been used to establish associations between the percentage of greenspace in a local area and health. Based on Census and IMD data, Mitchell and Popham (2007) concluded that the percentage of greenspace is associated with better health of the resident population, but that this also depends on the degree of urbanity and level of income deprivation.

The results of the analyses carried out in this paper support the findings of previous work that have begun to show potentially important relationships between socio-economic variables and the state of the environment. An increasing body of evidence suggests that an improved natural environment can result in changes in socio-economic characteristics (Smith and Phillips, 2001; Paguette and Domon, 2003; Sieg et al., 2004; Banzhaf and Walsh, 2008). Such evidence, in combination with increased understanding about the relationships between improved water environments and socio-economic change, could provide a catalyst to encourage future improvements of rivers and other watercourses, both for the environment and for people living close to them. Secondary data has the potential to play an important role in demonstrating theses links between water environments and socio-economic impacts. However, for this to be successful, further developments in the way in which these data are collected, analysed and reported are crucial. These issues are dealt with later in the paper.

Not all socio-economic components analysed in this paper showed significant differences between control and impacted sites. This pattern of some significant and some non-significant differences may reflect the 'true' effects of river restoration, in that such schemes only have an impact on certain socio-economic components. Alternatively, using secondary data as a base for analysis of socio-economic impacts might introduce constraints that limit the degree to which significant impacts can be detected. Any limitations could be particularly significant given the fact that social and indirect economic benefits generated from river restoration schemes are often difficult to identify (Findlay and Taylor, 2006). In light of this, some key limitations of the approach used in this paper, based on the data currently available for analysis in the UK, are addressed below.

4.1 Key limitations in the analysis of socio-economic impacts of river restoration schemes

The first limitation relates to data availability and the consequences for the sampling design used in this paper. Since the socio-economic datasets in the UK are only available for a limited number of dates, the temporal coverage and resolution do not allow the tracking of changes through time that could potentially have occurred due to river restoration schemes. This means that significant differences between the control sites and the impacted sites might have already been present before the restoration activity took place, and therefore not caused by the restoration scheme itself. Instead, the differences in socio-economic and demographic characteristics could be drivers behind the restoration activity, rather than reflecting responses to it. However, for this to be true two conditions must be met. Firstly, factors not related to the river restoration schemes must be responsible for the differences between control and impacted sites. A wide range of factors, such as employment opportunities, the standard of new or existing schools, or other macro-economic conditions, could be responsible for these differences. Such ‘external’ causal factors influencing the result is an issue faced in any place-based control-impacted design. Minimising this issue, and maximising confidence that any significant differences are associated with the river restoration activity, is dependent on using as robust criteria as possible to identify control-impacted pairs. The criteria used in the analysis, as described in the methodology, create what is believed to be a robust control-impacted sampling design. The second condition that must be met is that river restoration schemes must then occur in areas with ‘better’ existing socio-economic characteristics compared to the control sites, not only by chance but because of a specific reason. There is no evidence to suggest that this occurs, and since river restoration activities often follow an opportunistic approach rather than a targeted, strategic approach (Skinner and Bruce-Burgess 2005; Bernhardt et al., 2005), it is believed to be unlikely. Despite the fact that certain socio-economic characteristics such as demographics are believed to be related to pro-environmental behaviour (Carter, 2001; Kahn, 2002), it is not likely that the driving force behind the river restoration schemes analysed in this paper were determined by social factors. The vast majority of the restoration schemes included in this paper were funded and implemented by the

Environment Agency. The objectives of these schemes were almost exclusively environmental, and showed little sign of being driven by any public concern or desire.

A combination of the before-after approach and the control-impacted approach would potentially have provided a more robust sampling design, resulting in greater confidence in the inference that river restoration was associated with significant differences in socio-economic characteristics between control and impacted sites. Since this combined approach allows analysis before and after any given action, it is likely to be more effective in removing other potential causal factors driving differences between the control and impacted sites. One method often used to determine environmental impacts from a given action that combines the two approaches is the before-after control-impact or BACI approach (McDonald et al., 2000). It is however important to bear in mind that the BACI approach is not without limitations; it has been criticised in particular for relying on the use of single control and impacted sites (McDonald et al., 2000). Using several controls per case has the potential to generate more reliable results, but this assumes that multiple, robust control sites can be identified. Given the stringent criteria used in the selection of control sites in the analysis carried out in this paper, it would be a significant challenge to identify further sets of control sites for each impacted site that fulfil the criteria. It is believed that one robust control site rather than a number of weaker controls will result in a higher quality analysis, and as a consequence give a more accurate picture of the socio-economic impacts of water management actions. Fundamentally however, the availability of data in the UK at present cannot support a BACI design, although this situation may change in the future with increased data availability, as discussed below.

The limited temporal coverage and resolution of the secondary socio-economic data is also potentially important when considering the fact that different river restoration schemes were completed at different lengths of time before the collection of the secondary data used in the analyses. This could be important if the differences between the control and impacted sites were expected to change through time, or if different areas within which individual restorations have occurred were expected to respond at different rates. If data were available at a high temporal resolution then both of these issues could be addressed. Nevertheless, based on analyses of data used in this paper, there was no indication that time since completion of the restoration activity was related to the magnitude of the difference between a control and impacted site.

The second key limitation refers to scale of the river restoration activities analysed in this paper. The restoration schemes generally involve site specific activities covering a relatively small physical area, although the schemes used in the analyses span the typical range of river restoration activities occurring in the UK (see Table 2). The socio-economic data used to construct the IMD and the OAC represent population-level characteristics that can be affected both by local and by larger-scale factors. The fact that a number of the variables analysed in this paper did not show significant differences between control and impacted sites suggests that they may not be affected by the scale of river restoration schemes examined in this paper. Such variables may require larger-scale interventions, such as extensive urban redevelopment schemes to generate significant changes in their spatial distribution (Vickers et al., 2005).

4.2 Opportunities for using secondary data to explore socio-economic impacts of water management actions

Despite the above limitations there are also emerging opportunities to use secondary data to explore the socio-economic impacts of water management actions such as river restoration. Most limitations are caused by current data availability, and the consequences for the choice of methods that can be applied in the analyses. At present Census data from different years are not comparable, but this is likely to change in the near future. For the 2001 Census data, new geographies (OAs and SOAs) were introduced. The OAs were created as a real ‘statistical geography’ rather than being based on administrative boundaries that are often subject to re-organisation. Despite difficulties in keeping the same statistical boundaries through time due to changing population characteristics, there is a growing emphasis on publishing data using stable geographies. However, the introduction of these new geographies makes comparison of 2001 data with previous Census years difficult. Hence, the potential to re-release previous Census data, that would support time series analyses at the new geographies, is being explored (ONS, 2005). If past and future data were released at stable output geographies, a more sophisticated BACI approach could be applied to explore socio-economic impacts of river restoration activities. This could result in more certain conclusions regarding the magnitude and causes of differences between areas with a restoration action and areas without such an action. In addition, data collected over time would make it possible to explore whether delayed impacts occur some time after the implementation of an activity. Looking at data from one point in time does not allow this type of trend analysis, an approach which is often important when trying to establish impacts from improvements in the water environment. Comparable indices of deprivation that will become available in the future will, like Census data, increase the potential for exploring socio-economic impacts of water management actions.

The likely evolution of river restoration itself also suggests that the socio-economic data analysed in this paper could become increasingly important. To meet the demands of flood mitigation and for the achievement of objectives under the WFD, which are believed to be two key drivers for the future of river restoration, the schemes must move away from a focus on isolated river stretches and evolve into larger scale, more holistic restoration approaches (Skinner and Bruce-Burgess, 2005; Wharton and Gilvear, 2006). Any resulting socio-economic benefits at these larger scales are more likely to be reflected in the socio-economic indices, domains and variables reviewed in this paper. These indices, domains and variables are therefore likely to become increasingly important decision variables at these scales.

Future analysis of the socio-economic impacts of the full range of water management actions should also have important implications for associated decision making processes. If there is clear evidence of socio-economic impacts due to improvements in the water environment, this evidence could be used in a strategic approach in order to target where the benefits from specific actions, such as river restoration schemes, accrued. Hence a strategic approach, including clearly stated objectives, monitoring and project appraisals, to prioritise schemes generating real improvements is crucial. However, the decision making process behind river restoration schemes, certainly in the UK, is currently far from strategic (Skinner and Bruce-Burgess, 2005). Despite increasing numbers of river restoration schemes, most are still undertaken on an opportunistic basis when new funding and land availability possibilities arise, rather than being strategically planned. In addition, the decision to restore a stretch of a river is often driven by priorities other than the restoration itself, for example river restoration schemes are often undertaken as part of a larger flood mitigation or development scheme. Consequently, little planning for monitoring and post-project appraisal is invested in the river restoration scheme itself, making it difficult to provide the evidence base needed to justify a strategic approach (Skinner and Bruce-Burgess, 2005). Similar observations have been made in the USA, where the vast majority of river restoration schemes are carried out without stated objectives or any form of assessment or monitoring afterwards (Bernhardt et al., 2005). A strategic approach towards river restoration would not only help to maximise environmental and socio-economic benefits, but would also contribute to the monitoring requirements stated in Annex V of the WFD.

Skinner and Bruce-Burgess (2005) suggest a framework for such a strategic approach, and highlight the importance of considering the restoration scheme as part of a larger catchment rather than the river reach in isolation. According to these authors, a strategic basis for river restoration must include baseline data, objectives, method, installation, monitoring, post-project appraisal, maintenance and dissemination. Their framework is however from a strictly ecological perspective, but could be extended to include social and economic components related to the water environment. If environmental, social and economic components were combined in a strategic framework as a base for river restoration schemes and other water management actions, such a framework would be better able to capture the full range of benefits resulting from investment in the schemes. In turn this would support more accurate assessments of management options, leading to more robust decisions. However, for secondary socio-economic data to form a base for such strategic approaches they must be comparable over time and collected and released at a more frequent basis than they are at the moment in the UK. Ideally, data would be collected and released annually, covering the full range of indicators included in this paper.

Finally, policy- and decision-makers must better recognise the range of relevant values that may be affected as a consequence of water management actions. Current understanding of human values and the way to incorporate them in the decision making process is limited (Lockwood, 1999), although different integrated frameworks combining different types of values have been suggested to address this problem (see for example Lockwood, 1999; Morton and Padgitt, 2005; Gobster et al., 2007).The development of similar frameworks, able to integrate secondary data, such as IMD and Census data, with primary data, for example from interviews or questionnaires, is a pressing challenge, although it is outside the scope of this paper. Nevertheless, the importance of adopting a range of methods and data to fully understand the complex interaction between the water environment and human society should be fully recognised.

5. Conclusions

This paper describes an early attempt to develop a methodology and subsequently analyse the socio-economic impacts of river restoration schemes for an extensive resident population across a wide range of variables. The results show that significant differences exist between control and impacted areas for a range of socio-economic variables. Due primarily to limitations in the data currently availability, and consequently the scope of the analyses, and because of the typical scale of river restoration schemes, there are limitations in the extent to which socio-economic impacts of river restoration schemes can be detected. However, new datasets which allow comparisons through time are likely to be available in the near future. In addition, larger scale and more holistic water management actions are also likely to be carried out more frequently. These factors have the potential to increase the ability to explore associations between improvements in the water environment and socio-economic benefits using the secondary datasets examined in this paper.

Although significant differences were observed between some control and impacted sites, drawing conclusions about the causal relationships between river restoration and impacts on socio-economic components remains challenging. However, there are a number of mechanisms that could potentially drive associations between the nature of a local water environment and the socio-economic characteristics of the surrounding resident population. To explore these mechanisms more fully requires qualitative approaches to provide in-depth information on the relationships between people and their local environment. Ideally, information from both qualitative and quantitative approaches would be integrated into a single framework to examine the socio-economic impacts of water management actions. This framework should support a move away from opportunistic and towards strategic approaches to water policy formulation and implementation. Only when such strategic approaches are used to target socio-economic impacts during the design of water management actions, and to measure the impacts by evaluating the actions, will the aspiration for the integration of different sustainability objectives be achieved.

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