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The impact of forestry as a land use on water quality outcomes: An integrated analysis

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ABSTRACT

The adoption of the EU land use, land use change and forestry (LULUCF) regulation ensures that for the first time afforestation in Europe will contribute toward the achievement of European Union (EU) climate change commitments under the Paris Agreement. However, increased afforestation in Europe could have unintended environmental trade-offs that may hamper the achievement of EU Water Framework Directive targets. While much of the previous forestry research has focused on the potential negative impacts of afforestation and harvesting processes on water quality at a single point in time, this study applies an ordered probit model to investigate the impact of afforestation and forest cover (in a predominantly agricultural setting) on water quality over a 20-year period. In addition, we present an analysis of a simulated increase in afforestation and forest cover, and a corresponding decrease in agriculture area, on water quality. The results show an increase in water quality in 2.62% of cases. Both increased forest cover and the substitution of livestock have a positive impact on water quality outcomes. Despite the negative impacts associated with the process of afforestation, the long term positives associated with forest cover over the course of a forest rotation, make it a preferable land use option in terms of water quality relative to more seasonal agricultural land uses. Given the expected increase in afforestation in line with national policy, Ireland offers a unique opportunity to observe the outcomes of a large scale afforestation programme in a rural setting. The findings of this paper offer a deeper insight into the impacts of afforestation and forest cover over a meaningful time frame that is not available in site specific studies and studies focused on individual management interventions.

1. Introduction

Afforestation is a widely recognised climate mitigation strategy (Smith et al., 2014). In 2018, the EU adopted the land use and land use change and forestry (LULUCF) regulation to govern the inclusion of greenhouse gas (GHG) emissions and removals from the LULUCF sector toward climate mitigation targets (Forsell et al., 2018). The regulation also provides that member states' emissions from LULUCF should not exceed removals, also known as the 'no-debit' rule. For the first time, the sequestration potential of afforestation in Europe will contribute toward the achievement of EU Paris Agreement commitments. However, less attention has been given to the potential water quality trade-offs resulting from the process of afforestation.

The Water Framework Directive (WFD) requires EU Member States to achieve 'good ecological status' and 'good surface water chemical status' in all surface waters by 2015 (or subsequent cycles) (Council Directive, 2000).). Water quality outcomes are influenced by a range of land use and catchment characteristics (Donohue et al., 2005; Doody et al., 2012; Withers and Haygarth, 2007), the most well documented being agriculture and independent wastewater treatment systems, such as septic tank systems (STS) (Haygarth et al., 2003; Novotny, 1999; Richards et al., 2016; Tong and Chen, 2002). Afforestation defined by the IPCC as the "planting of new forests on lands which, historically, have not contained forests" (IPCC, 2006), and forest management interventions, such as forest harvesting, have been linked with negative water quality outcomes (Clarke et al., 2015; Kelly-Quinn et al., 2016; Rodgers et al., 2012).

In 2015, forest cover (32.6%) and agriculture (41%) combined, accounted for over 70% of EU Member States land cover (Eurostat, 2017). On average, 20% of the rural EU population are not connected to

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a wastewater treatment plant (OECD, 2018). Between 2005 and 2009 only 44% and 56% of rivers and lakes in the EU reached satisfactory condition (EEA, 2012). In Ireland, the Environmental Protection Agency (EPA) reports reductions in national water quality outcomes resulting from nutrient losses from agriculture, discharges from urban and private wastewater, forestry and other extractive industries (EPA, 2017). To the best of the authors' knowledge, to date, impact analysis of these land uses in an Irish context has only been considered in isolation, rather than as part of a holistic rural land use analysis.

The objective of this study is to assess the impacts of private afforestation and other rural land uses over time at the national scale in Ireland between 1991 and 2012. In addition, this study also aims to model the ex-post impact of land use change on water quality outcomes, utilising a dataset that combines information relating to agriculture, STS, afforestation, forest cover and EPA water quality. To meet EU and national emissions reduction targets to 2030, Ireland has a land use policy targeted at substantially increasing forest cover through private afforestation (DAFM, 2015). Recent analysis of GHG emissions in Irish agriculture have indicated that increasing afforestation to 10,000 ha per annum and rewetting organic soils in agriculture could yield 1.4 $MtCO_2e$ of sequestration annually (Lanigan et al., 2018). This study analyses the water quality impacts of a simulated increase in afforestation and forest cover, and a corresponding decrease in agriculture.

1.1. Context

1.1.1. Ireland & the Irish forest policy context

The Irish rural landscape is dominated by agricultural production, particularly livestock production. Agricultural area makes up 63% of total land area, and nearly 90% of this it utilised for livestock production (CSO, 2018). Intensive livestock production results in significant sustainability challenges in terms of water quality and GHG emissions reduction (Gerber et al., 2013; Weil and Brady, 2002). Ireland's climate is particularly suited to tree growth, which is almost twice the rate of the European average (O'Connor and Kearney, 1993). However, Ireland remains one of the least forested countries in Europe. In the most recent National Forestry Inventory (NFI) 2015 Ireland's forest cover was 11%, with conifer stands accounting for much of the forest estate (71.2%) (DAFM, 2018a, b). Between 1921 and the 1980s, afforestation was largely undertaken by the Irish State. From the 1980s, private afforestation, largely on former agricultural land, has been encouraged through financial incentives with virtually no state afforestation from 2000 onward (Ryan, 2016). Current Irish forest policy seeks to increase forest cover to 18% by 2046 through private afforestation (DAFM, 2014). Private afforestation is incentivised through state funded afforestation grants providing landholders covering 100% of establishment cost along with 15 annual premium payments (DAFM, 2015; DAFM, 2018a, b). However, despite generous incentives, afforestation rates have consistently fallen short of national targets (DAFM, 2018a, b).

1.1.2. EU 'no-debit' rule

The LULUCF regulation on the inclusion of GHG emissions and removals from the LULUCF in the EU 2030 Climate and Energy Framework was adopted in 2018 (EU 2018). The Framework targets a sector wide emissions reduction of 40% by 2030 compared with 1990 as a part of Paris Agreement commitments (Forsell et al., 2018; UNFCCC, 2015). Each Member State must ensure that emissions do not exceed removals in all of the land accounting categories, and must submit their National Forest Accounting Plans and proposed Forest Reference Level for the periods 2021 to 2025 and from 2026 to 2030 (EU 2018, Forsell et al., 2018). Accounting produces 'debits' or 'credits' (increased or reduced emissions) that count toward the achievement of climate mitigation targets. The aim of this system is to provide incentives for further action and policies in terms of climate mitigation, while creating disincentives for detrimental action (EU 2018). In short, forest carbon sinks must not decline beyond the proposed Forest Reference Level and sequestered carbon counts toward reduction targets.

1.1.3. Water framework directive

The EU WFD was adopted in 2000 with the aim of maintaining 'high status' of waters where it exists, prevention of deterioration in existing status, and achievement of at least 'good status' in all waters by 2015 (or subsequent cycles) (McNally, 2009). The WFD provides a structure for the protection of groundwater, surface waters, estuarine, and coastal waters. Ecological water quality is measured in five quality classes using a combination of biological quality elements, such as the macroinvertebrate fauna, macrophyte flora, fish communities, the supporting general physico-chemical conditions and hydromorphology (EC, 2011; EPA, 2015). The main parameter measured is the sensitivity of macroinvertebrates to pollution. The assignment is based on the departure of invertebrate populations from reference (pristine or high status) conditions (geographic, typological and temporal) (EPA, 2015).

2. Theoretical background

2.1. Factors impacting rural water quality

Fig. 1 highlights the main factors that impact rural water quality. Wastewater discharge, agriculture and forestry are the primary determinants of water quality in the Irish context (EPA, 2018). Sources of pollution to waters are classified into point and diffuse sources (Carpenter et al., 1998). STS are classified as point source and potentially easier to measure and regulate than diffuse sources. Agriculture on the other hand can result in diffuse sources of pollution. Afforestation may also result in diffuse pollution as a result of disturbance to soil during forest establishment and early forest management operations. These diffuse sources are more difficult to control and monitor, making analysis more difficult. In examining pressures on water quality over time, we hypothesise that pressures relating to agriculture, STS and afforestation are associated with negative water quality outcomes.

2.2. Waste water discharge

The common usage of STS in rural areas is a significant contributor to rural water quality outcomes (Dudley and May, 2007). The effectiveness of STS in containing nutrient load is largely dependent on the quality of installation (Gill et al., 2009; Withers et al., 2012). A study of STS in Belgium found that 52% of the sample did not meet legal effluent standards (Moelants et al., 2008). In Northern Ireland, a study classified over 40% of the sampled STS as posing a high risk to water quality (Arnscheidt et al., 2007). A significant proportion of the population in Ireland relies on STS (over 25%) (CSO, 2017).

2.3. Agriculture

The loss of sediment and nutrients, mainly Nitrogen (N) and Phosphorous (P), from agriculture are serious global problems (White et al., 2009). Stoate et al. (2009) assert that agricultural impacts on water quality arise from processes such as cultivation, application of inputs (fertilisers, pesticides) and drainage. These processes result in physical, chemical and biological changes in downstream waters, which can impact aquatic ecosystems (Stoate et al., 2009). Nitrates leached from soils pose a risk to the environment and human health (Jarvis et al., 2011; Weil and Brady, 2002).

Losses of P via, livestock grazing, tillage, and application of animal manure and chemical fertiliser also have negative environmental impacts (Weil and Brady, 2002). Losses can occur when soils become saturated resulting in overland flow, where poor soil infiltration coupled with heavy rainfall results in transportation of nutrients over ground surfaces (Carton et al., 2008). In Ireland, up to 80% of P losses occur during October to February due to intensive rainfall (Kiely et al., 2007).

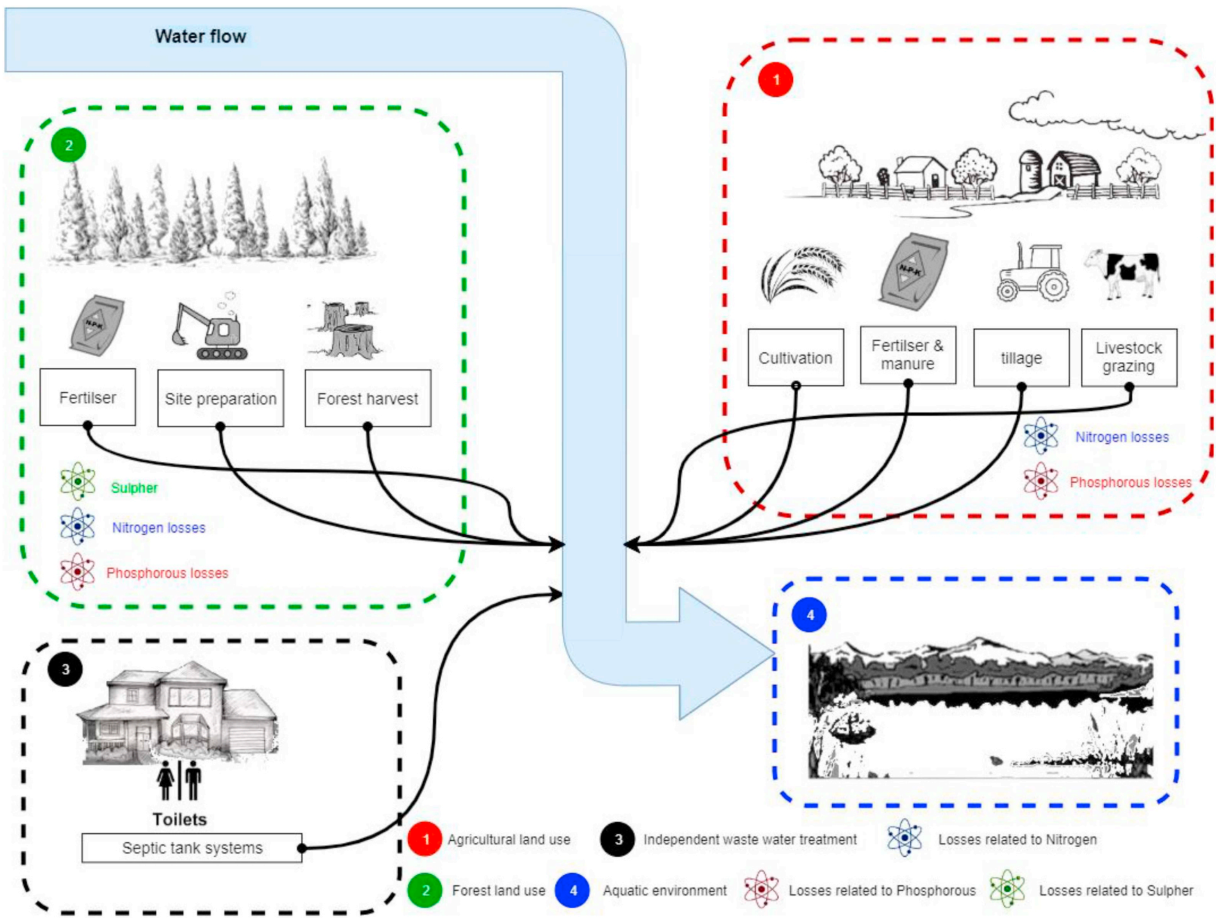

Fig. 1. Main factors influencing rural water quality outcomes.he Authors wish to acknowled

2.4. Forestry

Nutrient run-off can result from the processes of afforestation and management of forest stands (Clarke et al., 2015; Drinan et al., 2013). Afforestation practices, such as initial site preparation (mounding, drainage etc.) and fertiliser application, increase nutrient transfer (Binkley et al., 2004; Clarke et al., 2015). Road construction increases surface runoff and sedimentation (Arnaez et al., 2004), while compacted tracks and drainage ditches provide a route for runoff (Clarke et al., 2015). Further, forest harvesting results in nutrient and sediment loss (Clarke et al., 2015; Piirainen et al., 2007; Sundström et al., 2000). The removal of trees also reduces absorption capacity and interception by the tree canopy increasing nutrient loss (Birkinshaw et al., 2014; Clarke et al., 2015).

Short rotation conifers offer significant advantages in terms of wood yield and carbon sequestration (Kanowski, 1997) and many species perform well in less nutrient-rich soils, but require nutritional inputs to do so (Farrelly et al., 2009). The application of P fertilisers to such soils poses a significant risk to watercourses (Drinan et al., 2013; Rodgers et al., 2010). Further, conifer stands are associated with water acidification due to uptake of airborne pollutants such as Sulphur and N, which can increase water acidification (Drinan et al., 2013).

However, evidence suggests that forest cover can confer benefits with regards to water quality (Bauhus et al., 2010; Townsend et al., 2012; van Dijk and Keenan, 2007). Increased tree cover can offer protection from rainfall, nutrient and sediment loss through the development of a litter layer, understory growth and surface roughness provided by tree roots (Bauhus et al., 2010; van Dijk and Keenan, 2007). Further, fast growing species and dense forests retain more water than slower growing species (van Dijk and Keenan, 2007). Though nutrients are typically only applied on less fertile sites at initial afforestation, the application and resulting loss is typically lower than agriculture (May et al., 2009).

Water quality is vulnerable to both agricultural and forest land use systems, however, there is a significant temporal difference. Forest impacts associated with management practices occur infrequently over the duration of the forest cycle (30–50 years) (Clarke et al., 2015), while agricultural interventions (cultivation, application of inputs etc.) are generally seasonal in nature (Hooda et al., 2000; Stoate et al., 2009).

3. Methodology

3.1. Empirical approach

The dependent variable in this analysis is the EPA water quality Q-value data and uses an index of 1 to 5 to assess the ecological quality of water at each of the EPA monitoring stations. The categories in the variable are ordered, taking 5 discrete values. The distance from one category to the next is not constant as a larger change in an independent variable may be required to cross the threshold of one category than to cross to the next category. By using an ordered probit model it is possible to estimate the impacts of independent variables (systematic component) and the thresholds of the dependent variable (stochastic component) at the same time. Analysis of the data was conducted using the Stata statistical package.

The characteristics within river catchments (denoted X_{it}) determine water quality outcomes at each monitoring point (denoted by Y_{it}) in each catchment at time (t). The subscript it indicates the ith water quality monitoring point, i = {1….n}. The years 1991, 2002 and 2011 are represented by t. The scalar (denoted by Yi) takes on values of 1 to 5. As values increase, so does water quality to a maximum level of 5. Xit is a vector with k elements. The letter k indicates kth independent variable, $k = \{1....k\}$. X is an $(n \times k)$ matrix summarising each of the river catchments economic and land use characteristics. The nth row indicates the characteristics of the nth catchment.

$$Y_{it} = f(X_{it})\ \forall\ i = 1, \ldots, n$$

As the dependent variable is an ordered, qualitative variable, the relationship between Y and X is estimated using an ordinal response model. The level of water quality in a river catchment $(Y_{it}{}^{*})$ is assumed to be a continuous function of catchment characteristics (X_{it}) underlying the ordered probit model, a vector of parameters of dimensions $(k\ x1)$, denoted by β and a disturbance term (ε_{it}), which is normally, identically and independently distributed $\varepsilon \sim N(0, \sigma^2)$. Increasing values of Yit^* indicate an increasing level of water quality associated with that river system.

$$Y_{it}^{*} = \beta' X_{it} + \varepsilon_{it}$$

The ordered probit model is estimated using the method of maximum likelihood via the Newton-Raphson algorithm (Long, 1997). The panel format of the data (based on periods 1991, 2002 and 2011) necessitated using a panel estimator. The use of a random effects ordered probit allows us to take into consideration the existence of an additional normally distributed cross-section time invariant error term $N(0, \sigma_v{}^2)$, denoted by (ν_i). The error term in the random effects model (ε_{it}) can be is assumed to be of a composite nature:

$$\varepsilon_{it} = \nu_i + \omega_{it}$$

The term ν_i is the unobserved, time invariant, individual specific heterogeneity. This random effect is assumed to be unrelated to any of the catchment characteristics (X_{it}) in our model. The term ω_{it} is the idiosyncratic error representing unobserved factors that change over time (Wooldridge, 2016). The use of a random effects model allows for the inclusion of time invariant variables. As such, dummy variables for soils, year and county are included to capture time and place specific effects.

The EPA water quality data only records the categorical level that the monitoring point belongs, with values 1 to 5.[1] In this case

$$Y_{it} = \begin{cases} 0\ if\ Y_{it}^{*} \le 0 \\ 1\ if\ 0 < Y_{it}^{*} \le \mu_1 \\ 2\ if\ \mu_1 < Y_{it}^{*} \le \mu_2 \\ 3\ if\ \mu_2 < Y_{it}^{*} \le \mu_3 \\ 4\ if\ \mu_3 < Y_{it}^{*} \le \mu_4 \\ 5\ if\ if\ \mu_5 < Y_{it}^{*} \end{cases}$$

In other words:

$$Pr(Y_{it} = 1) = \Phi(\mu_1 - \beta X_{it})$$

$$Pr(Y_{it} = 2) = \Phi(\mu_2 - \beta X_{it}) - \Phi(\mu_1 - \beta X_{it})$$

$$Pr(Y_{it} = 3) = \Phi(\mu_3 - \beta X_{it}) - \Phi(\mu_2 - \beta X_{it})$$

$$Pr(Y_{it} = 4) = \Phi(\mu_4 - \beta X_{it}) - \Phi(\mu_3 - \beta X_{it})$$

$$Pr(Y_{it} = 5) = 1 - \Phi(\mu_5 - \beta X_{it})$$

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 ε. As the disturbances are normally distributed the probabilities are distributed according to the cumulative normal distribution Φ.

[1] Data are also recorded for values that fall between Q level categories. Cut points have been estimated for these intermediary values. For simplicity, the intermediary cut points are not displayed here.

3.2. Land use change simulation

This paper also aims to simulate the marginal impact of land use change from agriculture to forestry (both afforestation and cumulative forest cover) on water quality. This is achieved by altering the land use from agriculture to forestry within a catchment and simulating the consequential impact on water quality using the model of water quality previously specified. In this simulation, we increase forest cover and afforestation by 10% and reduce the agricultural area in each catchment by the amount equal to the increase in forestry, thus simulating the impact of a modest substitution of agricultural area. We assume that the number of animals per ha (stocking rate) remains the same, thus the total number of animals falls.

Given the low number of time periods ($T = 3$), the simulation utilises cut points from the pooled ordered probit model. Once the data have been adjusted we estimate the probability of returning a water quality value of 1 to 5 using a simulated dependent variable. The model we produce can be expressed as:

$$Y_{it}' = f(\beta X_{it} + \varepsilon_{it})$$

Thus the ordered probit simulation effectively estimates the following: the resulting Y_{it}' when X_{it}changes to X_{it}', following a land use change from agricultural land to forestry land. In other words:

$$Y_{it}' = f(\beta X_{it}' + \varepsilon_{it})$$

The model involves a deterministic component $\beta X_{it}'$ and stochastic component ε_{it}'. These two components must be derived separately.

First, we establish the deterministic component. To do this, we simulate a new dependent variable (Z). This is based on the threshold values (μ) for $Y_{it}{}^{*}$ in the pooled ordered probit.

$$Y_{it}^{*} = \mu_1, \ldots, j$$

The parameters (β) are estimated using Stata as are the values of the cut points (μ_i), however the stochastic component is not derived. In order to run the simulation, we generate a series of $\varepsilon_{it}\ 's$, so as to replicate the original dependent variable Y_{it} in a baseline simulation. We observe the original $Y_{it}{}^{*}$, therefore if Y_{it} takes a value of 2 for example, then we need a value of ε_{it} such that

$$\mu_1 < Y_{it}^{*} \le \mu_2$$

or

$$\mu_1 \le \beta X_{it}' + \varepsilon_{it} \le \mu_2$$

or, subtracting $\beta X_{it}{}^{'}$.

The probit index $(\beta X')$ is subtracted from our threshold values (μ) to give the values for Z.

$$\mu_1 - \beta X_{it}' \le \varepsilon_{it} \le \mu_2 - \beta X_{it}'$$

or

$$Z_1 \le \varepsilon_{it} \le Z_2$$

Where ε_{it} in an ordered probit is a standard normal random number.

$$Z_{it} = \mu_i - \beta X_{it}'\ \forall\ i = 1, \ldots, 5$$

At this stage, it is necessary to transform Z so that it returns a probability based on a standard normal distribution with a mean of 0 and a standard deviation of 1 i.e. $Z \sim N(0, 1)$.

$$normal(Z_i) = \int_{-\infty}^{z} \frac{1}{\sqrt{2\pi}} e^{-t^2}/2dx$$

It is now necessary to recover the term $\mathscr{E}$ in our model. In order to do this, we first generate a random uniform variate (a) with an interval of (0,1). We then generate an error term that is equal to the relevant lower bound cut point, plus a multiplied by the relevant upper bound cut point minus the lower bound cut point.

$$\mathscr{E}_{it} = \mu_i^{lb} + a(\mu_i^{ub} - \mu_i^{lb})$$

Lastly, we generate the inverse cumulative standard normal distribution for the error term, $\Phi(\mathscr{E})$.

We then combine Z and predicted $\mathscr{E}$ in order to simulate probabilities for the new dependent variable.

$$Y_{it}' = f\,(\beta X_{it}' + \mathscr{E}_{it})$$

We are now able to determine the probability that Y_{it}' falls into ordered Q-value categories, 1 to 5, based on the threshold values (μ) established for Y_{it}^*. The probabilities of the simulated values for Y_{it}' falling into each category are established by the following formulae.

$$Pr(Y_{it}' = 1) = P(\mu_0 \le Z_{it} + \varepsilon_{it} \le \mu_1)$$

$$Pr(Y_{it}' = 2) = P(\mu_1 \le Z_{it} + \varepsilon_{it} \le \mu_2)$$

$$Pr(Y_{it}' = 3) = P(\mu_2 \le Z_{it} + \varepsilon_{it} \le \mu_3)$$

$$Pr(Y_{it}' = 4) = P(\mu_3 \le Z_{it} + \varepsilon_{it} \le \mu_4)$$

$$Pr(Y_{it}' = 5) = P(\mu_4 \le Z_{it} + \varepsilon_{it} \le \mu_5)$$

3.3. Sensitivity analysis

In a deterministic sensitivity analysis, selected parameters are varied individually in a specified range, while the remaining parameters are held fixed, to establish the sensitivity of results to changes in the specified parameters (Halpern and Pandharipande, 2017; Radaideh and Radaideh, 2019). In this paper, we establish the average parameter values for Electoral Divisions (ED) that have experienced an increase in water quality post land use change simulation. Once these average inputs have been established, we then vary the forest cover, afforestation and organic N density input parameters up to 30%. These parameters are varied one at a time, while the remaining input parameters are held fixed.

3.4. Catchment delineation

The main units of spatial analysis for water quality are river sub-basins. The analysis uses the average value of EDs that fall within the sub-basin boundary and upstream from the Q-value monitoring point. The determination of upstream is based on elevation, thus any ED with a centroid elevation greater than the ED in which the Q-value monitoring point was located were included in the analysis.

Geographic Information System shapefiles (see Figs. 5 to 10 in additional resources) describing the monitoring stations and the river sub-basins were obtained from the EPA, Ireland. The individual monitoring points were joined to the river sub-basins in which they fall. A dataset relating Q-values from individual monitoring stations to the characteristics of the relevant river sub-basin and related EDs was generated following the approach of Howley et al. (2014).

4. Data

This analysis builds on data compiled by Howley et al. (2014). The data include Q-value data, spatially referenced industry data and septic tank distribution from the Small Area Census of Population, agricultural activity data from the Census of Agriculture, forest land cover data from the Forest Service and environmental spatial data from various sources (Howley et al., 2014).

4.1. Description of data & variables

4.1.1. The EPA water quality classification system

The EPA Quality Value (Q-value) scheme has been calibrated in line with the WFD classification and provides a historical record of Irish water quality outcomes from 1987 (EPA, 2015; Toner et al., 2005). Q-value ratings are utilised as the dependent variable in this analysis. The Q-value scheme is the most sensitive ecological assessment method available for detecting organic pollution and nutrient enrichment impacts on Irish rivers (EPA, 2017). Q-value data is taken every five years and a value from 1 to 5 is assigned. When the parameters evaluated have different Q-values for the same water body, the lowest Q-value will determine the ecological status, likewise for intermediate Q-values (e.g. Q 2–3) (EPA, 2015; Toner et al., 2005).

4.1.2. Agricultural & population data sources

Agricultural variables were derived from the Irish Census of Agriculture, which provides data on agricultural activities on farms within Ireland (CSO, 2002). Farms are classified by physical size, type and geographical location. Data from the Census of Agriculture in 1991, 2000 and 2010 are matched with data from the national population census (1991, 2002 2011). To account for the impact of STS and industry, the variables “septic tank density” and “commerce” were included and quantify STS per ha in each ED and the proportion of workers per ED engaged in the commerce industrial category. As census data is collected every 10 years, this data reflects the most recent data currently available.

The lowest level of spatial disaggregation of data was at the ED level. There are 3440 EDs in Ireland. However, not all ED's could be matched with agricultural and population census data for all years. Some agricultural census data was not available for EDs, while data for other EDs was supressed due to the small size of the ED, potentially leading to issues regarding confidentiality. In addition, the number of active water quality monitoring points in those EDs varies from year to year, as such, total observations also vary for the available years. After accounting for missing data, the total observations included in the analysis for the years 1991, 2002 and 2011 was 2530, 2790 and 2177, respectively. The average number of farms per included EDs is 60, with the maximum number of farms being 250. The average utilisable agricultural area per ED was 1755.23 ha. While the average ED area was 3030.72 ha.

In Ireland, over 91% of agricultural area is devoted to grassland (CSO, 2018). This analysis combines livestock numbers with organic N conversion factors (as per EU Nitrates regulations) for the different livestock types to derive the variable organic N density per ha. This was done to reflect the intensity of livestock-based production at the ED level. Cereal production requires the use of inorganic fertilisers and management practices that involve soil disturbance. The variable cereal share of land use is measured based on the proportion of land at the ED level that is used for arable crop production. Lastly, as pig production in Ireland differs from livestock in terms of intensity, a separate variable, pigs per km^2 was derived, based on data from the Central Statistics Office of Ireland.

4.1.3. Forest data

This analysis utilised Geographic Information System data to reflect spatial changes in forest cover in conjunction with a land cover classification for Ireland developed under the Forest Inventory Planning System and Irish Forest Soils project that was aggregated to ED level by Upton et al. (2014) to provide the necessary forest data. This dataset was used to derive the forest cover variables, reflecting the total proportion of land under forest cover per ED, and the afforestation variable, reflecting the proportion of new planting per ED. The available data relate to private grant aided forestry only, meaning that forests established by the State are not captured, however, it should be noted that State afforestation declined from the mid-1980s and effectively ceased from 2000 onward (Ryan, 2016). It should also be noted here that while both afforestation and forest harvesting may have impacts on water quality, activity data for forest harvesting in private forests are not currently available.

4.1.4. Additional data

Biological activity in water bodies is influenced by the environmental and physical characteristics of the watershed (Donohue et al., 2006; Donohue et al., 2005). In order to account for the impacts of environmental characteristics of watersheds, spatial specific variables were created for soil, geological and climatic data. Bedrock data from the Geological Survey of Ireland 1:100,000 bedrock shapefile (GSI, 2016) and soil data from the Teagasc EPA soil and subsoil map (Fealy and Green, 2009) were employed to incorporate geological and soil characteristics of EDs. A digital elevation model for Ireland at a 25 m resolution was used to obtain a series of elevation variables. A slope map was generated from the digital elevation model at the same resolution. Climactic data were derived from the 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.

5. Results & discussion

5.1. Results I: summary statistics

In examining changes in river water quality over time, the distribution of Q-values for the population census years (1991, 2002, 2011) is presented in Table 1. Values that fall between classification categories have been included. A progressive reduction in the lower water quality values can be observed (Q-values 1, 2 and 3), as can an increase in Q-value 4. However, there is also a decline in higher status Q-values 4–5 and 5.

In Table 2, summary statistics of the explanatory variables included in the analysis are presented. Septic tank and organic nitrogen density are calculated using the mean density per ED. Cereal share of land use, afforestation share of land use, and forest cover share of land use relate to the proportion of land utilised for cereal, forest cover, and afforestation, respectively. Commerce and blanket peat refer to the proportion of workers in an ED and the proportion of the ED area that is blanket peat. Forest cover IQR is a dummy variable derived from the Forest Cover share of land use variable. To derive the Forest cover IQR, Forest Cover share of land use is divided into four categories, which represent the interquartile range of forest share of land use. Environmental characteristics represent the average values of rain, temperature, elevation and slope in millimetres, degrees Celsius, metres and degrees, respectively, per ED.

5.2. Results II: pooled & random effects ordered probit

For the purposes of comparison, pooled and random effects ordered probit models were estimated. The pooled model ignores the time invariant individual specific effects accounted for in the random effects ordered probit. The results of the pooled (Model 1) and random effects (Model 2) ordered probit for the ecological quality of water sources are presented together in Table 3. The explanatory variables represent the weighted average of the variables in the EDs, or the proportion of area of the EDs upstream of the water quality monitoring site and in the water catchment area. In order to avoid potential issues regarding collinearity among the independent variables, a variance inflation test was conducted. This produced an average variance inflation factor of 6.28, which is below the threshold value (10) indicating very high correlation (Chatterjee and Hadi, 2012).

Table 1
Percentage share of water quality values (Q-values) for time periods 1991, 2002, 2011.

Q-value	1991	2002	2011
1	0.62	0.07	0.04
1–2	0.15	0.07	0.13
2	0.84	0.90	0.22
2–3	2.15	2.50	1.79
3	14.21	12.51	11.35
3–4	17.06	20.65	19.83
4	38.02	42.24	48.52
4–5	19.58	19.02	16.90
5	7.38	2.04	1.22

Q-value represents the quality of water ranked between 1 and 5, with 5 representing the best, and 1 representing the worst quality. Values that range between Q-values are represented by a range, for example, 1–2 represents Q-values that fall between category 1 and 2.

5.3. Commercial & residential share

Results from model 1 indicate a negative relationship between water quality and STS density. The year interaction is significant in model 1 and 2 indicating a positive outcome relative to the base year (1991). Previous research in Europe highlights the unsatisfactory condition of many of the sampled STS (Arnscheidt et al., 2007). In 2009, Ireland was found to be in breach of EU regulations on wastewater discharge (C188/08, 2009). In 2010, the code of practice for wastewater treatment and disposal systems was published, establishing best practice and incorporating EU guidelines (EPA, 2010). This may account for the more positive outcome in 2011. In addition, findings from model 1 and 2 show a negative relationship between the proportion of workers per ED (commerce) and water quality.

5.4. Agriculture share & environmental characteristics

Variation in climate is accounted for by the inclusion of variables for rain and temperature. Rainfall did not prove to be significant in either model. However, increased temperature in 2011 was negatively associated with water quality outcomes relative to 1991 in both models. Research has shown that water temperature increases can influence the distribution of many aquatic fauna and flora (Dallas, 2008).

The results of both models also indicate significant positive relationships with slope and elevation. Johnson et al. (2008) assert that water quality and drainage are influenced significantly by slope, soil type and geology. Lower anthropogenic pressures at upland sites result in improved water quality (Donohue et al., 2006). Furthermore, ED area (ha) is negatively associated with water quality outcomes, suggesting that larger EDs are more likely to have negative outcomes. In addition, the inclusion of x and y coordinates for the ED centroids highlighted a positive relationship with west and north regions, while south and east were negatively associated with water quality outcomes. This is likely to be due to a greater concentration of water quality pressures in the south and east relative to the north and west (see additional resources for shape files cereal share, organic N density and STS density).

As expected, model 1 indicates a significant negative association between agriculture and water quality. Livestock production (Organic N density) and cereal production (cereal share) are negative in both models. However, once individual specific effects have been accounted for, the relationships are not significant in model 2. Livestock production accounts for a much bigger proportion of agriculture than cereal. In 2013 there was 5 million ha of Utilisable Agriculture Area reported in Ireland. Of this 3.7 million ha was utilised for grassland based ruminant production systems (beef, sheep, dairy production), and 308,000 for cereal production (CSO, 2013). Diffuse sources of pollution are the primary origin of nutrients related to eutrophication (Magette et al., 2007). Agriculture in Ireland is one of the most significant factors influencing water quality outcomes for over half of all waterbodies in Ireland (EPA, 2018).

Results of the time interaction with organic N density make positive water quality outcomes more likely for the year 2011 relative to 1991 in both models 1 and 2. This positive outcome may be explained by the

Table 2
Definitions and descriptive statistics for variables utilised in the Water Quality Model.

Variable	Description	%	Mean	Median	Min	Max	SD
Dependent Variable							
River Quality	EPA river quality classification score	–	3.86	4	1	5	0.58
Agricultural, Commercial & Residential Share per ED							
Septic Tanks	Average Septic tank density per ED	–	44.05	8	0	2529	133.25
Cereal Share of Land Use	Proportion of ED under arable crops	–	0.05	0.004	0	1	0.09
Organic Nitrogen Density	Average organic N density per ha per ED (kgs)	–	104.38	107.92	0	247.24	31.01
Commerce	Proportion of all workers in ED working in Commerce	–	0.16	0.14	0	1	0.12
Forest Cover & Afforestation per ED							
Afforestation Share of Land Use (sq)	Proportion of new planting per ED squared	–	0.03	0	0	5.8	0.03
Forest Cover Share of Land Use	Forest cover proportion per ED	–	0.02	0.05	0	0.2	0.02
Forest Cover IQR	= 0 if no forest cover in ED	30.32%	–	–	–	–	–
	=1 if forest cover proportion in Q1	16.61%					
	= 2 if forest cover proportion in Q2	16.46%					
	= 3 if forest cover proportion in Q3	17.58%					
	= 4 if forest cover proportion in Q4	19.03%					
Environmental Characteristics of EDs							
Rain	Rain in millimetres	–	1034.10	1012.93	121.1	1854.6	247.64
Temperature	Temperature in Celsius	–	10.27	10.2	8.71	13.67	0.73
Area	ED area in ha	–	3030.7	2521.7	28.91	16,331.2	1830.92
Elevation	Elevation in metres	–	103.8	87	0	450	67.03
Slope	Slope in degrees	–	3.56	3	0	18	2.28
Blanket Peat proportion	Proportion of ED that is blanket peat	–	0.11	0	0	0.93	0.20

*Forest Cover *IQR = Interquartile Range of Forest Cover.

implementation of the EU Nitrates directive (Council Directive, 1991/676/EEC). Though the year of inception was 1991, the programme of measures was not realised till 2006 (DAFM, 2018b). These measures were implemented uniformly across Ireland. Thus measures introduced under national regulation to address the management of potential point and diffuse sources of nutrient transfer from agriculture (Buckley, 2012) potentially explain the reduction in effects of organic N.

5.5. Forest cover and afforestation

Consistent with literature that highlights the potential of trees to reduce nutrient and sediment losses (van Dijk and Keenan, 2007), both models show a positive relationship between forest cover and water quality. Analysis of the intensity of forest cover, utilising the interquartile range of forest cover (represented by Forest Cover per ha Quartile, 1–4), also yields a positive relationship across both models at lower density of forest cover.

As expected, the interactions of forest cover with peat and slope produced a negative relationship with water quality across the models. Wetland areas are vulnerable to changes in hydrological conditions brought about by land use changes and climate (Holden et al., 2004). The establishment of conifer species on less fertile peaty sites often requires the application of additional nutrients (Carey, 2006). In addition, slopes within catchments can influence the rate of nutrient export (Johnson et al., 2008; Kortelainen et al., 2006).

Outcomes related to blanket peat and forest cover may be relevant in the wider European context considering the levels of peatland drainage (Paavilainen and Päivänen, 1995), and related P transfer risk (Drinan et al., 2013; Rodgers et al., 2010). However, the Irish Forest Service has adapted its policies in recent years to minimise the possible impact of afforestation in sensitive areas. This has included regulation in areas of acid sensitivity and less productive sites. In combination with other policies, this has resulted in a decrease in the planting of lands (particularly peats) that require inputs of N and P at afforestation and/or major changes to drainage (Upton et al., 2014).

The process of afforestation has potentially negative impacts on water quality resulting from site disturbance in the form of ground cultivation, drainage, fencing and planting. However, such intrusive events are relatively rare during the forest cycle compared to seasonal land use activities in traditional agriculture. Further, the application of nutrients to forest sites is typically less than agriculture (May et al., 2009). Neither model produces a significant relationship between afforestation and water quality.

In summary, results indicate a negative relationship between agricultural activities and water quality over the time periods examined. The longitudinal nature of this study allows us to observe the temporal impacts of forest cover on water quality outcomes while taking into consideration other land uses and catchment characteristics. The initial impact of afforestation is negative, but the effect is not significant due to infrequent and low levels of disturbance over time. Further, the benefits of standing forest cover eclipse negative impacts associated with afforestation over time. The environmental co-benefits associated with long-periods of minimal disturbance outweigh the potential negative impacts of initial afforestation. The comparative frequency of management interventions associated with agricultural land uses make forest cover a less intrusive land use from a water quality perspective. However, a lack of activity data on forest harvesting and the significant gaps between time points results in a less than complete picture. Increased frequency of collection and the inclusion of additional harvest data would offer additional insight into the factors driving water quality outcomes in rural catchments. Despite this shortfall, the results give a better understanding of the factors that influence rural water quality outcomes. These results are relevant in both the Irish and European contexts. To provide further insight into the relationship between water quality and land use change, this study also simulates the impact of an increase in forestry and a corresponding decrease in agricultural area.

5.6. Results III: land use change simulation

To estimate ex-post impacts of policy, the use of simulation models has become increasingly common (Morrissey et al., 2013). Table 4 reports the results of a simulation that increases afforestation and forest cover in each ED by 10% and reduces agriculture area by the equivalent area (typically much less than 10%). In addition, Fig. 2 presents the marginal difference in water quality outcomes per Q-value category post simulation. We assume the same livestock stocking rate per ha. As such, we estimate the impact of a smaller agricultural area and less

Table 3
Results of ordered probit models of the impacts of rural land uses on water quality (Q-value) outcomes.

	Pooled Parameter Estimates (Model 1)	Random Effects Parameter Estimates (Model2)
Commercial & Residential Share		
Septic Tank Density	−0.0004* (0.0002)	−0.0004 (0.0003)
Year Interaction (Base 1991)	–	–
Septic Tank Density x 2002	0.0003 (0.0003)	0.0004 (0.0003)
Septic Tank Density x 2011	0.0005* (0.0003)	0.0008*** (0.0003)
Commerce	−0.2884*** (0.1120)	−0.3780** (0.1626)
Environmental Characteristics		
Rainfall	5.30e-06 (0.0002)	0.0002 (0.0003)
Year Interaction (Base 1991)		
Rainfall x 2002	0.00001 (0.0002)	−0.00003 (0.0002)
Rainfall x 2011	0.0001 (0.0002)	−4.24e-06 (0.0002)
Temperature	−0.0838 (0.0559)	−0.1230 (0.0766)
Year Interaction (Base 1991)		
Temperature x2002	−0.0135 (0.0214)	−0.0144 (0.0243)
Temperature x 2011	−0.0487** (0.0206)	−0.0545** (0.0257)
Area (ha)	−0.0001*** (0.00002)	−0.0002*** (0.00004)
Median Elevation	0.0029*** (0.0005)	0.0044*** (0.0008)
Mean Slope	0.1283*** (0 0.0160)	0.1900 *** (0.0282)
Blanket Peat Share	0 0.9233*** (0.2131)	1.4346*** (0.3511)
X Coordinate	−2.84e-06*** (9.10e-07)	−4.47e-06*** (1.62e-06)
Y Coordinate	3.53e-06 *** (8.92e-07)	4.15e-06** (1.64e-06)
Agricultural Share		
Cereal Share of Land Use	−0.5334*** (0.1819)	−0.2750 (0.2641)
Organic N Density	−0.0027*** (0.0009)	−0.0016 (0.0012)
Year Interaction (Base 1991)		
Organic N Density x 2002	−0.0002 (0.0025)	−0.0006 (0.0013)
Organic N Density x2011	0.0025** (0.0011)	0.0038*** (0.0014)
Forestry Share		
Forest Cover Share of Land Use	4.2032*** (1.0327)	6.0528*** (1.5834)
Afforestation (sq) Share of Land Use	−0.0232 (0.0838)	0.0203 (0.1003)
Forest Cover *IQR 1	0.2623*** (0.0724)	0.2640 *** (0.0990)
Forest Cover *IQR 2	0.0882 (0.0834)	0.1212 (0.1158)
Forest Cover *IQR 3	0.1054 (0.0793)	0.1694 (0.1054)
Forest Cover *IQR 4	0.0075 (0.0975)	0.0997 (0.1391)
Forest Cover *IQR 1 x M. Slope	−0. 0261 (0.0187)	−0.0160 (0.0259)
Forest Cover *IQR 2 x M. Slope	−0. 0247 (0.0224)	−0.0298 (0.0319)
Forest Cover *IQR 3 x M. Slope	−0. 0379** (0.0181)	−0.0550** (0.0232)
Forest Cover *IQR 4 x M. Slope	−0.0198 (0. 0202)	−0.0633** (0.0296)
Forest Cover *IQR 1 x Blanket Peat	−0.6079*** (0.2226)	−0.95340*** (0.3047)
Forest Cover *IQR 2 x Blanket Peat	−0.0562 (0.2535)	0.0245 (0.3556)
Forest Cover *IQR 3 x Blanket Peat	−0.7329*** (0.2405)	−0.9505*** (0.3090)
Forest Cover *IQR 4 x Blanket Peat	−0.7331*** (0.2080)	−0.9712 *** (0.2100)
Pseudo R2	0.1023	–
/sigma2_u	–	0.9985 (0.0611)
Prob > chi2	–	0.0000
N	7497	7497

*** denotes statistically significant at 1%, ** denotes statistically significant at 5%, and * denotes statistically significant at 10%, standard errors in parenthesis. Forest Cover *IQR = Interquartile Range of Forest Cover.

animals than before.

The model predicts the simulated water quality values in 2177 ED's for the year 2011. Of the 2177 values simulated, 2.62% of points showed enough improvement to move into another water quality category. The remaining points were unchanged. The majority of the modest improvement was seen in the 'moderate' (Q3) to 'good' (Q4) categories. Twenty-two monitoring points showed enough improvement to move into 'high status' (Q4.5 or Q5) categories.

Fig. 3 presents the results of a sensitivity analysis conducted on the averaged values of the independent variables from EDs that improved in water quality status. The independent variables related to forest cover, afforestation and organic N density have been varied, one at a time, up to 30% to investigate the sensitivity of the simulation results to change in those specific input parameters. Afforestation has a very slight negative impact, while organic N density increases have a more pronounced negative impact. However, the most significant impact is the positive effect of forest cover at increasing density. As such, though we see a positive impact from the substitution of livestock, the greatest impact results from the water quality benefits conferred by increased forest cover.

The combined impact of forest cover and reductions in livestock numbers drive the water quality increases. Fig. 4 shows the simulated change in water quality mapped to the respective monitoring points for 2011 and the change in organic N density in kg per ha at the quartile level. The majority of water quality monitoring points experiencing a change are in proximity to those areas that have, in general, experienced the larger declines in organic N density.

5.7. Looking forward

As noted in the most recent EPA (2019), Ireland has seen a decline in the highest Q-value sites (Q4–5 and Q5) over recent decades. In 1987–1990, Q4–5 and Q5 sites made up 31.6% of the total, in the 2019 assessment, these sites made up 17.2% of the total. The EPA (2019) recognises the water quality challenges as a result of pressures from agricultural land uses and waste water discharge. However, though

Table 4
Results of a land use change simulation of forestry replacing livestock.

	Actual River Quality									
Sim River Quality	1	1.5	2	2.5	3	3.5	4	4.5	5	Total
1	1	0	0	0	0	0	0	0	0	1
1.5	0	1	0	0	0	0	0	0	0	1
2	0	1	4	0	0	0	0	0	0	5
2.5	0	0	1	35	0	0	0	0	0	36
3	0	0	0	2	236	0	0	0	0	238
3.5	0	0	0	0	11	407	0	0	0	418
4	0	0	0	0	0	20	1036	0	0	1026
4.5	0	0	0	0	0	0	18	373	0	391
5	0	0	0	0	0	0	0	4	27	31
Total	1	2	5	37	247	427	1054	377	27	2177

Note: The table represents actual water quality values compared with simulated water quality values. Grey cells indicate points with no change, black cells indicate improved points. Results are presented for 2011 activity data.

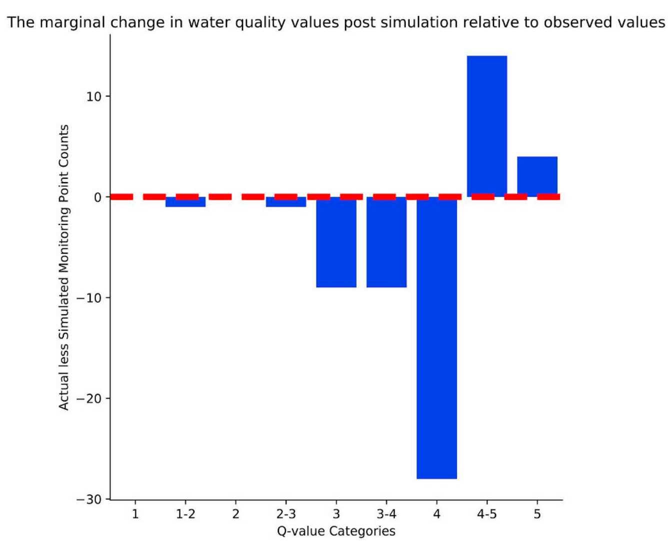

Fig. 2. Marginal difference in water quality outcomes post simulation relative to observed values
Note: The Y axis displays the observed Q-value numbers less the simulated Q-value numbers, while the X axis displays the water quality categories.

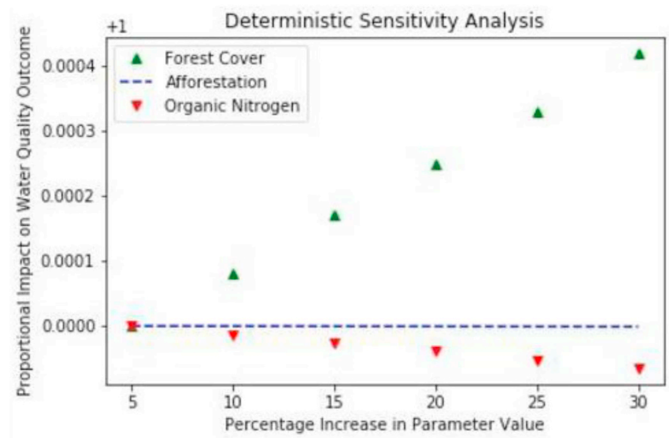

Fig. 3. Water quality model parameter sensitivity analysis.

recent reports recognise the potential negative impacts from poorly managed forested land, the potential for water quality benefits are less well recognised. The results of the land use change simulation have a small, but perhaps critical, impact in terms of water quality gains. The marginal increases in water quality resulting from well managed forest cover have the potential to contribute to plausible pathways for the restoration of high status Q-value areas. In addition, given the importance of forestry as a climate mitigation option for Ireland, afforestation has the potential to significantly enhance the achievement of multiple policy goals. However, the potential synergies must be overtly pursued in practice in order to be realised. As such, it is important to highlight the potential role of afforestation and forest cover to the development of sustainable pathways for multiple environmental policy goals.

6. Conclusions

The purpose of this paper was to examine the impact of private afforestation on water quality, using activity data on rural land use pressures. Previous research has highlighted the risk of sediment and nutrient disturbances associated with the process of afforestation and other specific forest management activities. Until now the impacts of forest cover and afforestation in an Irish context have not been analysed in relation to other land use pressures over a long time horizon.

Our study models the impact of afforestation and forest cover, while accounting for specific land use pressures within catchments between 1991 and 2011. In addition, we simulate 10% increase in afforestation and forest cover and a corresponding decrease in agricultural area. This

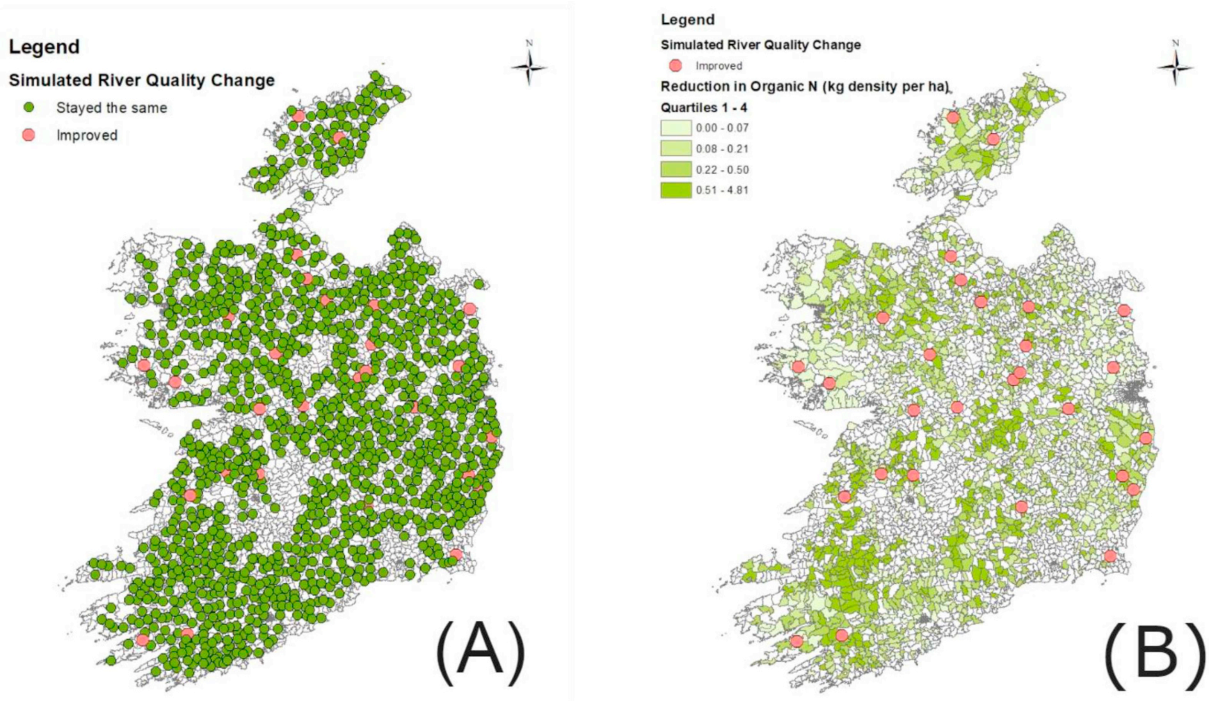

Fig. 4. Spatial mapping of water quality simulation results and resulting change in organic N density.

holistic analysis enables a better understanding of the impact of forestry on water quality outcomes within the wider land use context. In addition, greater insight informs the environmental sustainability of land use policy decisions targeted at climate mitigation.

The management of seasonal agricultural land uses, such as livestock, constitutes an important driver of water quality outcomes (Schulte et al., 2006). Long standing forest cover requires little if any nutrient inputs, resulting in a reduced risk of nutrient and sediment transfer to water courses (May et al., 2009; van Dijk and Keenan, 2007) and requires far fewer management interventions relative to seasonal agricultural land uses. This study offers additional insight by accounting for holistic catchment activity over an extended time period.

Analysis of activity data indicates a negative relationship with regards to water quality and agricultural activity and STS density. Though previous research has found a negative relationship between water quality and afforestation at a specific point in time, this study does not report a significant negative relationship over time, and finds forest cover is associated with significant positive water quality outcomes between 1991 and 2011. In addition, the results of the land use change simulation show that an increase in afforestation and forest cover by 10% and a corresponding decrease in agricultural activity could yield a positive overall impact on water quality, specifically with regards to the restoration of high status areas. As such, afforestation and well managed forest cover can contribute to plausible pathways to the restoration of high status Q-value sights, while also contributing to the achievement of climate mitigation targets.

Declaration

The authors confirm that this study has not been published previously, that it is not under consideration for publication elsewhere, that its publication is approved by all authors and tacitly or explicitly by the responsible authorities where the work was carried out, and that, if accepted, it will not be published elsewhere in the same form, in English or in any other language, including electronically without the written consent of the copyright-holder.

Declaration of Competing Interest

None.

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Appendix A. Additional Resources

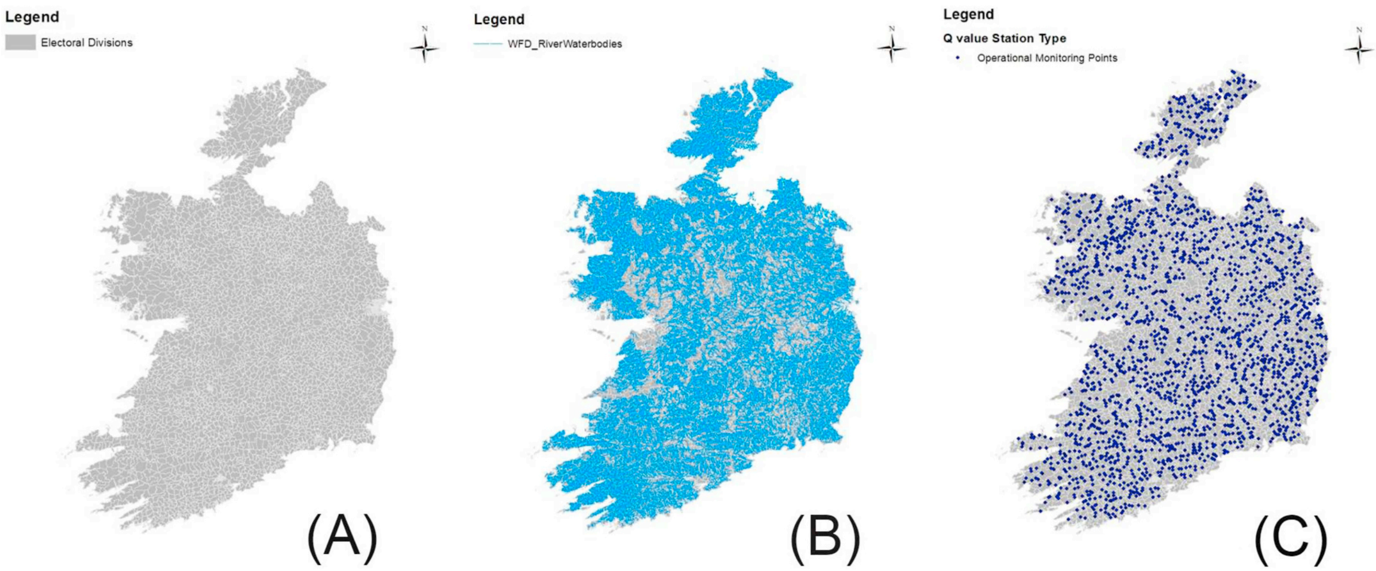

Fig. 5. Location of electoral divisions, rivers & EPA Q-values monitoring stations.

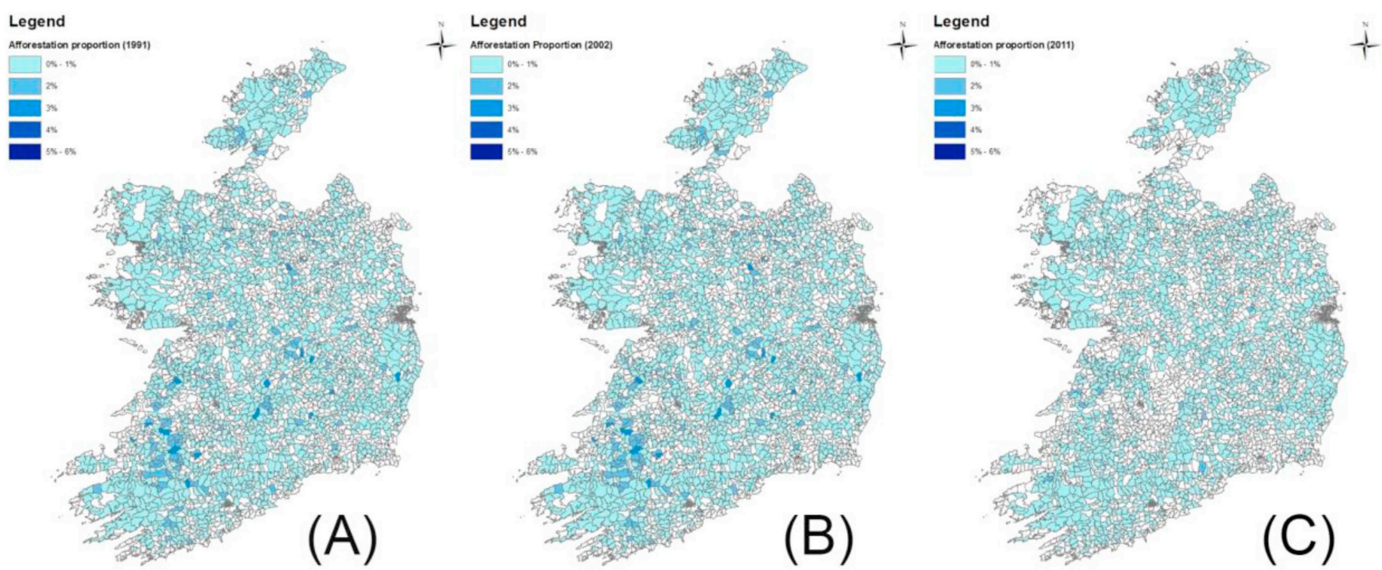

Fig. 6. Afforestation as a share of land use by electoral division for the years 1991, 2002, 2011.

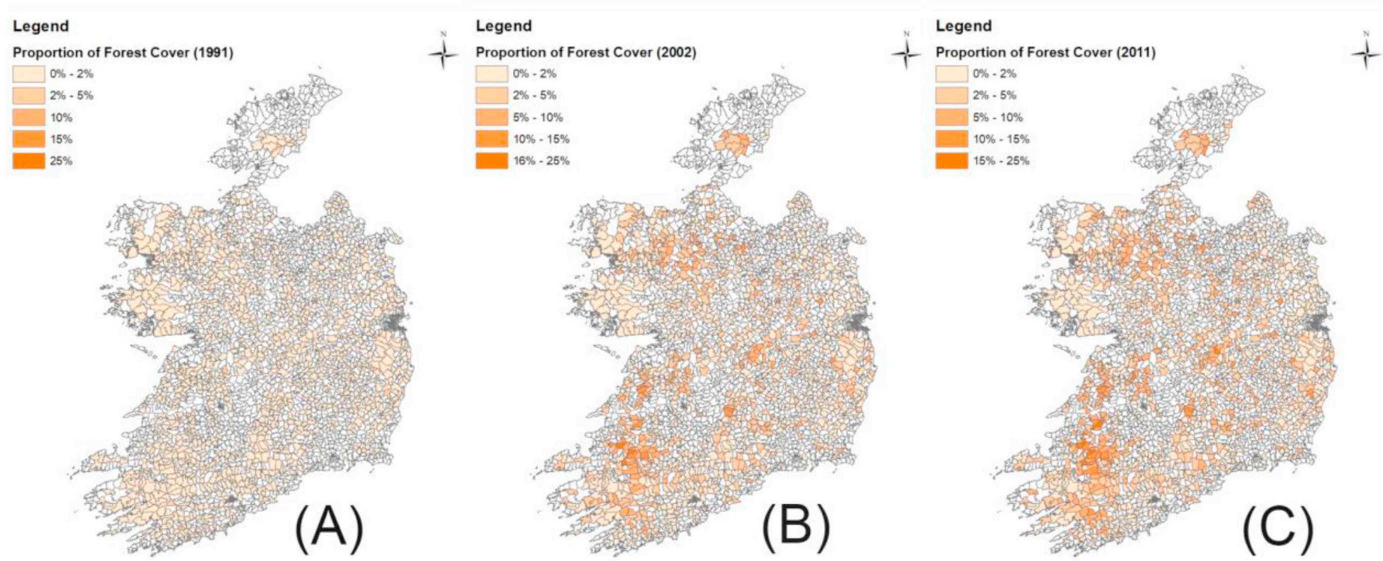

Fig. 7. Forest cover as a share of Land use by electoral division for the years 1991, 2002, 2011.

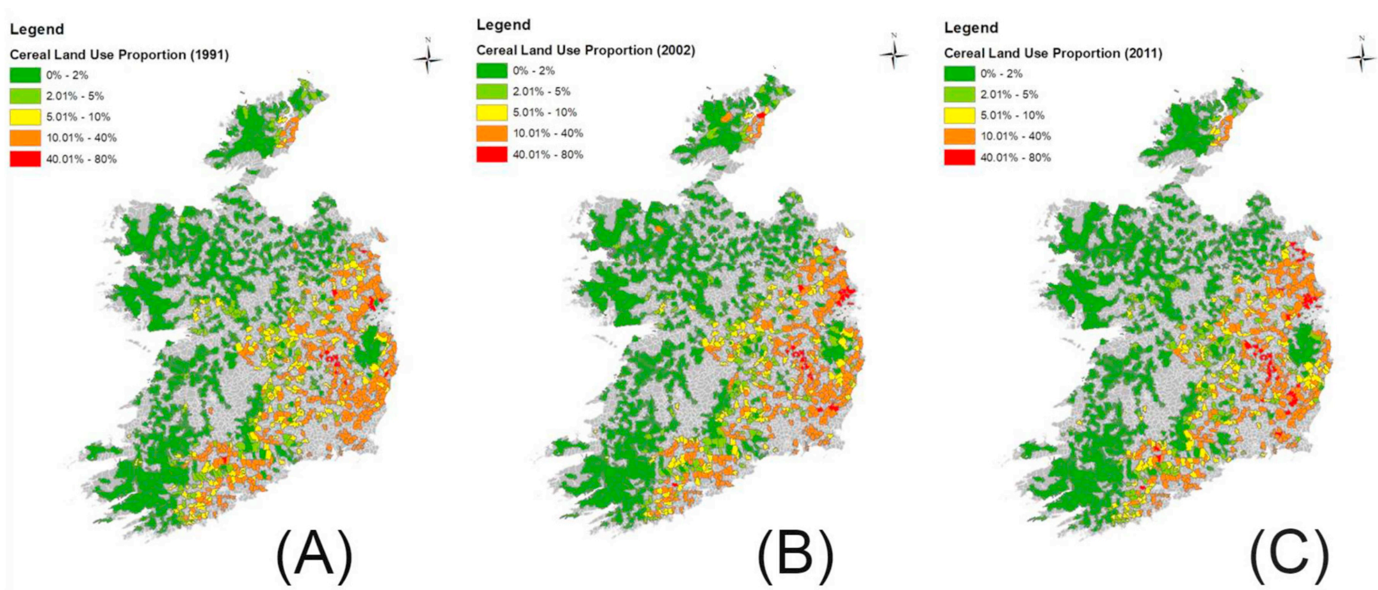

Fig. 8. Cereal share of Land use by electoral division for the years 1991, 2002, 2011.

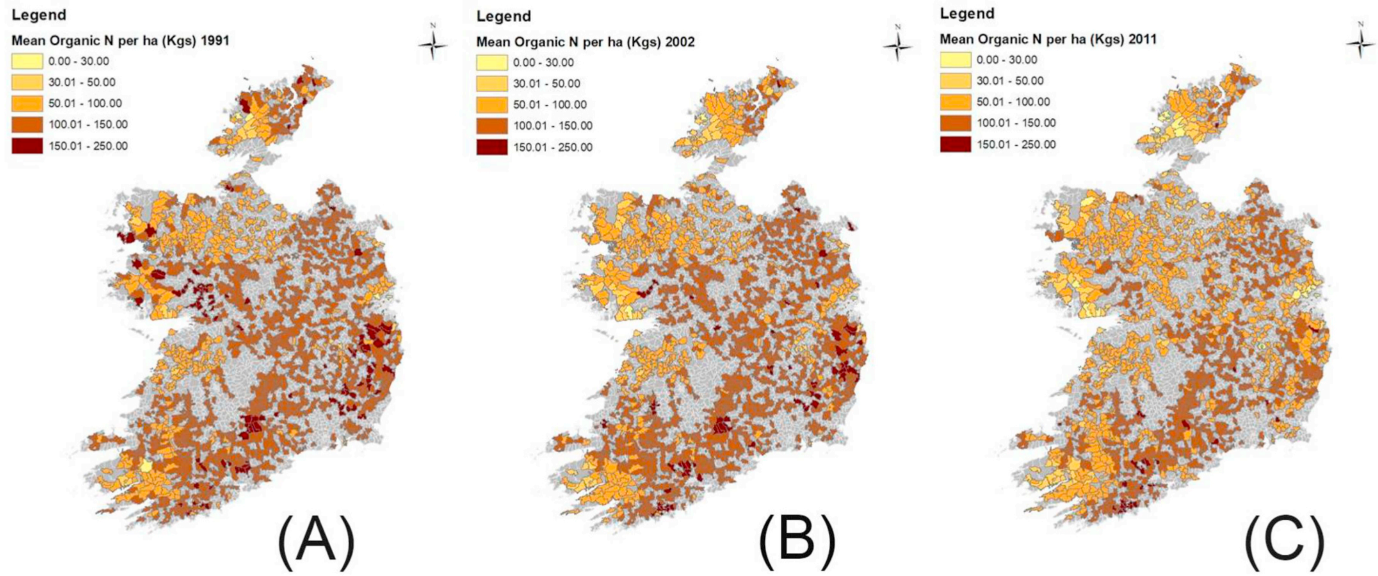

Fig. 9. Mean Organic N Density per ha (Kg) by electoral division for the years 1991, 2002, 2011.

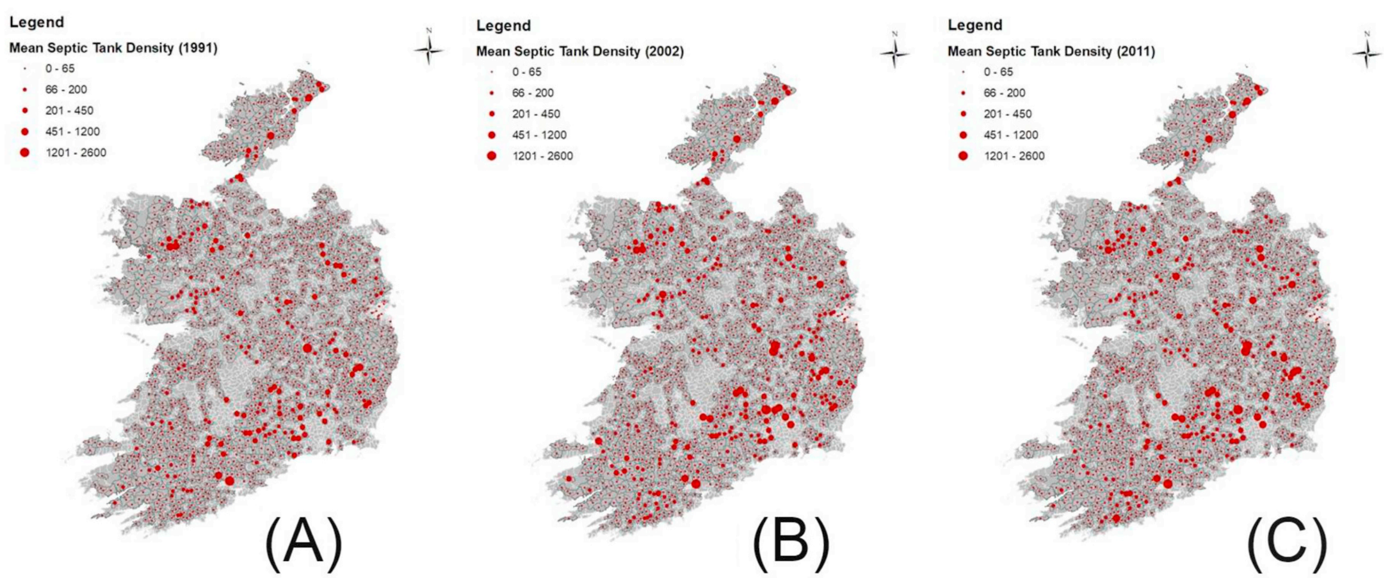

Fig. 10. Mean Septic Tank Density by electoral division for the years 1991, 2002, 2011

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