The Spatial Impact of Economic Change on RiverWater Quality 1991-2010

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Abstract

This paper, using Ireland as a case study, examines the relationship between economic activities and river water quality. The stipulation from the EU water framework directive (WFD) that all surface waters in the EU must be of 'good ecological status' by 2015 necessitate a quantitative understanding of the major determinants of water quality. Within this context, this paper combines a number of spatial datasets relating to agricultural, residential and industrial activities as well as the level of forest cover to examine the major economic influences on the ecological quality of water resources. It is hoped that providing a comprehensive understanding of the effect of a variety of economic activities that influence the ecological quality of water will be an important tool in the management of risk and will allow for more appropriate land use planning aimed at restoring and maintaining water quality as required by the WFD. Results indicate that the level of forestry, industrial activity, the intensity and type of agricultural activity and the type of wastewater treatment in an area are all critical factors affecting the quality of our water resources. Moreover, the results highlight the importance of a spatial dimension to any analysis as the principal factors affecting water quality often differ across river catchments.

Keywords: water framework directive; agricultural activity; ecological quality

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1. INTRODUCTION

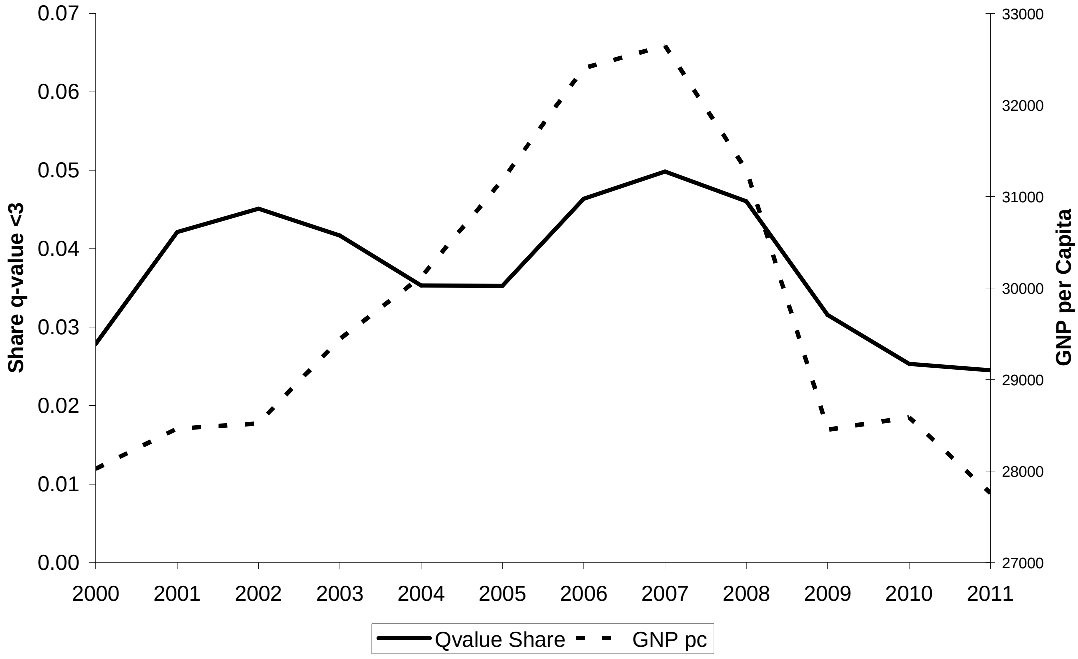
The Water Framework Directive (WFD)(2000/60/EC) adopted by the European Commission (2000) requires the integrated management of water resources throughout the EU. It can be considered as the first framework for EU action in the field of water policy management as it commits all Member States to ensure that all surface water bodies are of 'good status' by 2015 (2000/60/EC). Ecological status is an expression of the quality of the structure and functioning of aquatic ecosystems and is measured using a cross-section of biological, hydrological and physico-chemical parameters. If a water body is already at "good status" or "high status" then it should be maintained; there can hence be no deterioration in quality.

According to the EPA (2012b) 29% of river channel in Ireland is at a level below good status. The actions required to meet WFD obligations will necessitate increased understanding about the factors that affect river water quality. A general conclusion from the existing literature is that land use has a significant effect on river water quality (Woli et al., 2008; Varanka and Luoto, 2012). Much of the research examining the role of land use has been focused on the agricultural sector (Vatn et al., 1997, 1999, 2006; Lennox et al. 1998; Brady, 2003; Cuttle et al. 2006; Brouwer et al., 2008, Fezzi et al. 2008; Pulido-Velázquez et al., 2008; Volk et al., 2008). While research has demonstrated a link between agriculture and water quality, accurate source apportionment at different scales remains a crucial gap in our understanding of the relationship between agriculture and sub-optimal water quality outcomes (Withers and Haygarth, 2007).

In addition to land use, landscape characteristics such as slope, soil and bedrock have been observed to affect river water quality (Silva and Willams, 2001; Varanka and Luoto, 2012). A small number of studies have examined the role of climatic variables such as precipitation and temperature (Larned et al., 2004). Studies that have simultaneously considered different environmental drivers such as, for example, land use, geomorphological and climatic variables are, however, rare. In addition, the spatial scale in most investigations of water quality-environment relationships has been local, i.e. one specific catchment as opposed to regional or national-scale analyses. In the case of lake water quality, Curtis and Morgenrath (2013) estimated using multivariate analysis based on spatial data the effects of land-use and catchment characteristics on lake water quality over the 2004-2009 period. Results from this analysis attributed the variation in water quality across Irish lakes to a range of contributory factors including septic tanks, urban waste water treatment, phosphorous excreted by livestock as well as geomorphological and climatic variables.

Many dimensions of human economic activity can impact upon water quality…The period from 2000 to 2011 can be regarded as a very volatile period economically In Figure 1, we report the trends in the share of unsatisfactory Q-Values over time, noting a correlation with economic activity.

Figure 1 Share of Q-Value (Bad or Poor) over time



Note: Share of Q-value 1-2

The Environmental Protection Agency (EPA, 2006) has indicated that water quality in Ireland is currently at a level below that required by the WFD. More specifically, the EPA found that 29 per cent of river channel length and 8 per cent of lake surface area examined were of unsatisfactory water quality. Nineteen per cent of the estuarine/coastal water bodies examined were eutrophic (over-enriched) while 3 per cent were potentially eutrophic. Finally, 57 per cent of the groundwater sampling locations were contaminated by faecal coliforms and approximately 25 per cent of the groundwater locations examined exceeded the national guideline value for nitrate concentration of drinking water.

The literature on the economics of pollution control on river systems is relatively robust (Aftab, 2007; Xepapadeas 1997; Shortle and Horan 2001), as is the literature relating to the cost and benefits of implementing both the Nitrates Directive and the WFD (Hennessy et al. 2005; Hanley, 1998; Andrews et al., 2002; Barton, 2002; Bateman, 2006) and also the literature relating to the use of irrigation controls in particular to control non-point river pollution (Weinberg et al. 1993; Brooker and Young 1994; Helfand 1995; Albiac et al. 2001).

Much of the economic literature relating to water quality has been concerned with measuring the costs and benefits of water quality improvement. Bateman et al. (2006) outline a proposed methodology for undertaking an integrated cost-benefit analysis of the WFD in relation to the agricultural sector in Britain. This methodology aims to assess the agricultural costs and non-market benefits associated with introduction of different policies to fulfill WFD requirements. In terms of modelling the costs (2006: 224), they propose an integrated hydrological-economic model developed within a project, Catchment Hydrology, Resources, Economics and Management (ChREAM). The main distinction of this project is that it has interdisciplinary dimension – it integrates hydrological, economic, agronomic and geographical elements to model different scenarios of land use and hydrology at the catchment scale (facilitating WFD implementation) and their impact upon farm revenues and profits. This model of agricultural land use tries to link economic analysis with models of nutrient transport, diffuse pollution and consequent biological effects within the water environment. In order to capture the economic benefit of improved water quality, they propose a method using a stated preference surveys using either contingent valuation or choice experiments.

Cuttle et al. (2006), using a number of ad-hoc analyses, try to quantify the cost of an inventory of alternative mechanisms in the UK to achieve improved water quality. Recent studies suggest that to meet WFD goals, management of agricultural practices will have to be changed drastically. Some suggested changes have included the reduction by 50% of the application of fertilisers to crops and grass, sheep stocking rates to be halved and a reduction in cattle stocking rates by 25% (Haygarth et. al., 2003; Bateman, 2006). Although the evidence that links agricultural activities and water pollution has been argued in many papers, the relationships reported, despite being statistically significant may often lack robustness due to the simplicity of the approaches employed and the complexity of the systems under analysis. More recently Withers and Haygarth (2007), reporting on research assessing the contribution of agriculture to eutrophication across Europe, concluded:

" The precise role of agriculture in eutrophication still remains poorly understood and accurate source apportionment at different scales and relevance to impacts remains a crucial gap in our research portfolio that needs addressing" (2007, S1:4)

Some of the economic evaluation work relating to water quality analysis that has been undertaken in Ireland includes Hennessy et al. (2005) who considered the potential impact of nitrates restrictions on the Irish agricultural sector, and in particular, the potential for expansion within the dairy industry after the abolition of milk quotas in 2014. The Environment and Heritage Service (2005) considered the pressures from economic activity on the water quality and the principal users of water in Northern Ireland. The report also presents information in regard to cost recovery of water treatment and highlighted the gaps that exist in ensuring cost effectiveness in implementing a water improvement programme. A similar analysis was undertaken by Joyce (2003), again for the Republic of Ireland. Hutchinson et al. (2004) also undertook for Northern Ireland a survey attempting to quantify the non-market value of improved water quality. No comparable data is available for the Republic of Ireland. Also, in the case of Northern Ireland, it has been argued that agriculture was the primary cause of poor water quality in water bodies (Lennox et al., 1998). However, there has only been a limited amount of work undertaken that attempts to model the full array of factors influencing water quality in Irish water bodies. It is worth noting that there is very little statistical evidence available that comparatively examines the influence of residential and commercial septic tank systems on water quality.

There are a number of research papers that have attempted to connect agriculture to poor water quality water with reported significant results. Using data from 1999-2002, Donohue et al. (2005) analysed the potential influences of human activity on the ecological quality of water. They linked catchments characteristics and water chemistry with the ecological status of 797 hydrologically independent rivers throughout Ireland, finding that both human settlement (in terms of urban land use and by extension, population density) and agricultural activities (in terms of pasture/arable land use and animal stocking density) were related to water quality. The paper used bivariate statistical analysis, thus while focusing on upstream activities within water catchment, the analysis did not capture the complex hydrological interaction between activity, weather and local environmental conditions such as slope, soil and geological attributes. A detailed measure of population density in a rural setting was not included in the analysis.

As Donohue et al. (2005) point out diffuse nutrient pollution has been shown many times to provide a substantial risk to the quality of surface waters. Previously, Foy et al. (1995) and Haygarth and Jarvis (1999) also both demonstrated the strong linkages between diffuse nutrient pollution and water quality in river and lakes. The issue of how diffuse nutrient emissions vary with season and management has also been examined by Jennings et al. (2002) and the appropriate resolution of the source—pathways link, which is important for both understanding and managing pollution transport, has been analysed by Heathwaite et al. (2003).

In this paper, we add to this literature by investigating the importance of a number of key drivers of water quality levels in Irish river systems by combining data from EPA water quality monitoring stations with spatially referenced information on the river catchments, information from the Irish census of agriculture, septic tank density data and population density data in a Geographical Information System (GIS) framework.. The main factors associated with water quality in Irish rivers are assessed using an ordered probit model.

In this study, we develop an ordered probit model using recorded water quality ratings (Q-values) by the Environmental Protection Agency across river monitoring stations as our dependent variable, to explain variations in river water quality. Specifically, we combine a number of spatial datasets to explore the effect of land use, geomorphological and climatic variables on river water quality during the period 1991-2011 across the Republic of Ireland. One further contribution of our research is that by utilising the panel nature of our dataset we are able to examine if the effect of various land use activities on river water quality has changed over time. Globally our results are in line with existing literature in that we find agricultural activities to have an important effect on river water quality. However, our analysis indicates that there is an important temporal dimension as the effect of both livestock and cereal production on river water quality has diminished over time. Further variables reflective of land use that we find to be significantly related with river water quality include forestry cover, the location of landfills and proportion of households served by a septic tank sewage treatment system.

It could be argued that the observed effect of our explanatory variables reflecting different land use activities is due in part because these variables are correlated with other potentially more important explanatory geomorphological and climatic factors. For example, any finding of a negative effect between agricultural variables and river water quality could arise in part to the association between the location of various agricultural activities with geomorphological and climatic variables such as rainfall and soil type. To control for the effect of these potential confounding variables, a variety of geomorphological variables such as measures of elevation, slope, soil type as well as climatic variables such as rainfall and temperature are included as covariates in our regression analysis. This allows us to provide a more robust examination of the effect of different land use activities on river water quality than previous work. Our analysis highlights the important effect of biophysical characteristics and climatic factors as we find slope, elevation, and soil type to be important predictors of river water quality. In the section that follows we describe our methodological approach which is followed with a discussion of our results and associated policy implications.

Utilising these datasets, the main factors associated with water quality in Irish rivers are assessed using an ordered probit model. It is hoped that providing a comprehensive understanding of the variety of human economic activities that influence water quality will be an important tool in the management of risk and will allow for more appropriate land use planning aimed at restoring and maintaining water quality as required by the WFD. The paper continues as follows: Section two outlines the variety of datasets used in this analysis. Section three provides an overview of the ordered probit methodology used in the modelling process. Section four continues with a discussion of the results from the ordered probit model. Finally, this paper concludes with a discussion of its main findings and their implications for land use planning.

2. THEORETICAL FRAMEWORK

Until the 1950s, many European water standards were based on dilution. For example, no treatment was required if 1 part of untreated sewage was diluted by 500 parts of receiving water flow. Today, the quality status of water bodies and their pollution is understood in a more comprehensive manner, expressed as integrity, which can be either physical or chemical. Physical integrity implies habitat conditions of the water body that would support a balanced biological community. Chemical integrity refers to chemical composition of water and sediments that would not be injurious to the aquatic biota (Novotny, 2003).

The quality of water in a particular water body determines the possible use of the water. Water quality is defined as the physical, chemical and biological characteristics of water and is usually described in terms of certain criteria and standards (Diercing, 2009). Novotny states that the environmental water quality criteria and standards currently used throughout the world are either stream (ambient) or effluent (emission) (Novotny, 2003). Water quality depends on the local geology and ecosystem and human activities can negatively affect water quality (Donoghue et al., 2006; O'Donoghue et al., 2010). It is a complex issue and thus assessed on a number of indicators ranging from water temperature and pH and a set of chemical components like heavy metals to biological and microbiological parameters (Novotny, 2003).

For each intended use there are many parameters expressing water quality (USEPA, 2006). Both single compound (e.g. biochemical oxygen demand (BOD), ammonia, nitrate, dissolved oxygen, phenol etc.) and multiple compound parameters (oil and grease, whole effluent toxicity, coliforms etc) are used (Novotny, 2003). EPA Ireland lists 101 principal parameters of water quality (EPA, 2001). Drinking water indicators include alkalinity, colour of water, pH, odour, dissolved metals, salts, microorganisms, metalloids, dissolved organics, radon, heavy metals, pharmaceuticals, hormone analogs, and other substances (EPA, 2001). These are also important in assessing environmental water quality in general and fall into three groups of environmental indicators; 1) physical indicators; 2) chemical indicators; 3) biological indicators.

Physical indicators assess water temperature, conductivity, total suspended solids (TSS), transparency or turbidity, total dissolved solids (TDS), odour of water, colour of water and the taste of water (Boyd, 2000; Wheeling Jesuit University, 2004). There are many parameters of *chemical assessment* of water quality (EPA, 2001). Among the most important indicators of chemical water quality are pH, biochemical oxygen demand (BOD), chemical oxygen demand (COD), heavy metals, nitrates, orthophosphates, pesticides among others (Boyd, 2000; Wheeling Jesuit University, 2004). *Biological assessment* is one of the most common methods of assessing the ecological condition of streams and is based on the biodiversity of the stream, specifically on the presence and abundance of members of insects such as mayflies (Ephemeroptera), caddisflies (Trichoptera) and the more ecologically sensitive stonefly (Plecoptera). These indicator species are usually site-specific and vary from region to region (Metcalfe-Smith, 1996; Boyd, 2000; Wheeling Jesuit University, 2004).

This paper is concerned with ambient water quality and status as defined by the WFD (Directive 2000/60/EC). WFD directs the Member States to conduct an assessment of the water body status and to establish of the classification schemes for 1) biological, 2) hydromorphological (quantity and dynamics of flow, inputs to and from groundwater, presence or absence of impediments to fish movement (river continuity), depth and width, structure of river substrate and riparian zone), 3) chemical and physico-chemical quality elements, including establishing chemical and physico-chemical standards (: a) *General components (physico-chemical)*, e.g. dissolved oxygen, nutrients and temperature ;b) *Specific relevant pollutants* are those identified by Member States as being discharged in significant quantities, e.g. metals; c) *Priority substances* (also includes the dangerous substances) (EPA, Ireland, 2007). The *biological quality elements* grouping comprises four specific elements: phytoplankton, macrophytes, invertebrates, fish. In addition, for each surface water body, the ecological status must be identified as: high; good; moderate; poor; bad.

When assigning the water quality status the lowest status assigned to either the biological quality element, general components (physico-chemical), and hydromorphological elements or failure to achieve the standards set for the specific relevant pollutant will determine the ecological status that can be assigned to the water body. This determines the Q value assigned to a water body (Table X) (EPA, Ireland, 2007).

Table 1. The EPA scheme of Biotic Indices ('Q Values') relates to Water Framework Directive status categories.

Biotic Index (Q)	Status	Boundary EQR value
Q5, Q4-5	High	High /Good = 0.85 Good/Moderate = 0.75
Q4	Good	
Q3-4	Moderate	
Q3, Q2-3	Poor	
Q2, Q1-2, Q1	Bad	

Source: EPA, Ireland, 2007. The intermediate values (Q1-2, 2-3, 3-4 etc.) denote in-between conditions.

One of innovations of the WFD is that unlike previous regulations it requires Member States to manage water bodies on a river-basin scale, which is a natural hydrological and geographical unit (Blöch, 2004). This would require co-operation on European and international level in cases where a river basin lays on the territories more than one country. Brian Moss (2008) states that almost anything that happens on the catchment affects the water. He even compares the relationship between catchment and receiving water to that of a house and a waste been (Moss, 2008).

In order to describe unwanted changes to water resources the word pollution is often used and refers to poor water quality. At its most inclusive, the term pollution can be used to describe all unwanted environmental effects of human activity. Pollution happens where the defined standards for water quality are exceeded.

There are numerous sources of pollution. On the basis of the sources of pollution a distinction is made between diffuse pollution and point source pollution. Where the specific source or location can be identified, pollution is described as point source. In the case of diffuse pollution, the source of pollution cannot be readily identified as it originates from air, land surface, and subsurface zones and from drainage systems and results from the interaction between weather events and the landscape. The pollution from these sources is difficult to monitor and control (Ritter, 2001; Lally et al., 2009). Novotny (2003) states that diffuse pollution is often a result of use and misuse of land, and the causes of the pollution are mostly socioeconomic; encouraged by tradition, government subsidies, foreign demands for cheap products, and lack of information about pollution and polluting behaviour (Novotny, 2003). Tietenberg (2006) identifies the following main sources of non-point pollution: runoff from streets, agriculture, atmospheric deposition, forest management and industrial disposal systems (Tietenberg, 2006).

Urban Pollution and Street Runoff

Urban pollution is one of the leading sources contributing to pollution in streams and the wider environment. The increasing population of the Earth is putting more and more pressure on ecosystems each year. When rural areas are urbanised this causes dramatic changes in the environment in general and the local hydrology in particular. Changes in the hydrology include increasing runoff and decreasing evapo-transpiration as well as deep and shallow infiltration through increasing imperviousness of the surface and the channeling of runoff into gutters. This results in increased runoff volume and velocity and decreased recharge of aquifers directing the runoff to surface water bodies and increasing the risk of flooding. The waste from population and traffic density, commerce, production, as well as pets accumulates on impervious pavements – roads, sidewalks, parking lots, driveways, rooftops etc – and is washed off into storm gutters or directly into streams during rainfall. Other problems caused by imperviousness of the urban areas include proportional increase in flooding and deterioration of habitat; urban stream-bank erosion as a result of increased flow volume and velocity; siltation and thermal shocks

(Novotny, 2003; USEPA, 2003; Waters et al. 2011). Ellis and Mitchell (2004) report that urban runoff may “prejudice” “good” ecological potential of the water bodies.

In water quality assessments, the principal focus has been on examining the impact of the agricultural sector. There is, however, a wide variety of economic activities that can have a significant negative impact on water quality. One recent study which did attempt to examine the effect of a broader variety of factors on river water quality was by Donoghue et al., (2006). In this Irish study, the effect of residential density as well as agricultural intensity on the ecological quality of water was examined. Linking catchment characteristics and water chemistry with the ecological status of 797 hydrologically independent rivers throughout Ireland, Donoghue et al. found that both human settlement (in terms of urban land use and by extension, population density) and agricultural activities (in terms of pasture/arable land use and animal stocking density) were related to water quality. Goldar and Banerjee (2004) also assessed the impact of a diverse range of factors on water quality in India. This study found that industrialization, irrigation intensity and fertilser use were all negatively associated with water quality.

Septic Tanks

Another important source of diffuse water pollution is septic tanks. The scale of the septic tank problem has led to the introduction of the EU Waste Framework Directive (2006/12/EC). One of the features of Irish culture and landscape is a large number of dispersed farms and rural dwellings that form a widespread rural area. This usually means that these households are not connected to sewage treatment systems and have individual septic tanks installed. When septic tanks are installed and managed incorrectly the ground water pollution may be a consequence resulting in a large number of untreated chemicals and bacteria entering the environment (Swarup et al., 1992). Ireland has over 400,000 septic tanks throughout the territory, which contribute to the problem of the water pollution (EC, Ireland, 2011).

Soil Erosion

Soil Erosion is a natural process that results from natural land denudation and depends on the potential of the rainfall and/or wind to erode the soil surface (erosivity). However, there are also sediment loads into the aquatic systems that are a direct result of the human land-use activities (mainly construction and land ploughing) (Boyd, 2000; Merrington *et al*., 2002; Novotny, 2003).

Agriculture as a Source of Pollutants.

One of the most debated of all sources of diffuse pollution is agricultural activities. There are a number of reasons for the debate. On one hand, it is hard to determine when agricultural activities become polluting activities. On the other hand, there is an opposition of goals - to feed Earth’s rapidly growing population and the need to achieve improvements in water quality under the Water Framework Directive (Directive 2000/60/EC). There are many factors that determine the probability of pollution from agricultural land – the types of land use, crop, animals, land-use management etc. The land-use activities that are more likely to negatively affect water quality are dry-land cropland, irrigated cropland, pastureland, forestland, confined animal feeding operations, aquaculture and nurseries. There are a number of pollutants that commonly originate from agricultural land: nitrogen, phosphorus, pesticides, pathogens and sediment (Merrington et al., 2002; Novotny, 2003)

In addition to the sources of diffuse pollution, it is useful to note that this type of pollution has a significant stochastic component that depends on fluctuations in weather and other environmental factors (Olmstead, 2010). This is in line with a previous study by Donoghue et al. (2005) who established that seasonal rain and catchment morphology variations have significant effect on nutrient variation in Irish rivers. Schulte et al. (2006) conducted a study on interactions between agriculture, meteorology and water quality in Ireland. Their study confirmed that there are regional differences in nutrient loss attributed to differences in soil characteristics and rain intensity and/ or quantity variations. They also noted substantial inter-temporal variation in agro-meteorological conditions (Schulte et. al., 2006).

Given the multiple potential sources of poor water quality, it is important to examine the effect of a diverse range of factors on the ecological quality of water resources. Previous studies have often been limited in scope often focusing on the impact of changes in one particular sector (generally agriculture) or being focused on one specific river catchment when evaluating the determinants of water quality. This paper adds to this literature by combining a number of spatial datasets relating to agricultural, residential and industrial activities to determine the major factors affecting water quality throughout Ireland. More specifically, data from the EPA water quality monitoring stations throughout the country are combined with the 2000 Irish census of agriculture which provides spatial information relating to agricultural activity, the 2002 Small Area Population Statistics (SAPS) which also provides spatially referenced information on septic tank and population density data and finally forestry cover data from the forest service in a Geographical Information System (GIS) framework.

3. METHODOLOGY

In this study, we will utilse the EPA Q-value system, which uses an index from 1 to 5 to assess the ecological quality of water resources at each monitoring point. This results in an ordinal dependent variable that takes on five discrete values (5 means higher water quality status than 4, which means a higher status than 3, and so on). However, it is unlikely the distance between each of the categories will be constant. In other words, it may take a bigger change in an independent variable to get over the "threshold" into one category than it takes to get into the next category. An ordered probit model estimates both the effects of the independent variables (through the systematic component) and the thresholds of the dependent variable (through the stochastic component) at the same time.

Characteristics such as physical land use, population densities and economic activity of the river catchments, denoted X_i, determine the level of water quality, denoted Y_i, at the monitoring points in each catchment. The subscript i indicates the i^{th} water quality monitoring point, i = {1, …, n}. *Yi* is a scalar that takes the values of 1, 2, 3, 4 and 5. Larger values indicate higher water quality. *Y* is an (n x 1) vector indicating the water quality level at each monitoring point. The i^{th} element of the vector indicates the i^{th} water quality monitoring point's level. X_i is a vector with *k* elements. The letter *k* indicates the k^{th} independent variable, *k = {1, …, K}*. *X* is an (*n x k*) matrix summarizing each river catchments economic and land use characteristics. The nth row indicates the characteristics of the nth catchment. Therefore, we can state that:

$$Y_i = f(X_i) \quad \forall \quad i = 1, \cdots, n$$

Since the dependent variable is an ordered, qualitative variable, we estimate the relationship between *Y* and *X* with an ordinal response model. Assume that the level of water quality in a river catchment, denoted Y_i*, is a continuous function of catchment characteristics, denoted X_i, a vector of parameters of dimension *(k x 1)*, denoted β, and a disturbance term, ε, which is normally, identically, and independently distributed, $\varepsilon \sim N(0, \sigma^2)$. Increasing values of Y_i* indicate an increasing level of water quality associated with that river system.

$$Y_i * = \beta' X_i + \varepsilon$$

However, the EPA water quality data only records the categorical level to which the monitoring point belongs. The probabilities of falling into ordered Q-value categories, 1 to 5 are given by the following:

$$\Pr(Y_i = 1) = \Phi(\mu_1 - \beta' X)$$

$$\Pr(Y_i = 2) = \Phi(\mu_2 - \beta' X) - \Phi(\mu_1 - \beta' X)$$

$$\Pr(Y_i = 3) = \Phi(\mu_3 - \beta' X) - \Phi(\mu_2 - \beta' X)$$

$$\Pr(Y_i = 4) = \Phi(\mu_4 - \beta' X) - \Phi(\mu_3 - \beta' X)$$

$$\Pr(Y_i = 5) = 1 - \Phi(\mu_4 - \beta' X)$$

where the μ's are unknown threshold parameters (cut-points) to be estimated with β, and the ranking depends on certain measurable factors *x* and certain unobservable factors Є. Since the disturbances are normally distributed, these probabilities are distributed according to the cumulative normal distribution, Φ. The ordered probit model is estimated using the method of maximum likelihood via the Newton-Raphson algorithm (Long, 1997).

Catchments

4. DATA

In order to model the relationship between water quality and upstream economic activity, we need data in relation to

- Water Quality
- Agricultural Intensity
- Settlement Intensity, specifically the use of septic tank based waste water treatment
- Economic intensity

In this section the data used in this paper and the manner in which the different data sources were combined in a GIS framework is outlined. These datasets include the EPA water quality monitoring (Q-value) data, spatially referenced industrial activity and septic tank distribution data from the small area Census of Population, levels of agricultural activity from the Census of Agriculture, and forest land cover data from the Forest Service.

The EPA Water Quality Classification System

In Ireland, the Quality Rating System has been used to monitor the ecological quality of streams and rivers since 1971 (Flanagan and Toner, 1972; McGarrigle et al., 2002). The connection between Q-values and orthophosphate concentrations in rivers has previously been used as the basis of national legislation with a view to controlling eutrophication in Irish waters (DELG, 1998). One further advantage of the Quality Rating System by the EPA is that it has established links with a number of specified elements in Annex V of the Water Framework Directive (Donohue et al., 2006).

Over 3000 sites on some 13,200km of main river channel are included in the current national survey and assessed using the Quality Rating System to characterise water quality (EPA, 2008). The Quality Rating System is a method whereby a Quality-index is assigned to a river or stream based on macroinvertebrate data, but also takes into consideration aquatic macrophytes and phytobenthos. The possible scores (Q-values) range from 1, indicative of extremely poor ecological quality to 5, indicative of minimally impacted conditions (i.e. pristine/unpolluted). Such a compression of biological information inevitably results in a loss of meaningful information; however such a classification is essential if this information is to be meaningfully represented within an economic framework.

The connection between Q-values and orthophosphate concentrations in rivers has previously been used as the basis of national legislation with a view to controlling eutrophication in Irish waters (DELG, 1998). One further advantage of the Quality Rating System by the EPA is that it has established links with a number of specified elements in Annex V of the Water Framework Directive (Donoghue et al., 2005). The Q-values from a set of 2548 river sites that were monitored by the Irish Environmental Protection Agency in 2005 were analysed in this study. Where a mid point was used in rating the Q values for certain monitoring points the lower value was applied in the model presented later, i.e. if for example the rating was given as 1-2 rather than 1 or 2 then a value of 1 was taken for that monitoring point for the purpose of this analysis. Plotting the trend in the three year moving average share of Q values that are moderate or worse (values 1-3) over time, we note the general downward trend over time in Figure 2, from around 20% in the early 1990's to around 17% post 2005.

In table 2, we report the distribution of Q-values for the years of Census data we use in this paper. This reflects the overall trend seen in figure 1. Figure 1 however masks a decrease of the share with the best water quality (High). Thus we see an increase in the concentration of Good water quality, increasing to 68.4% in 2011. It should be noted that these reflect the districts that contain agricultural data. Q-value points that occur in catchments with only urban catchments are not included in this analysis.

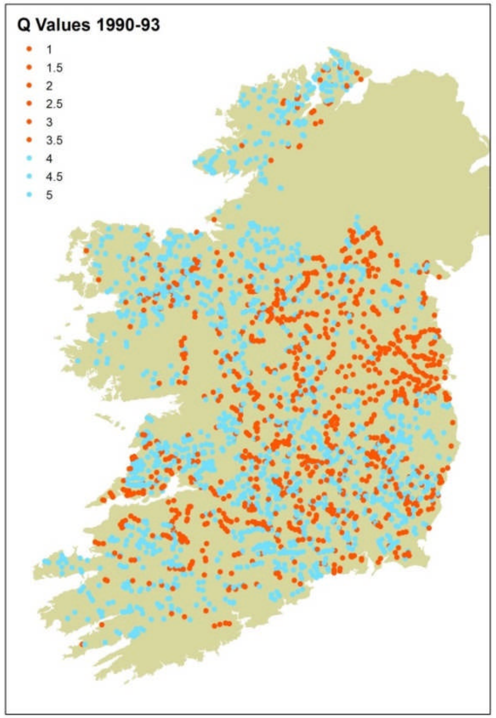
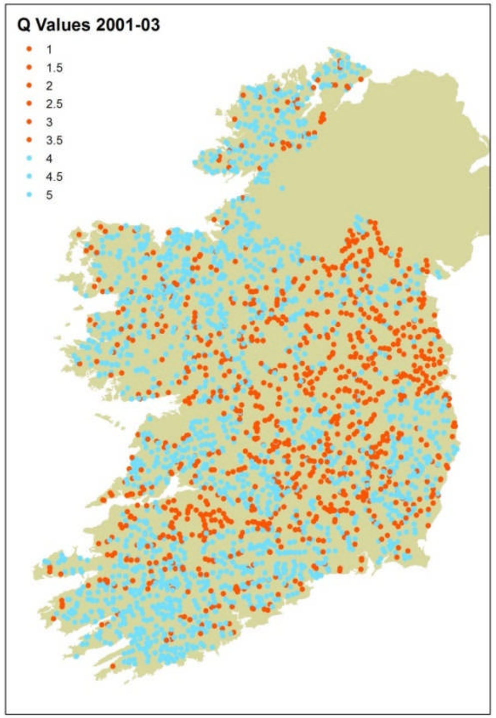
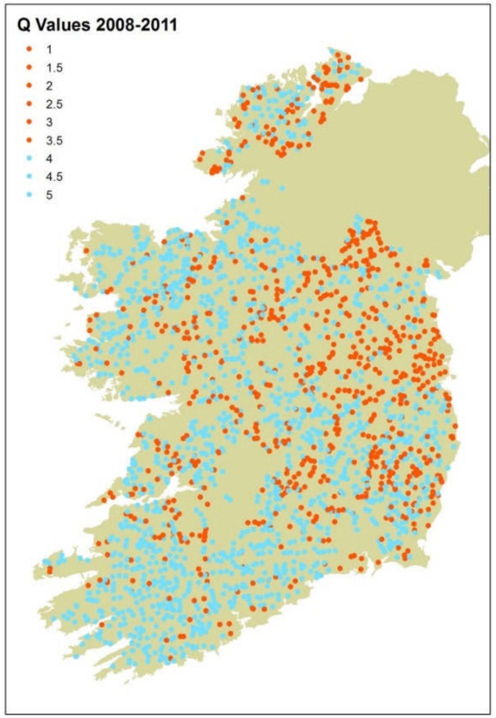
Figure 2 Share of Q Values 1-3

Note: 3 year moving average, reflecting sampled q-value points

Table 2 Share of Q-Values per year

QV	1991	2002	2011
1	1.9	0.2	0.7
2	7.1	4.0	2.0
3	16.4	16.07	14.2
4	37.7	56.7	68.4
5	26.9	23.0	16.6

Figure 3 Poor and Satisfactory QV Values at times of Agricultural Census (1990-2; 2001-3; 2008-2011)

1990-1993	2001-2003	2008-2011
		

In figure 3, we report the spatial incidence of Q-values 1-3, water quality points with unsatisfactory values. Visually, we see a shift to the North and East over time, with a reduction in the share of unsatisfactory points in the South and West. We see an increase in points in the commuting area around Dublin and in coastal towns and estuaries. We note continued concentrations in East Donegal, West Limerick and in the border area, reflecting intensive pig, poultry and other agricultural activity in combination with specific soil characteristics.

The Irish Census of Agriculture

The second dataset used in this paper is the Irish Census of Agriculture. The objective of the census is to collect data relating to agricultural activities on all farms within Ireland (CSO, 2002). The census classifies farms by physical size, type and geographical location. A key requirement in determining a geographic assessment of the respective contribution to water pollution from a sectoral perspective is the availability and resolution of spatial data pertaining to these sectors. In Ireland, the lowest level of spatial disaggregation for publicly provided data is at the Electoral Division (ED)[1] level. Of the 3,440 Electoral Districts in the country, 2,850 contain farms; the average number of farms in each of these ED's is 53 (min 10, max 320). We utilise data from the 1991, 2000 and 2010 Censuses of Agriculture. We combine these below with the Censuses of Population that are closest for 2000/2002 and 2010/2011. However as we find significant auto-correlation in the data, this does not greatly affect our results.

The specific variables from the census of agriculture used in this analysis include the proportion of farmland in each ED under crops, the number of pigs per hectare in each ED and finally livestock density in each ED. The main source of diffuse pollution from grassland based sectors such as livestock rearing come from the release of large amounts of nitrous oxide. The main sources of nitrous oxide are: nitrogen fertilsers and manure and urea deposited by grazing animals (Monteny et al., 2006). The figures for livestock density were combined with Irish EPA conversion factors for different livestock types to produce an estimate of organic nitrogen produced per ED. Whereas livestock production in Ireland is extensive in nature, pig farming tends to be more localized and intensive. As such a separate variable representing the intensity of pig production was included in the analysis. The final agricultural related variable utilized in this analysis was the intensity of cereal production. In contrast to grassland based farm activities, cereal production requires much larger applications of chemical fertilizers with higher concentrations of phosphorous and potassium.

Approximately 90% of agricultural land in the Republic of Ireland is in grassland. The main source of diffuse pollution from a grassland based production system is due to incidental loss of nitrogen and phosphorus. The main source of these nutrients to land in agricultural production is through chemical fertiliser applications and organic manure spread or deposited by grazing animals. In this analysis, livestock numbers were combined with the organic N conversion factors (as per EU nitrates based Good

[1] Formerly known as District Electoral Division (DED). The term Electoral Division was changed on 24 June 1996 (Section 23 of the Local Government Act, 1994).

Agricultural Practice Regulations, Government of Ireland, 2010 S.I. No. 610 of 2010) for different livestock types to produce an estimate of organic nitrogen produced per hectare per ED. This was adopted to reflect the impact of livestock based production at the ED level. Whereas livestock production in Ireland is extensive in nature, pig and poultry production tends to be more localized and intensive. Separate variables representing the intensity of pig and poultry production was included in the analysis based on the number of total pig and poultry (birds) at the ED level. The final agricultural related variable utilized in this analysis was the intensity of cereal production. This variable is based on the percentage of land in the ED in arable production. Cereal production requires larger applications of chemical fertilizers than other types of agricultural activity. Additionally, ploughing and crop sowing activity can disturb soil and has the potential to increase sediment (and associated phosphorus) entering a watercourse.

Forestry Cover Data

To provide information on the level of forest cover within each ED a land cover classification for Ireland developed by Teagasc under the Forest Inventory Planning System and Irish Forest Soils (FIPS–IFS) project was used. The FIPS–IFS land-cover data set was developed using GIS and remote sensing, along with ground-truthing provided by field sampling. The mapping unit employed in the FIPS–IFS land-cover data set was 1 hectare. The main class in the FIPS–IFS land-cover data set that we include in our analysis is a combined variable for mature forestry and immature forestry and scrub. This forest cover GIS data has been updated by Farelly, (2007) to reflect spatial changes in forestry cover in Ireland in recent years. The forest cover data used in this paper therefore is representative of forestry in Ireland in 2005. In terms of water quality one might expect the level of forest cover in a catchment to contribute to measured water quality either positively by acting as a filter or negatively if there is active forestry felling or ground preparation taking place, thus leading to sediment erosion and nutrient runoff. We utilize two variables in our model, afforestation in a particular year and cumulative afforestation.

In terms of water quality one might expect the level of forest cover in a catchment to contribute to measured water quality either positively by acting as a filter or negatively if there is active forestry felling or ground preparation taking place for new forestry, thus leading to sediment erosion and nutrient runoff. Forestry can also contribute to the artificial acidification of run-off waters if the planting has occurred on acid-sensitive soils.

Census of Population, Small Area Population Statistics (SAPS)

The Central Statistics Office (CSO) as part of the National Census of Population collect data pertaining to the structure and services to residential dwellings in Ireland including the number of rooms per house, toilet facilities, internet connections and sewerage facilities in each ED. In relation to sewerage facilites, the EPA (2006) found that the presence of septic tanks, which are the main method for wastewater treatment in rural households, have a significant negative impact on water quality and therefore a variable representing the proportion of households in each ED that have septic tanks was included in the analysis.

It was also thought useful to include a measure of economic activity within each ED. The SAPS dataset classifies all workers within each ED under eight industry types: Agriculture, Forestry and Fishing, Manufacturing, Construction, Commerce, Transport and Communications, Public Administration, Education, Health and Social work or Other Industry. This allows one to quantify the number of workers within each industrial category in each ED. A variable representing the proportion of all workers belonging to each of these industrial categories was included in the ordered probit model. By combining the agricultural, forestry and census data described above with the associated Q values for the EPA monitoring stations, it is possible to examine the major economic factors affecting river water quality. To this end, an ordered logit model is developed where the dependent variable is river water quality as measured by the Q-value index.

In the relevant Census years, we see that there are respectively 345900, 407768, 437652 houses with septic tanks in 1991, 2002 and 2011, with the density increasing by 20% over the period of the study, concentrated in the first 11 years.

Landfill Site Data

Historically the Republic of Ireland had a heavy reliance on landfills for the disposal of waste products. Landfills in Ireland were brought under the regulatory control of the Environmental Protection Agency under the Environmental Protection Agency Act, 1992 and the Waste Management Act, 1996 (EPA, 2011a). The 1999 Landfill Directive (Council Directive 99/31/EC) was a major milestone in the regulation of landfills in the EU, as it specified the technical requirements for landfill design, operation, closure and aftercare. It was later supplemented by a Council Decision which specified the criteria for the acceptance of waste at landfills (EPA, 2011a). Landfills pose a potential risk to water quality due to leachate. Compliance with legislation is assessed by the EPA through the completion of site inspections and audits and monitoring of emissions and the quality of the environment. The location and co-ordinates of landfill facilities in Ireland was secured from the EPA based on their licensing regime (EPA, 2011b). A total of 77 landfills had a license to operate in 2011 and a variable was then constructed to indicate if a landfill facility was within 3 kilometres upstream of a Q value monitoring site.

Environmental Data

Environmental and physical characteristics of watersheds can also play an important role in water quality (Donohue et al., 2005; 2006). To account for this in the model a series of spatial variables were created that describe the soil, geological and climatic characteristics of watersheds. For example, bedrock data from the Geological Survey of Ireland (GSI) (1:100,000 bedrock shapefile (GSI, 2013)) and soil data from the Teagasc EPA soil and subsoil map (Fealy et al., 2009) were employed to attain data on geological and soil characteristics. A digital elevation model (DEM) for Ireland at a 25m resolution was employed to derive a series of elevation related variables. A slope map was generated from the DEM at the same resolution. Climactic data were derived from models developed by Sweeney and Fealy (2003). Polygon based data were intersected with the ED shapefile to derive the area of soil and bedrock categories in each ED. For raster data the average, median, maximum, minimum and range were calculated across each ED.

Septic Tanks

According to the Central Statistics Office (CSO, 2012), on-site domestic waste water treatment systems collect, treat and discharge waste water from almost 500,000 households in Ireland. Previous studies have found that the presence of septic tanks, which are the main method for wastewater treatment in rural households, can have a significant negative impact on water quality (Clabby et al., 2008; Macintosh et al., 2011; Curtis and Morgenroth, 2013). The European Court of Justice found that Ireland had not met legal obligations to regulate the waste water generated in our unsewered areas as required by the 1975 Waste Framework Directive (75/442/EC). In response to this finding, the National Inspection Plan for Domestic Waste Water Treatment Systems was set up. Under this plan, risk-based inspection of septic tanks and other on-site treatment systems will be carried out. This means that inspection will be concentrated in areas where waste water discharges present a high risk to human health or the environment (EPA, 2013). To examine the effect of septic tanks on river water quality, using small area population statistics (SAPS) we derive a variable indicating the number of septic tanks per km2 in each ED.

Catchment Delineation

River sub-basins maps were used to establish the relevant upstream area for each water quality monitoring point. Each monitoring point was joined to the river sub-basin that it fell within. Shapefiles describing the monitoring stations and the river sub-basins were attained from the EPA (http://gis.epa.ie/GetData/Download). Thus, a dataset relating Q-values from individual monitoring stations to the characteristics of the relevant river sub-basin and related upstream EDs was created. Upstream was determined based on elevation. ED's with centroid elevation greater than the centroid elevation of the ED in which the Q-value monitoring point was recoded was deemed upstream and average upstream values were derived for the analysis.

Summary Statistics

We now report some summary statistics in the data. Table 3 describes the average Organic N per hectare by Q-Value point. Generally there is an inverse monotonic relationship between organic N per hectare and Q-value between Q-value 5 and 3. However the relationship is not clear between 3 and 1. At all Q-values, we see steady decline in the density. This reflects in the latter period a general decline in animal numbers. In the earlier period, the number of cattle increases, but this is offset by a decline in the number of dairy animals which have a higher organic N per hectate coefficient in addition to a substantial decline in the sheep population.

Table 3 Average Organic N per Q-Value 1991-2011

QV	1991	2002	2011

1	118.1	104.9	108.0
2	113.7	108.4	90.7
3	118.4	113.4	93.6
4	111.0	103.9	91.4
5	106.0	96.1	81.6
Total	113.1	107.0	92.0

Table 4 describes population density by Q-value. In 1991, the relationship was inverse monotonic; with the population density increasing as the Q-value fell. However in 2002 and 2011, Q-value 2 exhibits a decline in the population density relative to Q-value 3. This is particularly the case for 2011; albeit the cell size is very small. What is noticeable is that while the population density of the Q-value 5 areas has declined as High water quality has become concentrated in the most remote areas with the very lowest population densities, the population density for the worst water quality are has risen dramatically, reflecting a relationship primarily with urban centres. Overall, reflecting the rise in the population from 3.5 million in 1991 to 4.6 million in 2011, the density has risen over time.

Table 4 Average Population Density per Q-Value 1991-2011

QV	1991	2002	2011
1	543.3	1170.7	1496.5
2	429.6	510.8	440.8
3	313.4	533.0	853.9
4	281.4	233.5	300.9
5	147.2	90.6	91.0
Total	288.4	341.0	475.8

In table 5, we report the average septic tank density over time. The relationship is more or less inverse monotonic. Overall the septic tank density has risen. This growth has primarily occurred in areas with Q-values of 4 and 1 and 2. This growth has occurred differentially for the worst water quality areas.

However the relationship is starker for septic tanks than for organic N per hectare. In 2011 (1991), the septic tank density for Q-value 1 was 6.3 (2.9) times that of Q-value 1, while the organic N per hectare ratio was 1.32 (1.11). Meanwhile the ratios for Q-value 3 to Q-value 1 for respectively septic tank density was 1.7 (1.64) and 1.15 (1.12) in 2011 (1991). Thus for those areas with Q-value 1, the situation would seem to have gotten worse for both input measures over time, albeit with a declining share of areas with this level. This is particularly the case for septic tanks, reflecting areas with higher population density.

Table 5 Average Septic Tank Density per Q-Value 1991-2011

QV	1991	2002	2011
1	84.8	147.8	173.3
2	57.4	43.5	78.6

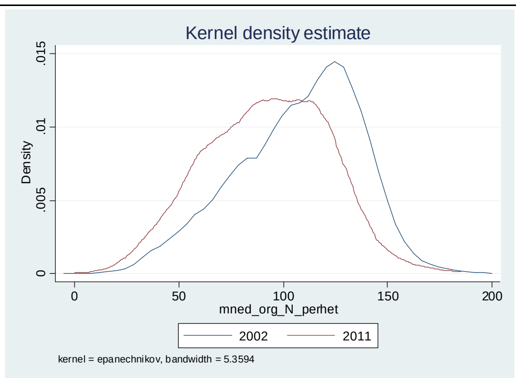
3	47.8	58.2	46.3
4	35.1	40.5	47.2
5	29.1	23.7	27.3
Total	39.7	46.3	47.5

In Figure 3, we report the kernel density functions of the distribution of organic N per hectare and the septic tank density. While the mean organic N per hectare decreased between 1991 and 2002, the distribution did not change significantly with small increase in spread.[2] The change is much more dramatic during the period 2002 to 2011 with a greater increase in spread and a clear reduction in mean. For septic tank density, the change in the distribution is not significant over time, with most of the changes occurring at the extremes which is not picked up in the graphed kernel density as we truncated the top to more easily see the mode.

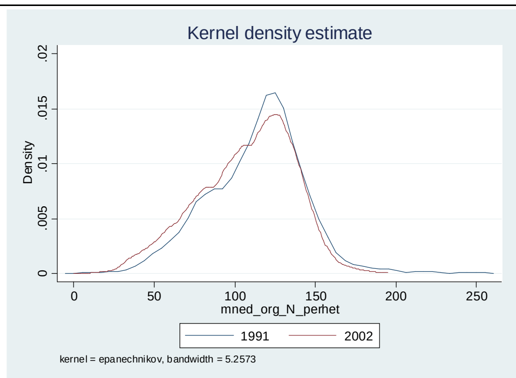
Figure 4 Kernel Density Functions

Organic N per hectare

2002-2011

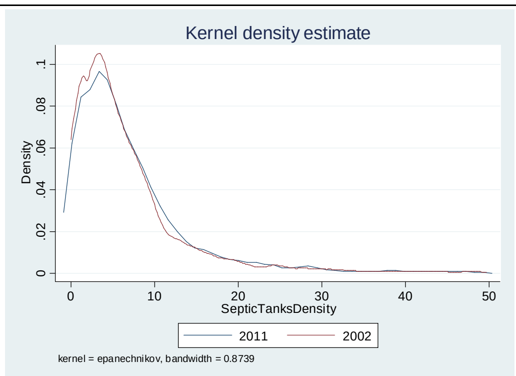


1991-2002

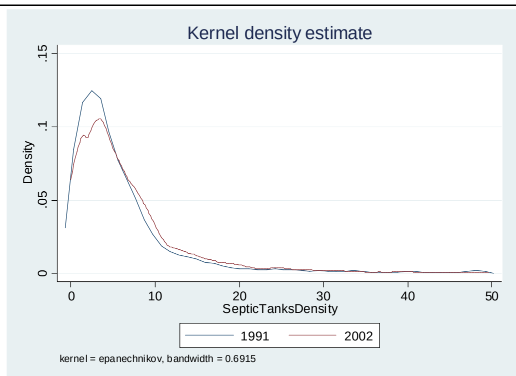


Septic Tanks per Ha

2002-2011



1991-2002



5. Results

[2] It should be noted that when we refer to 2002 (2011), that Census of Agriculture numbers refer to 2000 (2010)

In this section we report the order probit results for our models describing the relationship between the explanatory variables and the water quality variables. In our analysis, we utilise 3 models:

1. A basic model capturing the underlying relationship with the main explanatory variables, septic tank density and organic N per hectare

2. A model incorporating time trends with the main explanatory variables

3. A more complex model incorporating environmental variables

The primary coefficient estimates and associated standard errors for the chosen model specification are then presented in Table 6. The ordered probit analysis was conducted to determine the major factors affecting the ecological quality of water sources measured at the EPA water quality monitoring points in each river catchment.

As the dependent variable, the Q-value, is categorically ordered, an ordered probit model was utilised. This takes the explanatory variables and estimates the probability of being in each category of water quality status (1 to 5).

The coefficients of the ordered probit model indicate whether the explanatory variables are positively or negatively related to improved levels of water quality status (Long, 1997).

In model 1, the density of septic tanks is statistically significant at the 1% level and works in the anticipated direction; the higher the density of septic tanks in the relevant ED's the lower the value of the Q-value index at the monitoring point in the river catchment; reflecting the summary statistcs above. In Ireland, wastewater from a significant proportion of the population (generally in rural areas) is treated by small-scale on-site systems (septic tanks) where connection to a sewer is unfeasible. The results in Table 5, model 1 would suggest that this system is unsustainable if goals in relation to water quality are to be achieved.

Grassland based farm enterprises, namely beef, sheep and dairy production dominant Irish agriculture and account for approximately 80 percent of overall agricultural output value. Large amounts of nitrous oxide from animal manure as well as urea are deposited by grazing animals on the land in these farm enterprises (Monteny et al., 2006). The results in Table 5 would suggest that the quantity of organic nitrogen produced per hectare in the associated ED's is statistically significant (at the 1% level) and negatively associated with measured Q-values. Therefore, in line with other research the results suggest that the more intensive the farm livestock rearing enterprise is, as measured by organic nitrogen production per hectare or the intensity of pig production, the lower the likelihood of achieving a higher Q value.

Model 2 interacts these variables with time dummies. Here the coefficients without interactions retain the same signs and maintain significance. For septic tanks, the relationship significantly worsened between 1991 and 2002. However for the period 2002, there was no significant change in the relationship. For organic N per hectare, there is no statistical difference between 1991 and 2002. However from 2002-2011, the relationship becomes statistically better.

In Model 3, we incorporate more Agricultural and Industrial sectoral variables as well as environmental variables, whose coefficients are reported in table 7 and soil variables whose coefficients are reported in table 8 in the appendix. Overall the fit of the model improves significantly through the addition of these contextual variables. However the underlying conclusions of model 2 in terms of the inter-temporal relationship between sepctic tanks and Agricultural organic N per hectare remain largely the same.

Table 6. Ordered Logit Regressions

	Model 1			Model 2			Model 3		
	Coef.	S.E.	p-value	Coef.	S.E.	p-value	Coef.	S.E.	p-value
Septic Tank Density	-0.00041	8.83E-05	0	-0.00029	0.000143	0.041	-0.000079	0.000153	0.606
Organic N Density	-0.00347	0.000386	0	-0.0056	0.000656	0	-0.00184	0.000891	0.039
Year Interaction									
Organic N Density x 1991				0.000731	0.000949	0.441	-0.001507	0.00108	0.163
Organic N Density x 2011				0.004512	0.000997	0	0.002793	0.001139	0.014
							-0.053834	0.098415	0.584
Year Dummy									
2002				-0.08055	0.10836	0.457	-0.359155	0.133955	0.007
2011				-0.56565	0.108456	0	-0.593915	0.134631	0
Landfill within 3km							-0.256719	0.12962	0.048
Industry Share									
Industry							-0.053834	0.098415	0.584
Construction							0.058886	0.112262	0.6
Commerce							-0.267241	0.117826	0.023
Transport, storage and communications							0.266624	0.183477	0.146
Public administration							0.06682	0.230851	0.772
Education, health and social							-0.08639	0.123345	0.484
Other							-0.245077	0.174633	0.161
Pigs per Ha							4.24E-06	3.96E-06	0.284
Poultry per Ha							-5.82E-07	5.41E-07	0.281
Afforestation at time t							-3.41E-06	0.001471	0.998
Cumulative Afforestation							0.000458	0.00023	0.047
Pseudo R2	0.0045			0.0074			0.0918		
N	7941			7941			7441		

We find for example that the location of a landfill site within 3km to be associated with poorer water quality. The share of particular industrial sectors in general does not have any statistical relationship.

Areas with a higher share of commerce, a largely white collar office based sector, has a negative relationship. A negative relationship between population density and water quality has been widely reported (Donoghue et al., 2006) and it could be that as most Commercial employment is in urban areas then this variable is capturing the effect of population density. Thus this has more to do with the association with urban centres than an explicit relationship with the pollution of the sector.

We do not find any relationship with afforestation in the period of the Q-value data collection. Rather as forestry tends to be planted in poorer remoter land, that areas with higher accumulations of forestry tend to have better water quality. The sparse spatial and intertemporal nature of the measurement points however cannot pick up local impacts of plantation and harvesting. Also the period of our study ignores the large scale afforestation that occurred before 1991. Subsequent plantations largely occurred in small blocks within farms. This result is consistent with Novotny (2003) who suggests that undisturbed forests or woodland represent the best possible protection for land from sediment and pollutant losses. Woodlands and forests have low hydrologic activity, due to high surface storage in leaves (interception), ground, mulch, and terrain roughness. Novotny (2003) also points out that even lowland forests with a high groundwater table (containing wooded wetlands) absorb large amounts of precipitation and actively retain water and contaminants.

In table 7, we report the relationship with environmental variables. The most significant are slope and elevation which are associated with remoter areas. The X (Y) coordinate coefficient is negative (positive), indicating poorer water quality is poorer in the East and South, where the population centres are and where Agriculture is concentrated. Conditional on X,Y coordinates rainfall is not significant. Temperature is negatively significant at the 10% level reflecting poorer water quality in the South.

We note that the time gap between water quality measurement (we use the value most recently after the relevant Census date) is not statistically significant reflecting the strong inter-temporal auto-correlation in water quality meausures.

Table 7. Environmental Coefficients in Model 3

	Coef.	S.E.	p-value
Area of District	-0.001416	0.000956	0.139
Environmental Characteristics			
Rainfall	0.00011	0.000158	0.486
Average Temperature	-0.094336	0.049556	0.057
Median Elevation	0.003696	0.000439	0
Mean Slope	0.122289	0.01243	0
Time between River Quality Measurement and Census	-0.01083	0.009213	0.24
Coordinates			
x_coord	-3.74E-06	8.4E-07	0
y_coord	3.02E-06	8.49E-07	0

Note: Soil Shares variable coefficients reported in the appendix

6. DISCUSSION

This paper undertook an exploratory data analysis concerned with determining the effect of both agricultural and non-agricultural economic activities on the ecological quality of water resources. To achieve this aim, a number of spatial datasets relating to agricultural, residential and industrial activities as well as the level of forest cover were combined within a GIS framework. Results indicate that septic tank density, and variables related to agricultural activity such as the level of organic nitrogen per hectare, the proportion of land used for the growing of cereals and intensity of pig farming were all negatively associated with water quality. In addition, the numbers of construction and public administration workers in each ED were also negatively associated with water quality. It could be hyothesised that these variables are capturing the effect of construction activity and population density on water quality. One final variable included in this analysis was the degree of forest cover which was found to be positively associated with water quality.

In relation to the agricultural sector, this analysis would suggest that the intensity of farming has a significant negative impact on water quality which is supportive of previous work discussed earlier (see Fezzi et al., 2008; Cuttle et al., 2006 and Haygarth et al., 2003). Given the strong association between agricultural activity and water quality it has been widely reported that the agricultural sector will need to undergo significant structural change if WFD requirements are to be met. Some of these suggested changes include reductions in the use of fertilizers and a reduction in sheep and cattle stocking rates.

Recent policy changes to the CAP could conceivably lead to a much lower level of agricultural activity (Oglethorpe, 2005; Osterburg and von Horn, 2006). More precisely, under the mid term review (MTR) of the Common Agricultural Policy (CAP) in 2003, member states within the EU agreed to implement a system of single farm payments (SFP) which were decoupled from production (Ackrill, 2008). Under this new system, farmers are paid a lump-sum cash payment based on historical payments, whereby actual production is not needed to receive support. The move towards decoupling of payments can be seen as significantly reducing the incentive for farmers to produce. This disincentive could, in turn, lead to a significant reduction in the intensity of farming practices (Howley et al., 2010a). For instance, it has been estimated that as a result of the move towards decoupling, the numbers of suckler cows and sheep will fall by 25 and 42 per cent respectively between 2005 and 2020 in Ireland (see Howley et al., 2010b). That said, while the agricultural sector is set to undergo significant structural changes as a result of recent changes to the CAP it is unlikely that these changes will be enough to fulfil requirements under the WFD.

In relation to the residential sector, it is also clear that the main option available for rural households when it comes to treating waste, namely septic tanks, is having a significant negative effect on the ecological quality of water resources. It is interesting to note that the effect of a 1 percent change in septic tank density on water quality was two times greater than a similar reduction in organic nitrogen associated with livestock density. The analysis presented here would suggest that appropriate forest management can have a beneficial impact on the ecological quality of water resources. Benefits such as open access recreation have often been put forward as a non-market benefit of forests and this analysis would suggest that benefits in relation to water quality could be one further advantage of good forest management.

To sum up, the analysis presented in this paper highlights the important relationship between land use and water quality. In particular, the level of forestry, construction activity, population density, the intensity and type of agricultural activity and the type of wastewater treatment in an area are all critical factors affecting the quality of our water sources. Moreover, the results highlight the importance of a spatial dimension to any analysis as the principal factors affecting water quality will often differ across river catchments. It is clear from this analysis that no one sector is responsible for adverse water quality and in turn the solution will depend on a multi-sectoral approach aimed at addressing the multitude of factors affecting water quality. In this regard, it is hoped that the analysis provided here will be an important tool in the management of risk and will allow for more appropriate land use planning aimed at restoring and maintaining water quality as required by the WFD.

The timing of the study, relying primarily on 2000 Census of Agriculture Data may have a specific impact on the results as there have been likely changes in the period since 2000. Lalor et al. (2010) report a reduction in soils with excessively high levels of P over that period; At the national level, P fertiliser use has declined by 6 kg ha-1 (55 %) for grassland and 5 kg ha-1 (16-30 %) for arable crops between 2003 and 2008. The proportion of tested soils with excessive P (Index 4) has declined from 30 % to 22 % between 2007 and 2011 (Lalor et al., 2010), falling to 18% in 2012. Research on Teagasc’s Agricultural Catchment Programme (ACP) has shown that on 5 catchments, between 6 and 26 % of soils had excessive P status, showing the legacy of historic P surpluses (Wall et al., 2012). Large spatial variability was found at farm and field scale, indicating scope to correct imbalances with better nutrient management.

Later data that is now available from the 2010 Census of Agriculture could usefully add to our understanding of these processes. However it may be difficult to observe the impact of the reductions in high levels of P in the period since 2007 as there are significant lags in relation to the impact of changes on farm to changes in water quality. Schulte et al. (2010) et al. used a ‘Soil P Decline’ model to evaluate this expectation for 4 ACP catchments. At a field P deficit scenario of -7 kg P ha^{-1} it was predicted that an average of between 5 and 20 years would be required for all Index 4 soils to reach index 3.

Paradoxically there is a concern that more recently below optimum, with increases in the proportion of land with Phosphorous Index 1 and 2 increasing from 40% in 2007 to 59% in 2012, which will lead to reduced farm level productivity (Shortle, 2013).

River water quality is affected by a combination of geomorphological (e.g. soil type, slope, elevation), climatic (e.g. precipitation) and anthropogenic factors (e.g. agricultural practices, forestry, landfills and septic tanks). Understanding how anthropogenic and natural factors affect water quality and how the relationships change over time will help water resource managers to target efforts aimed at improving river water quality. One advantage of this work is that by combining a number of spatial datasets we were able to simultaneously examine, at the national level, the effect of different drivers, of river water quality.

In relation to the agricultural sector, globally our results in line with the existing literature in that various agricultural activities such as livestock, cereal and pig production were found to have a significant negative effect on river water quality. It has been widely reported that if statutory obligations in relation to water quality are to be met then significant changes need to be undertaken by the agricultural sector throughout Europe (Haygarth et. al., 2003; Bateman et al., 2006). As such, much recent research has investigated the effectiveness of various farm management mitigation measures for alleviating harmful impacts on water pollution. Within livestock enterprises, it has been found that N loss can be mitigated by changes in manure storage and manure application strategies (Chambers et al., 2000). For example, Lalor et al. (2011) reports that 9% more N is available for plant uptake from manure if it is spread in spring as opposed to summer and up to 10% more N is available if manure is spread by using a trailing shoe as opposed to a splash plate. Livestock dietary manipulation has also been shown to improve N use efficiency by animals, reducing N excretion and hence its entry to the wider environment (van Groenigen et al. 2008; Luo et al., 2008). Finally the use of cover crops has been shown to be very effective in terms of reducing N losses (Hooker et al., 2008). Landowner options to reduce Phosphorous run-off into water bodies include optimizing fertilizer P use-efficiency, refining animal feed rations, using feed additives to increase P absorption by the animal, applying manures to soils with a nutrient deficit and targeting conservation practices where they can be effective such as cover crops, buffer strips and adaptive management of critical source areas of P export from a watershed (see Sharpley et al., 2000 for a review). Notwithstanding the significant negative effect of agricultural activities on river water quality, it is important to note that our analysis indicates that this effect has significantly reduced over time. This could be a reflection of reduced level of production as well as a variety of policy programmes and measures, such as cross compliance obligations and good agricultural practice regulations introduced in response to the EU Nitrates Directive.

The EPA has identified leachate management from landfill sites as a regulatory challenge facing the sector and a priority area. Results from this analysis would support this view as an active landfill site upstream of a monitoring station was associated with lower Q value outcomes. This suggests additional management or engineering solutions might be required to address this situation along with increased monitoring and enforcement. In Ireland and in the UK, wastewater from a significant proportion of the population (generally in rural areas) is treated by small-scale on-site systems (septic tanks). The design of many septic tanks reflect the historical legacy of the wastewater disposal infrastructure prevalent during periods when regulations and environmental awareness were not so rigorously defined or implemented. Due to problems in relation to poor design, management (leaks, lack of emptying) and being inappropriately sited (e.g. close to a watercourse), it has been widely shown that septic tanks can have a negative effect on water quality (Withers et al., 2011). Our analysis would support this view and validates policy initiatives that have been introduced in recent times to address this pollution source. These initiatives require homeowners to register and monitor the effectiveness of their septic tank disposal system.

To conclude, our analysis illustrates how river water quality is affected by a combination of natural and anthropogenic factors, the relative influences of which change over time. No one sector is responsible for adverse river water quality and in turn the solution will depend on a multi-sectoral approach aimed at addressing the multitude of factors affecting water quality. In this regard, it is hoped that the analysis provided here will be an important tool in the management of risk and will allow for more appropriate land use planning aimed at restoring and maintaining water quality as required by the WFD.

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APPENDIX SOIL COEFFICIENTS FOR ORDINAL LOGIT MODEL 3.

Table 8. Coefficients on Soil Variables in Model 3

	Coef.	S.E.	p-value
Soil Type			
Wind-blown sands undifferentiated	0.00157	0.000609	0.01
Mineral alluvium	0.002002	0.000464	0
Marl type soils	0.001111	0.000701	0.113
Derived from mainly non-calcareous parent materials	0.00078	0.000457	0.088
Derived from mainly non-calcareous parent materials	0.000869	0.000458	0.058
Derived from mainly non-calcareous parnet materials	0.001263	0.000465	0.007
Derived from mainly non-calcareous parent materials	0.00133	0.000607	0.028
Derived from mainly non-calcareous parent materials	-4.41E-05	0.000603	0.942
Predominantly shallow soils derived from non-calcareous rock or gravels with/without peaty surface horizon	0.000669	0.000456	0.142
Derived from mainly non-calcareous parent materials	0.001269	0.000466	0.006
Blanket peat	0.000892	0.000458	0.052
Derived from mainly calcareous parent materials	0.000768	0.000457	0.093
Derived from mainly calcareous parent materials	0.000641	0.00046	0.164
Derived from mainly calcareous parent materials	0.001048	0.000492	0.033
Derived from mainly calcareous parent materials	0.000941	0.000603	0.119
Derived from mainly calcareous parent materials	-0.00123	0.001423	0.387
Predominantly shallow soils derived from calcareous rock or gravels with/without peaty surface horizon	0.000711	0.000797	0.372
Derived from mainly calcareous parent materials	0.000806	0.000462	0.081
Cutaway/cutover peat	0.000836	0.000456	0.067
Fen peat FenPt	0.000607	0.000623	0.33
Lacustrine–type soils	-0.000128	0.000692	0.853
Made/Built land	-0.000645	0.00056	0.249
Beach sand and gravels	0.000934	0.000768	0.224

Marine/ Estuarine sediments	0.001127	0.000631	0.074
Raised bog	0.029392	0.023307	0.207
Scree	-0.000128	0.000511	0.802
Water (including lakes, reservoirs and larger rivers)	0.000175	6.15E-05	0.005
Basalts & other Volcanic rocks	-0.001153	0.00113	0.307
Cambrian Metasediments	-4.51E-05	0.000104	0.665
Devonian Kiltorcan-type Sandstones	-0.001274	0.001116	0.254
Devonian Old Red Sandstones	0.000115	5.73E-05	0.044
Dinantian (early) Sandstones, Shales and Limestones	-0.001234	0.001111	0.267
Dinantian Dolomitised Limestones	-0.001299	0.001113	0.243
Dinantian Lower Impure Limestones	-0.001258	0.00111	0.257
Dinantian Mixed Sandstones, Shales and Limestones	-0.00106	0.001111	0.34
Dinantian Pure Bedded Limestones	-0.001249	0.00111	0.26
Dinantian Pure Unbedded Limestones	-0.001166	0.00111	0.294
Dinantian Sandstones	-0.001151	0.001112	0.3
Dinantian Shales and Limestones	-0.001088	0.00111	0.327
Dinantian Upper Impure Limestones	-0.001273	0.001112	0.252
Granites & other Igneous Intrusive rocks	0.000153	6.63E-05	0.021
Namurian Sandstones	-0.001392	0.001112	0.211
Namurian Shales	-0.001471	0.001112	0.186
Namurian Undifferentiated	-0.001371	0.001111	0.217
Ordovician Metasediments	-2.87E-05	6.69E-05	0.668
Ordovician Volcanics	-4.33E-05	8.47E-05	0.609
Permo-Triassic Mudstones and Gypsum	0.001921	0.003553	0.589
Precambrian Marbles	-0.001089	0.001114	0.328
Precambrian Quartzites, Gneisses & Schists	1.41E-05	6.53E-05	0.829
Silurian Metasediments and Volcanics	0.000025	6.82E-05	0.714
Westphalian Sandstones	-0.001689	0.00116	0.14

		9	8
Westphalian Shales	-0.001226	0.00111 8	0.27 3
Calcareous	0.000367	0.00119 8	0.75 9
Non-calcareous (Siliceous)	-0.000976	0.00046 1	0.03 4