MODELLING THE ECONOMICS OF WATER POLLUTION IMPROVEMENT IN THE AGRICULTURAL CONTEXT

A dissertation submitted

by

Aksana Chyzheuskaya

B. Econ. Sc. (BSEU), M. Econ. Sc. (NUIG)

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DECLARATION

I declare that this thesis has not been submitted as an exercise for a degree at this or any other university and is entirely my own work. All research contained herein that is not entirely my own but is based on research that has been carried out jointly with others is duly acknowledged in the text wherever included.

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Aksana Chzheuskaya
THESIS SUMMARY

This thesis consists of six separate yet connected chapters dealing with the impact on farm income of implementing the Water Framework Directive (WFD (Directive 2000/60/EC). The first chapter is an introduction to the research conducted in this thesis. The second chapter is a review of the literature on water quality and water pollution in general and on the water quality as intended in the WFD. The review in chapter two is undertaken to provide a comprehensive understanding of water quality and water pollution, the current policy challenges and the gaps in the existing research that must be addressed.

The third chapter of the thesis is an econometric analysis of the key drivers of water quality in Irish river systems. The analysis is undertaken by combining data from the Environmental Protection Agency Ireland (EPAI) water quality monitoring stations with spatially referenced information on the river catchments, information from the Irish census of agriculture, septic tank density data, population density and environmental 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. The aim of this analysis is to establish the contributing sources that are associated with the lower water quality in Irish rivers.

The fourth chapter introduces a simulation model with some extensions and sensitivity analysis. This model is developed with a view to provide decision-makers with information about the impacts of environmental policies on farm’s operations and income. In particular, the policies that affect farms’ nitrogen (N) balance are considered. The model aids the economic assessment of the effect of N mitigation measures in Ireland. Within the microsimulation framework, econometric estimations are employed to simulate the changes in farm income and farm N balance that would arise from N pollution reduction measures on dairy farms in Ireland. As a case study, the two mitigation measures that have previously been assessed by Hennessy et al. (2005) and Fezzi et al. (2008) are considered, namely: 1) a stocking rate reduction to achieve a maximum organic nitrogen of 170 kg/ha; 2) a 20 percent stocking rate reduction. Then, as a model extension, the cost-effectiveness of seven farm level N mitigation measures is compared through calculating the Cost per Unit Abated
(CPUA) displaying the cost of each N unit reduction for each measure and ranking the abatement measure by CPUAs. This chapter also attempts to fill the gap in the existing research literature and provide stakeholders with economic analysis informing the decision-making process concerning policies for the agricultural sector. The country average CPUA for each measure as well as CPUA ranking for individual farms are reported and discussed. Furthermore, the sensitivity analysis of the CPUA results and their ranking to price changes is conducted. Usually the price volatility analysis is related to the potential negative effects on consumer welfare especially in poorer households from rising food prices. In chapter VI the effect of price volatility on CPUA is explored instead. In particular, this study highlights that the assessment and in turn the ranking of mitigation measures will depend on the commodity and input prices prevailing in the particular period when the analysis is conducted.

The fifth chapter is a case study that is conducted using a spatial microsimulation model. In this chapter, the costs associated with protecting an endangered species - namely the freshwater pearl mussels (FWPM) Margaritifera Margaritifera and Margaritifera Duravensis – are calculated in the relevant river catchments. The importance of this analysis is threefold. First, the use of the spatial microsimulation model in order to assess the policy impact at a relevant spatial level. Second, the requirement to protect the FWPM protected under the Habitats Directive (Council Directive 92/43/EEC), the Wildlife Acts ((Wildlife Act No 39 of 1976), amended 2000) and the Water Framework Directive (Directive 2000/60/EC). Finally, the falling numbers of these pearl mussels suggest a decline in water quality. The declining numbers of once abundant fresh water pearl mussels is one of the indicators of deteriorating water quality highlighting the need for a habitat protection policy in Ireland. This mollusc is not only a very sensitive organism that signals the water pollution problem but it is a unique and endangered species that has to be preserved for future generations. However, protecting the endanger species is a costly business, which is highlighted in this chapter.

The overall conclusions and discussion of the research undertaken in this thesis and the research findings are given in chapter six and this concludes the thesis.
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<td>COA</td>
<td>Census of Agriculture</td>
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<tr>
<td>CSO</td>
<td>Central Statistics Office</td>
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<tr>
<td>COD</td>
<td>Chemical Oxygen Demand</td>
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<td>CPUA</td>
<td>Cost Per Unit Abated</td>
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<tr>
<td>DAFM</td>
<td>Department of Agriculture, Food and Marine</td>
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<td>DC</td>
<td>Direct Costs</td>
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<td>DED</td>
<td>District Electoral Division</td>
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<td>DEM</td>
<td>Digital Elevation Model</td>
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<td>ED</td>
<td>Electoral Districts</td>
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<td>EEC</td>
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<td>EPAI</td>
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<td>EU</td>
<td>European Union</td>
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<td>FADN</td>
<td>Farm Accountancy Data Network</td>
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<td>FIPS–IFS</td>
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<td>FGM</td>
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<td>FWPM</td>
<td>Fresh Water Pearl Mussel</td>
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<td>GIS</td>
<td>Geographical Information System</td>
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<td>GHG</td>
<td>Green House Gas</td>
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<td>GM</td>
<td>Gross Margin</td>
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<td>GO</td>
<td>Gross Output</td>
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<td>IFA</td>
<td>Irish Farmers’ Association</td>
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<td>LU</td>
<td>Livestock Units</td>
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<td>ME</td>
<td>Marginal Effects</td>
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<td>NFS</td>
<td>National Farm Survey</td>
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<td>ND</td>
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<td>N</td>
<td>Nitrogen</td>
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<td>NUIG</td>
<td>National University of Ireland, Galway</td>
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<td>NUTS</td>
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<tr>
<td>OECD</td>
<td>Organisation for Economic Co-operation and Development</td>
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<td>OLS</td>
<td>Ordinary Least Squares</td>
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<tr>
<td>P</td>
<td>Phosphorus</td>
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<td>REPS</td>
<td>Rural Environmental Protection Scheme</td>
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<td>Simulation Model of the Irish Local Economy</td>
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<td>SBMP</td>
<td>Sub-Basin Management Plans</td>
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<td>TDS</td>
<td>Total Dissolved Solids</td>
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<td>TSS</td>
<td>Total Suspended Solids</td>
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<td>UK</td>
<td>United Kingdom</td>
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<td>WFD</td>
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Chapter I.

INTRODUCTION TO THE THESIS

1.1 Introduction

The WFD adopted by the European Commission in 2000 (Directive 2000/60/EC) requires the integrated management of water resources throughout the EU and has stringent deadlines for implementation. One of these deadlines is the requirement for all Member States to ensure that all water bodies are of ‘good ecological status’ by 2015 (Elnaboulsi, 2009, Olmstead, 2010). While previous directives such as the Nitrates Directive (Council Directive 91/676/EEC) focused on input based measures such as the threshold amount of fertiliser applied per hectare of agricultural land, the WFD focuses on outcomes such as improved water quality. The Environmental Protection Agency Ireland (EPAI) reported that in 2009 only 52 percent of rivers were of good ecological status, 15.3 percent of ground water sites were of poor status and 36 percent of estuaries and coastal waters in Ireland did not reach “good” water quality (EPAI, 2010). These findings indicate that, despite efforts to reduce pollution of water bodies in Ireland, more work needs to be done to alleviate further. As there are many potential causes of water quality deterioration, an integrated approach to water quality policy implementation by all sectors of society is required. As such, information on all the factors affecting the level of water quality in the river systems is a key requirement for the development of policy that will ensure the WFD targets are met.

Besides introducing stringent deadlines for water quality improvements, the WFD highlights the need to internalise costs as per the polluter-pays principle and advocates the use of cost-benefit analysis when determining which measures to implement in order to achieve the WFD goals (Görlach and Pielen, 2007). To fulfil this requires a range of information has to be known. In particular, the marginal abatement cost and the marginal damage cost functions need to be known. Equating these curves allows the efficient level of pollution to be determined. However, determining an efficient level of pollution requires a lot of data at a refined spatial resolution and such data is rarely available to policy-makers or researchers. Thus, it may not always be possible to determine the efficient level of pollution. In light of
the limited economic resources policy-makers can still choose cost-efficient instruments to achieve these standards, even if the standards are themselves not efficient (Olmstead, 2010).

A range of research on water quality has been conducted in Ireland and internationally with some studies focusing on the relationship between human activities and water quality (Curtis and Morgenroth, 2013, Donohue et al., 2005) and others on the cost of water protection (Görlach et al., 2004, Görlach and Pielen, 2007). Hanley and Black (2006) conducted a cost-benefit analysis of the WFD implementation in Scotland from both a micro and macro-level perspective. Their micro-level case-study analysis showed that the costs were disproportionately larger than the benefits. At the macro level they compared the cost of implementing the WFD on the impacted industries and the benefits to the whole country, which showed that the benefits outweighed the costs (Hanley and Black, 2006). Fezzi et al. (2010) undertook an integrated cost-effectiveness analysis of different policies needed to fulfil the WFD requirements in relation to the agricultural sector in Britain (Fezzi et al., 2010). They considered four different measures which would result in reduced nitrogen (N) emissions and compared the costs of each measure for a case-study catchment. Cuttle et al. (2007) quantified the cost of 44 mechanisms to control diffuse water pollution from agriculture in the UK (Cuttle et al., 2007). Lennox et al. (1998) compared the incidence of agricultural water pollution in Northern Ireland with those in England and Wales and found that neither penalties nor grant aids influenced the number of incidents of silage and non-silage related pollution from farms in the long run, calling into question the effectiveness of such policies in mitigating pollution (Lennox et al., 1998). There has also been a wide range of research in other EU member states concerned with examining the effects of agricultural activities on water quality and the cost of strategies aimed at abating the negative impacts of agriculture (Vatn et al., 2006, Vatn et al., 1999, Vatn et al., 1997, Brady, 2003, Brouwer et al., 2008, Pulido-Velazquez et al., 2008, Volk et al., 2008). Findings from these studies generally suggest that to meet WFD goals, agricultural practices will have to be changed drastically. Some proposed changes include a reduction of fertiliser applied to crops and grass and a reduction in cattle and sheep

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1 The efficient level of pollution is the level where marginal damage curve intersects the marginal abatement curve; while an efficient policy tool is the tool that achieves pollution reduction at a least cost.
stocking rates (Bateman et al., 2006, Haygarth et al., 2003). However, the linkages between agricultural activities and water pollution at present are far from clear. There is even less knowledge on the costs that farmers, and as a result consumers of agricultural products, will bear as a consequence of such implementation and whether these measures will be effective in pollution mitigation.

Agriculture has been the main focus of research on the WFD implementation (Heathwaite et al., 2005, Humphreys, 2008, Moss, 2008b) while other major diffuse sources of water pollution have been omitted from the majority of the analyses. There are a wide variety of economic activities that can have a significant negative impact on water quality including construction, septic tanks and street runoff (Novotny, 2003, Olmstead, 2010, Ritter and Shirmohammadi, 2000). One Irish study which attempted to examine the effect of a broader variety of factors on river water quality was conducted by Donohue et al. (2006). The effect of residential density as well as agricultural intensity on the ecological quality of water was examined. After linking catchment characteristics and water chemistry with the ecological status of 797 hydrologically independent rivers throughout Ireland, Donohue et al. (2006) 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) influenced 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 fertiliser use were all negatively associated with water quality. These studies indicate that in order to meet the WFD targets the Member States may have to shift their attention from agriculture to include a broader range of diffuse pollution sources.

In the meantime, agriculture is still at the centre of the debate and despite extensive research in the area of water pollution mitigation from agricultural activities, there is still a gap in the existing literature in general and in the Irish context in particular. Stemming mainly from data unavailability or the complexity of the human activity/environment/pollution interaction, existing research consists of mainly small area case-studies or studies based on a representative (average) farm data. Notwithstanding the fact that such research provides a useful insight into the subject of pollution from agricultural activities, it however does not provide enough
information to inform policy decisions. Thus, there is a need for a framework that would inform policy decisions in a comprehensive manner – providing information about both economic and environmental outcomes of considered environmental policies. The overall aim of this thesis is to contribute to the existing literature by aiming to close the existing gap and develop one of the “building blocks” in the framework. In order to achieve the overall aim a number of objectives have been outlined for this thesis. These are described in the next section.

I.2 Research Objectives

Despite the large volume of work that has been carried out internationally and nationally on water quality and the implementation of the WFD, gaps still remain in the literature that need to be filled. This thesis seeks to fill some of these gaps and as such to provide policy-makers and other stakeholders with more information on the economics of WDF implementation. In particular, the main objective is to establish a framework that will be useful in assessing the possible economic impact of water resources protection under the WFD on individual farms in Ireland. This, in turn, will help to determine whether the cost of particular measures is disproportionally high or if not, what measures offer the most cost-efficient way to mitigate pollution at the farm level. To achieve this the more specific objectives of the thesis are to:

- establish the statistical relationship between human activities and water quality downstream in order to get an insight into the role of agricultural activities in relation to water quality;
- develop a micro-simulation model to assess the impacts of N reduction mitigation measures on farm income and, thus, to expand knowledge regarding the cost of WFD implementation;
- develop a framework that allows stakeholders to assess the cost-efficiency of different mitigation measures by calculating the cost per unit abated (CPUA) for alternative measures;
- assess the sensitivity of the CPUAs ranking to changes in market conditions (i.e., prices) and to inform decision-makers as well as policy-makers about the limitations of cost-efficiency information;
• conduct a case-study of the protection of the fresh water pearl mussel as an example of the potential annual cost to farmers if the protection of a particular water species was enforced;
• develop a modelling framework that can be used by other researchers in related studies (e.g., in assessing GHG mitigation measures).

I.3 Structure of Thesis

The research in this thesis starts with a review of the literature regarding the determinants of water quality and water pollution, existing research in this area and the key legislation for water protection (chapter II). This literature review includes only the literature that is relevant to the topics in the following chapters and is in no way exhaustive of the multi-dimensional and multi-disciplinary topic of water pollution mitigation and the implementation of the WFD.

The literature review undertaken in chapter II indicates that there are multiple potential sources of poor water quality, thus, it is important to examine the effect of a diverse range of factors on the ecological water quality. The analysis of the literature also revealed that the focus of existing research has often been limited in scope, focusing on the impact of changes in one particular sector (generally agriculture) (e.g. Heathwaite et al. (2005)) or focusing on one specific river catchment or site when evaluating the determinants of water quality (e.g. Schulte et al. (2009)). This allowed the gaps in the existing research to be identified and the direction of the further research to be determined.

Chapter III of this thesis represents an exploratory model, which, through combining a number of spatial datasets relating to agricultural activities, septic tanks density and environmental characteristics, seeks to gain insights into the relationship between human activities and impaired water quality in Ireland and 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 2010 Irish census of agriculture, which provides spatial information relating to agricultural activity, the 2011 Small Area Population Statistics (SAPS) which also provides spatially referenced information on septic tank and population density data, which have been enhanced by forestry cover data from the Forest Service and
environmental data in a Geographical Information System (GIS) framework. This analysis helps to understand the possible sources of diffuse pollution in Irish rivers. There is a lack of robust research that links the water quality in specific water sampling points and human activities in an Irish context. The statistical analysis conducted in chapter III suggests that organic N per hectare may be influencing the water quality in Irish rivers. Hence reducing organic N per hectare may lead to improved water quality.

Chapter VI of this thesis discusses the possible N mitigation measures and a microsimulation model that is developed with a view to simulating the impacts, such as the change in N production and in farm income at the farm level, associated with policy responses to existing regulations. This model aids the assessment of the economic impact of N mitigation measures in Ireland. In order to validate the model, the two mitigation measures that have previously been assessed in a different methodological framework by Hennessy et al. (2005) and Fezzi et al. (2008) are considered, namely: 1) a stocking rate reduction to achieve a maximum organic nitrogen of 170 kg/ha; 2) a 20 percent stocking rate reduction. The simulated results are compared to those of Fezzi et al. (2008) and Hennessy et al. (2005). After the model is validated, it is further extended to include five more mitigation measures. The full seven measures assessed within this chapter’s modelling framework are: 1) reduction of fertiliser application by 20 percent; 2) LU reduction to achieve N 170 kg/ha; 3) reduction of stocking rates by 20 percent; 4) utilisation of new more expansive feeds to reduce N dietary intake; 5) fencing off streams; 6) higher performing dairy breeds; 7) efficient slurry application.

The purpose of this model is to provide policy-makers with an economic analysis to aid in the decision-making process concerning policies for the agricultural sector. The cost-effectiveness of different farm level N mitigation strategies is investigated by calculating national and farm-specific CPUAs for each measure displaying the cost of each N unit reduction for each measure. The measures are then ranked according to their cost effectiveness. A sensitivity analysis of the CPUA ranking is then conducted. The need for a sensitivity analysis is based on the hypothesis that the assessment and in turn the ranking of mitigation measures depends on the commodity and input prices prevailing in the period when the analysis is conducted. Two
sensitivity analyses of the results are performed. In order to illustrate the potential impact of price volatility on the cost effectiveness of different policy options, first the results are simulated for the years 1997-2009 holding quantities of inputs and outputs constant at their 2008 levels. Then price indices for different years are applied to determine whether the ranking of the CPUAs for the mitigation measures stays constant. In addition, the simulation exercise is repeated varying both prices and quantities over time and the ranking of CPUA is compared again. This is done to understand how informative the CPUA analysis is and what policy implications the results of such an analysis might have. In particular, the CPUA analysis is formally used to formulate policies based on efficient resource allocation – i.e. the least cost approach. However, as the analysis conducted in chapter VI illustrates, such approaches may not be reliable as their results may be dependent upon prevailing market conditions.

In chapter V the modelling framework developed in the previous chapter and the spatial microsimulation model described by O’Donoghue et al. (2012) are used to calculate the cost of protecting the populations of the fresh water pearl mussel (FWPM) in Ireland. These species are protected under a number of existing regulations (listed in chapter V) and have to be protected unless the cost of such protection is disproportionally high. As there is still no information on the cost of FWPM protection, this chapter provides stakeholders with the costs of some measures outlined in the Sub-Basin Management Plans (SBMP) where populations of FWPM are present. Moreover, the importance of assessing policy impacts at relevant spatial levels by utilising spatial microsimulation modelling techniques is highlighted. Following the recommendations put forward in the SBMPs, five measures that would reduce farming activities in the FWPM catchments are considered: 1) the reduction of fertiliser by 20 percent; 2) the reduction of livestock units (LU) to achieve 170kgN/ha; 3) a 20 percent LU reduction; 4) switching from tillage to beef production and 5) the fencing off of water body banks by 10 meters. These measures may lead to a reduction of pollution from agricultural activities in general (as has been outlined in the previous chapters) and a knock on effect is that they may aid in efforts to protect the FWPM.
The general conclusions to this thesis as well as limitations of the analysis conducted and the approaches used are provided in chapter VI. The potential future research output that may stem from the results of this thesis is also outlined in this chapter.

1.4 Overview of Methodology

The methodology used in a research study arguably determines the scope of the potential use of the model’s results. As no model is perfect and each model has limitations in terms of how the results obtained through it can be transferred to real life situations, the methodologies used in this thesis were carefully considered and the limitations arising from their usage are outlined in each chapter.

In chapter III an ordered probit regression analysis is used to examine the statistical relationship between water quality and human activities. This approach was chosen because the EPAI Q-value system 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 and hence the relationship between the dependent and independent variables is estimated with an ordinal response model. The EPAI water quality data only records the categorical level to which the monitoring point belongs. The ordered probit model is estimated using maximum likelihood via the Newton-Raphson algorithm to account for the probability of falling into ordered Q-value categories, 1 to 5 (Long, 1997). The results obtained through this model are used to get an insight into the relationships between the variables of interest.

In chapter VI a microsimulation methodology and statistical regression analysis are utilised to develop a modelling framework allowing one to simulate the effect of N mitigation measures on farm income in Ireland. This methodology was chosen as microsimulation techniques allow the analyses to be conducted at a farm level. Thus, the effect of the potential policy measures can be assessed at a micro level. These techniques have been widely used for many years and are an effective tool for evaluating the socio-economic impacts of different mitigation options where it is difficult or impossible to conduct a real life experiment (Merz, 1993, O'Donoghue et al., 2012). Microsimulation is also useful as it can be carried out using various techniques, for example, linear programming (Hennessy et al., 2005), partial
budgeting (Fezzi et al., 2008), and econometric regression analysis (Fezzi et al., 2010) amongst others. In this thesis ordinary least squares (OLS) regression is chosen for building the simulation model. The approach used in this thesis is similar to Fezzi et al.’s (2010) study who used a linear regression model to estimate the change in farm gross margin that arises from different policy measures and O'Donoghue and Lennon (2014) who used regression analysis to build a farm income generation model. This model is then extended to allow the calculation and subsequent ranking of the CPUAs of the N mitigation strategies at the farm level for cost-efficiency analysis. CPUA analysis has gained popularity in recent years. It has been used by many researchers and policymakers in the environmental domain as an effective tool for policy assessment and to evaluate developments in agriculture, especially for greenhouse gas (GHG) emissions and emissions trading (McKinsey & Co., 2007, UNFCCC, 2007, Moran et al., 2008). It has also been used by environmental economists to aid in prioritising different investment decisions. In the context of water quality, CPUA is used to calculate the cost of different policy measures to reduce pollution. The ranking of the CPUAs’ for different measures allows one to evaluate and rank the mitigation costs from cheapest to the most expensive, thus, making it an essential tool for informing policy decisions. In order to outline the limitation of CPUA analysis the sensitivity of the results arising from such analysis is conducted.

In order to show how the model developed in chapter VI of this thesis can be applied in an existing policy framework, it is used in chapter V to calculate the cost of protecting the fresh water pearl mussel (FWPM). This study has a spatial dimension – the protection of the FWPM affects only catchments where these species are present -, thus, the impact has to be assessed in the relevant areas. In order to limit the effects of each measure to the farms located in the FWPM catchment the results generated using the model developed in chapter VI are transferred to a simulated model - SMILE (Simulation Model of the Irish Local Economy). SMILE simulates spatially representative households (including farms) at the electoral district (ED) level using a number of data sets (O'Donoghue et al., 2012).

This analysis also highlights the limitation of existing models in informing policy decisions and the need for further analysis. It also shows how developing compatible
models helps to extend the existing research and provide the building blocks for developing a refined analytical framework for decision-making.

I.5 Outputs Resulting from the Research conducted in this Thesis

The research conducted in this thesis has been presented for peer review on a regular basis in the form of conference presentations, seminar presentations, working papers and journal submissions. A list of written documents (including papers submitted for publication and working papers) is presented in Table I.1.

Table I.1. List of Written Work Resulting from this Doctoral Research

<table>
<thead>
<tr>
<th>Title</th>
<th>Authors</th>
<th>Publication Details</th>
</tr>
</thead>
</table>

The presentations of the research outcomes are listed in Table I.2.
Table I.2. List of Presentations

<table>
<thead>
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<th>Year</th>
<th>Presentation</th>
</tr>
</thead>
<tbody>
<tr>
<td>2013</td>
<td>Modelling the Marginal Abatement Cost of Mitigating Nitrogen Loss from Agricultural Land, AES Conference, University of Warwick, UK, April 8th, 2013</td>
</tr>
<tr>
<td>2011</td>
<td>Measuring the Marginal Abatement Cost of Reducing Nitrogen pollution in Agriculture. Department of Economics, NUI, Galway</td>
</tr>
<tr>
<td>2010</td>
<td>Effect of the different policy options to reduce nitrate leaching on dairy farm income in Ireland. IEEN conference, Teagasc, Athenry</td>
</tr>
<tr>
<td>2009</td>
<td>Effect of Regulations on Performance of Service Sector. Department of Economics, NUI, Galway</td>
</tr>
<tr>
<td>2008</td>
<td>Economics of Water Pollution. Department of Economics, NUI, Galway</td>
</tr>
</tbody>
</table>

I.6 Collaborations and Contributions to this Thesis

The research in this thesis spans a range of disciplines (e.g. economics, agricultural economics, biology, animal science, geography, hydrology, etc.) and hence required input from a number of collaborators with expertise in many fields who provided specific information and data. The collaborators and contributors are acknowledged in the form of co-authorship to the papers and are listed in Table I.1. However the substantive work in this thesis was conducted by the author.
Chapter II

WATER QUALITY AND WATER POLLUTION: SOURCES AND IMPLICATIONS

This chapter is an introductory literature review on the concepts of water quality and water pollution, evolution of the legislation aimed at protecting water resources and the place that the agricultural sector has in the debate on water pollution. Prior to any economic assessment of water quality and water pollution, it is critical that these two concepts are properly understood in the modern context. It is then possible to address the gaps in the existing research related to these two concepts.

II.1 Introduction

Water can arguably be considered the most important natural and economic resource that exists on planet Earth. It is a very common substance and it is essential for the survival of all known forms of life as human bodies consist of about two-thirds water, woody plants 80 to 90 percent, and bacteria and other microorganisms 90 to 95 percent (Boyd, 2000). Water plays an important physiological, ecological and socio-economic role. Today, industrial countries use approximately 500 tons of water per person per year (Mäler and Vincent, 2005). Until recently water was considered an abundant resource. But population growth, increasing demand, pollution and climate change have resulted in a severe scarcity of water in many parts of the world (Tietenberg, 1988).

It is not only the quantity but also the quality of the water that has diminished as a result of pollution during the 20th Century. There are many implication of water pollution. The economic implications of water pollution concern extra costs imposed on those who are using it for production (the cost of cleaning) and health costs imposed on those who are using it for household use or recreation. If polluted water bodies continue to deteriorate, they will become unavailable for future generations and will be lost for any beneficial uses. Water quality and water pollution are two sides of the same coin; however, the regulatory and economic implications are different. Water quality is often mentioned in relation to the potential usage of water while water pollution is mentioned when the water quality has been negatively altered by human activities.
Pollution is an economic externality and, thus, there is strong rationale for government intervention as market mechanisms fail to include the externalities that occur during economic activities or many uses of water resources into the price of goods and services and to reach an equilibrium level of pollution leading to severe deterioration of water resources in many parts of the world. There is a call for internalising the cost of damage caused by pollution based on the polluter pays principle (Directive 2000/60/EC). However, in order to implement this principle the sources of pollution need to be known and the costs of damages/remediation need to be calculated. This is where policy-makers face a challenge. Most of the pollutants in the water bodies at present come from the diffuse sources which are very difficult to establish. There is also spatial and seasonal variability associated with diffuse pollution which makes it even more difficult to develop robust decision rules and put forward regulation. Diffuse pollution originates from different human activities and the sources are well known at this stage – runoff from impervious urban surfaces, sedimentation from construction works, improperly functioning septic tanks, and agricultural activities (Ritter and Shirmohammadi, 2000, Novotny, 2003). They are discussed in more detail later in this chapter.

Protecting water bodies comes at a great cost and in times of economic downturn and limited public economic resources there is a constant need for efficiency in public funds allocation. There is still a lack of information on economic costs and benefits of existing mitigation options to inform policy-makers. Thus, this thesis adds to the existing literature in providing more information for decision-makers in the Irish context. However, before proceeding to empirical data analysis it is important to understand the background issues associated with water quality and water pollution and the regulatory demands that are currently in place.

The remainder of this chapter is structured as follows. The definition and discussion of water quality and water pollution is given in section II.2. Section II.3 examines agricultural activities as a source of pollution. In section II.4 an overview of the existing regulations is given. The conclusion is presented in Section II.5.
II.2 Water Quality and Water Pollution

Boyd (2000) states that "the concepts of water quality and water quantity were probably developed simultaneously, but throughout most of human history, there were few ways for evaluating water quality beyond sensory perception and observations of the effects that certain waters had on living things" (Boyd, 2000). It was not until the middle of the 20th Century that these concepts became a topic of the environmental debate and by the end of the 1970’s - a topic for legislators.

Until the 1950s, many European water standards were based on dilution (for example, in Britain no treatment was required if one part of untreated sewage was diluted by 500 parts of receiving water flow). Today, the understanding of the quality status of water bodies and their pollution evolved into an expression of the water quality as water body’s integrity - either physical or chemical. Physical integrity implies habitat conditions of the water body that would support a balanced biological community. Chemical integrity refers to the chemical composition of water and sediments that would not be injurious to the aquatic biota (Barbour et al., 2000, Novotny, 2003, Novotny et al., 2005, Allan et al., 2006).

Water quality is defined as the physical, chemical and biological characteristics of water in relationship to a set of standards (Diersing, 2009). Water quality reflects the composition of water as affected by nature and human cultural activities and can be expressed in terms of both measurable quantities and narrative statements (Novotny, 2003). The water quality parameters were created in relation to human health and safety (water quality for human use) or environmental water quality, also known as ambient water quality. The standards for human use depend on the intended use of water (e.g., potable water quality standards) and are very strict. Ambient water quality standards vary significantly due to different environmental conditions, ecosystems, and intended human uses. Even if the water is used for non-drinking purposes like irrigation, swimming, fishing, rafting, boating, and industrial uses, the presence of toxic substances and certain microorganisms can still present health hazards. Even the ground water quality can be impaired as the input to aquifers can include water from artificial sources such as septic tanks, irrigation, artificial recharge, and sewer leakage (Novotny, 2003). The focus of this chapter and the thesis
is on ambient water quality and the potential impact that human activities may have on it.

Water quality depends on the local geology and ecosystem and human activities can negatively affect water quality (Donohue et al., 2006, Curtis and Morgenroth, 2013, Huang et al., 2014). Water quality is a complex issue, thus, it is assessed based on a number of indicators ranging from water temperature and pH balance and a set of chemical components like heavy metals to biological and microbiological parameters (Novotny, 2003).

**PARAMETERS OF WATER QUALITY**

There is a number of ways to classify water quality standards which could be primary or secondary (for potable water); ambient or emissions; numerical, chemical-based or narrative. The primary standards are standards that regulate substances that potentially affect human health, and secondary standards concern aesthetic qualities such as taste, odour and appearance (Wheeling Jesuit University 2004). Novotny (2003) states that the environmental water quality criteria and standards currently used throughout the world are either stream (ambient) or effluent (emission).

For each intended use and water quality benefit 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). EPAI lists 101 principal parameters of water quality (EPAI, 2001). These parameters fall into three groups of environmental indicators: 1) physical indicators; 2) chemical indicators; 3) biological indicators.

**Physical 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). Particulate matter, turbidity, and colour are important water quality parameters. Turbidity is caused by the particles that restrict light penetration and affect the growth of plants in aquatic ecosystems. Suspended particles also settle creating
bottom habitat degradation, one of the examples of which is the degradation of the freshwater pearl mussel habitat (discussed later in this thesis). Water for drinking purposes and for some industrial uses is degraded by significant amounts of turbidity or colour, and treatment is often necessary to remove these substances before it can be used. The primary source of suspended soil particles in water is soil erosion. Furthermore, organic particles originate from leaves falling into bodies of water, from overland flow, from microscopic organisms (plankton and bacteria) that inhabit natural waters, and from the remains of dead aquatic organisms (Boyd, 2000).

Chemical Assessment

The chemical assessment of water quality constitutes a number of indicators that include pH, biochemical oxygen demand (BOD), chemical oxygen demand (COD), heavy metals, nitrates, orthophosphates and pesticides among others (Boyd, 2000, EPAI, 2001, Wheeling Jesuit University 2004). The biochemical oxygen demand (BOD) and the chemical oxygen demand (COD) are important in water quality evaluation because oxygen is used by aquatic organisms in respiration. If the quantity of organic matter in streams is high, more dissolved oxygen is used for decomposition by the decomposing organisms. High BOD in streams can create dissolved oxygen reduction downstream and if extremely high, parts of the stream can develop anaerobic conditions (Boyd, 2000). Dissolved oxygen plays a vital role in aquatic ecosystems. In the absence of dissolved oxygen anaerobic microorganisms prevail and release many reduced substances (e.g. ammonia, nitrite, ferrous iron, hydrogen sulphide, and dissolved organic compounds) into the water as a result of the decomposition of organic matter. In the absence of adequate dissolved oxygen, aerobic microorganisms and “clean water" species disappear and only those organisms that can tolerate highly polluted conditions thrive. Such ecosystems have low biodiversity and impaired for most beneficial uses (Boyd, 2000, Kannel et al., 2007).

The pH variable in the chemical assessment of water quality is not given much attention in the policy debate. However, many chemical reactions that determine water quality are pH dependent; the changes in pH levels may have detrimental effects on aquatic flora and fauna (Boyd, 2000).
Nitrogen (N) and phosphorus (P) are other indicators of the chemical water quality assessment. They have been a focus of a numerous research including this thesis (Zhou et al., 2000, Hennessy et al., 2005, Camargo and Alonso, 2006, Bunting et al., 2007, Fezzi et al., 2008, Lally et al., 2009). Excessive ammonia nitrogen, nitrate and phosphorus in water contribute to eutrophication (nutrient enrichment and excessive plant growth) (Humphreys, 2008). In turn, excessive plant growth causes wide daily fluctuations in dissolved oxygen, and low dissolved oxygen concentrations at night, which may be harmful to aquatic animals. Blue-green algae are often abundant in eutrophic waters, and these algae are subject to sudden die-offs that can lead to dissolved oxygen depletion, they can also be toxic to other organisms. Some species may also impart an “off-flavour” to fish or cause taste and odour problems in drinking water. Dense algal blooms in waters used for recreational purposes limit visibility in water and can cause bad odours reducing water bodies’ aesthetic value (Boyd, 2000, Thiel et al., 1995).

Testing for pesticides is also important as it is reported that 9 of the 12 most dangerous and persistent organic chemicals are pesticides (Gilden et al., 2010).

**Biological Assessment**

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 related to the presence and abundance of members of insects. These indicator species are usually site-specific and vary from region to region (Boyd, 2000, Metcalfe-Smith, 2009, Wheeling Jesuit University 2004).

The word pollution is often used to describe unwanted changes to water bodies and refers to poor water quality. At its most inclusive, the term pollution can be used to describe all unwanted environmental effects of human activities (Merrington et al., 2003). The terms pollution, contamination, nuisance, and water degradation have often been used synonymously to describe faulty conditions of surface and ground waters (Novotny, 2003). In relation to water quality standards, pollution occurs at a point where the defined standards for water quality are exceeded.

Concerns over water pollution did not come to the spotlight until the middle of the 20th century when, in one example, the stench of the Thames River in London
became so unbearable that the British Parliament recessed during the affected periods. In another example, flammable wastes discharges from the greater Cleveland-Akron industrial area caused Cuyahoga River in Ohio to catch fire. It was not until 1972 that it was recognized that a significant portion of pollution reaching water may be originating from diffuse sources (Novotny, 2003).

It is common to classify pollution and/or pollutants by the types of pollutants (fund pollutants and stock pollutants); and the source of pollution (point pollution and non-point (diffuse) pollution). Fund pollutants are those for which the environment has some assimilative capacity. If the absorptive capacity is high enough relative to the rate of injection, they may not accumulate at all. Fund pollutants include degradable pollutants, thermal pollution, persistent pollutants and infectious organisms. Degradable pollutants can cause problems even if assimilative capacity of a water body is low because the process of organic waste decomposition consumes oxygen, which, in turn, causes the stream’s oxygen level to fall increasing fish mortality. Thermal pollution is caused by raised water temperature as a result of human activities (Bendini, 2011). Thermal pollution lowers the dissolved oxygen content and can result in serious ecological changes. Persistent pollutants are synthetic chemicals that are not sufficiently broken down. They accumulate in water bodies causing environmental problems. Infectious organisms (bacteria and viruses) in surface water and groundwater originate from domestic and animal wastes and wastes from industries such as tanning and meat packaging (Nair, 2010, Tietenberg, 2006).

Stock Pollutants are pollutants that accumulate in the environment. These are considered to be the most troublesome pollutants. These pollutants include inorganic chemicals and minerals, heavy metals (lead, cadmium, and mercury) which accumulate in both watercourses and in the aquatic food chain. Recently, medical waste and residuals from soaps and perfumes have also been found in water bodies (Tietenberg, 2006).

On the basis of the sources of pollution a distinction is made between diffuse pollution and point source pollution. This classification is most commonly used in policy debates regarding water pollution and ways to mitigate it. 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. Diffuse pollution is caused by the interaction between weather events and the landscape and is difficult to monitor and control (Ritter and Shirmohammadi, 2000). Novotny (2003) states that diffuse pollution is often a result of use and misuse of land, and its causes are mostly socioeconomic, encouraged by tradition, government subsidies, foreign demands for cheap products, and lack of information about pollution and polluting behaviour.

**Urban Pollution and Street Runoff**

Urban pollution is one of the leading sources of pollution in water bodies and the wider environment. The increasing population of the Earth is putting more and more pressure on ecosystems each year. Urbanisation causes dramatic changes in the environment in general and the local hydrology in particular. These changes include increased runoff and decreased evapo-transpiration, problems with deep and shallow infiltration due to increased imperviousness of the surface and the channelling of runoff into gutters increasing the risk of flooding. Once the impervious area of the watershed exceeds 10 percent, there is likely to be substantial deterioration (NEMO, 2005). 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 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).

The US Geological survey (Barnes et al., 2002) found that urban streams carry high traces of hormonal medication, anti-depressants, birth-control medication as well as high concentrations of pesticides and elevated concentrations of phosphorus. Other dangerous pollutants that are carried in street runoff include faecal matter of urban animals, hydrocarbons, metal, asbestos, car oils, tire wear, break-down parts, lubrication fluids, cigarette butts, chewing gum and other rubbish (Novotny, 2003, Yannopoulos et al., 2013).
SEPTIC TANKS

Septic tanks are more and more often mentioned in relation to water pollution. In Ireland, a large number of dispersed rural dwellings that often are not connected to sewage treatment systems and have individual septic tanks installed (Macintosh et al., 2011). When septic tanks are installed and/or managed incorrectly, they can become a major source of the ground water pollution as a result of a large number of untreated chemicals and bacteria entering the environment (Swarup et al., 1992). The recognition of septic tanks as a significant source of pollution has led to the introduction of the EU Waste Framework Directive (Directive 2006/12/EC). Ireland has over 400,000 septic tanks throughout the territory, which contribute to the problem of the water pollution (EC Ireland, 2011). Failure to regulate septic tanks in Ireland has led to a fine of 3 million Euros for Ireland (TheJournal.ie, 2013) and subsequent introduction of the Water Services (Amendment) Act (Water Service Act No. 6 of 2013).

II.3 Agriculture as a Source of Pollutants.

Agricultural activities and their contribution to water pollution have been a centre of the debates on all levels. Despite the number of sources of diffuse pollution, agriculture is one source of diffuse pollution that has been under the most scrutiny from researchers and policy-makers. It is a sector that, on one hand, is responsible for feeding an exponentially growing Earth’s population and, on the other hand, is under the constant pressure to reduce emissions from its activities that may cause pollution (Brown et al., 2000). It has seen major advances in the last century which allowed the sector to intensify and increase its output to meet the growing demand (Donal et al., 2001). It is due to the advances in the chemical industry in the post WWII period that allowed agriculture to solve the hunger problems (Merrington et al., 2003). However, a few decades later it became apparent that the path of the intense chemicals use is unsustainable and the damaged caused to the environment may be counteracting the growth achieved.

There are a number of pollutants that, if improperly used, may originate from agricultural land: nitrogen (N), phosphorus (P), pesticides, pathogens and sediment (Baldock, 1996, Conway and Pretty, 2009, Isherwood, 2000, Merrington et al., 2003,
Novotny, 2003). However, a number of factors determine the probability of pollution from agricultural land: management factors (the types of land use, crop, animals, land-use management, etc.); environmental factors (soil type, extreme weather events, etc.) (Donohue et al., 2005, Merrington et al., 2003, Novotny, 2003, Schulte et al., 2006, Ritter and Shirmohammadi, 2000). The land-use activities that are more likely to negatively affect water quality are dry-land cropland, irrigated cropland, pastureland, commercial forestland logging, confined animal feeding operations, aquaculture and nurseries (Merrington et al., 2003, Novotny, 2003).

**NITROGEN**

N has been a focus of political debate since 70’s, it has resulted in introduction of the EC Nitrates Directive (Council Directive 91/676/EEC) in 1991. It is one of the most abundant elements in the environment and, in itself, is not a pollutant. It only becomes a pollutant when it is inappropriately managed or when due to interaction between land-use and weather events it is lost to the wider environment from agricultural land. The role of N in agricultural systems cannot be overstated. Using N fertilisers increases yields and in certain crops mineral N may also help to improve the quality of the harvested produce. It is proved to be no coincidence that with increased agricultural production in the second half of the 20th Century, there has been an increase in N concentration in rivers, lakes and underground aquifers (Merrington et al., 2003, Ritter and Shirmohammadi, 2000).

There are a number of pathways by which N is lost from an agricultural soil (Lord and Anthony, 2000). The most desirable route is via crop uptake and/or removal by grazing or harvest since this produces both economic return for farmers, and is not a cause of pollution. But this is not always the case because N (available in the soil for crop uptake) is also vulnerable to loss from the soil (without economic return and a potential pollutant) through NO\(_3\) (nitrate) leaching, gaseous emissions during the denitrification and nitrification, and NH\(_4\) (ammonia) volatilisation (Merrington et al., 2003). It has been estimated that about 90 percent of all NH\(_4\) volatilization in Western Europe originates from agriculture and less than 10 percent from other sources (Ritter and Shirmohammadi, 2000). The potential for N runoff/leaching varies spatially and temporarily depending on soil characteristics and seasonal weather variations (Schulte et al., 2006). The factors that encourage nitrate loss from

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agricultural land are over-fertilization of crops and grassland; excessive livestock numbers; inappropriate use of manure and exposure of bare soil during the winter drainage period (Bateman et al., 2006).

It is not the loss of N from agricultural land per se that is the primary concern of policy-makers, but the effect that the excess N in surface and ground water has on human health and on the wider environment. N is known as a relatively non-toxic chemical that is rapidly excreted from the body, but it can be reduced to potentially toxic nitrite in the mouth and gut that can cause methaemoglobinaemia in young babies and gastric cancer (Boyd, 2000, Merrington et al., 2003). Excess of N in water bodies contributes to the problem of eutrophication of waters, causing changes in species; fish mortality; taints and problems with filtration of water for public water supplies increasing the cost of purification and impaired amenity value of the water bodies (Conley et al., 2009, Merrington et al., 2003). Nitrite (NO₂) can be absorbed from water by fish and other organisms causing functional anaemia known as brown methemoglobinemia and lower ability of blood to transport oxygen (Boyd, 2000).

Significant control of N leakage from agriculture can be achieved through the encouragement of management practices that minimise the opportunities for N to accumulate in the amount that is susceptible to loss (Cuttle et al., 2004, Ribaudo et al., 2001). There have been a number of studies (including this thesis) that examined the economic costs of reducing N loss from agricultural activities, just to mention a few – Cuttle et al. (2007), Fezzi et al. (2008), Hennessy et al. (2005), Lally et al. (2009). The main findings of these studies is that mitigating N pollution from agricultural land will come at a cost to farmers with the benefits of these measures are still unquantified.

**Phosphorus**

Another nutrient in water bodies that originates from agricultural activities (as well as other human activities) and can become a pollutant is P. Being a macronutrient that is essential for all forms of life, phosphorus has the function of energy transfer and storage in organisms and is a component of DNA, plays a role in plants photosynthesis, respiration, cell enlargement and division and growth, promotes early root formation and growth, and as is responsible for the quality of grain, fruit and
vegetable crops. P is important not only for plant and crop well-being, but it is also of major importance for other living organisms including animals (Merrington et al., 2003).

As agricultural production has intensified over the years, high P amounts have been applied regularly on most types of soil. A portion of this fertiliser P often becomes unavailable to the crop because of reaction with soil minerals, so more P than will be used by the crop is usually applied (Ritter and Shirmohammadi, 2000). P may also be introduced into the environment as a by-product of animal production (in manure and/or slurry), which is applied to land (Ritter and Shirmohammadi, 2000). The efficiency of P fertilisers (for plant uptake) is generally between 11 percent and 38 percent (Withers and Sharpley, 1995) and almost 70 percent of P consumed by cattle and over 80 percent consumed by sheep in concentrated feeds, fresh grass and silage is excreted (Haygarth et al., 1998). The surplus P accumulates in the soil and can potentially be lost and transported to water bodies. This can happen due to soil erosion in the form of sediment loss or via surface runoff (Merrington et al., 2003, Sharpley and Tunney, 2000, Withers, 1996). The accumulation of P levels in soils over the last 50+ years also had an impact upon plant species diversity in grassland meadows and semi-natural habitats (Merrington et al., 2003).

Aquatic systems are very sensitive to excess inputs of P (Conley et al., 2009). In the EU, it has been suggested that as much as 50 percent of the anthropogenic P entering aquatic systems is from urban waste-water (Mariën, 1997), with a strong correlation between population density and P concentrations in river waters. However, these inputs of P from point sources are easier to identify and so industrial and sewage discharges are brought into line under the Urban Waste Water Treatment Directive (Council Directive 91/271/EEC). Yet there is evidence that suggests that these reductions of P input from the point sources have had little effect on P content in aquatic systems, suggesting diffuse sources of P (or possibly P release from sediments) in the water bodies (Boyd, 2000, Merrington et al., 2003).

**PESTICIDES**

Pesticides have been used since Roman and Greek times. But it is the development of the chemical industry during the 1930’s and 1940’s that ensured commercial
availability of pesticides within the following 30-40 years (Merrington et al., 2003). They are used for crop production as well as in a range of veterinary products to control parasites in livestock farming (Bigler et al., 1992). Today there are more than 30 classes of chemicals with pesticidal properties that help to protect crops from natural parasites (weeds, fungi, insects, etc.) at a relatively low cost (Merrington et al., 2003, Ritter and Shirmohammadi, 2000).

Ideally, a pesticide would only affect its target organism; it would be non-persistent and have no polluting effects on the environment. But in reality pesticides often pollute the environment by affecting non-target organisms directly or in the form of residues or persistent compounds. According to Pimentel (1995) less than 0.1 percent of applied pesticides reach their target organisms (Pimentel, 1995).

Paris et al. (1981), Paris and Lewis (1973), Paris (2011) and Novotny (2003), among others found that the newer phosphate insecticides have half-lives in soils ranging from 1 to 30 days and become greater environmental hazards after the break-down (Paris and Lewis, 1973, Paris et al., 1981, Parris, 2011, Novotny, 2003). As a result of such persistence, ground and surface waters are sometimes polluted by pesticides (Merrington et al., 2003). The major losses of pesticides to the environment are through volatilization into the atmosphere and aerial drift, runoff to surface water bodies in dissolved and particulate forms, and leaching to groundwater (Merrington et al., 2003, Ritter and Shirmohammadi, 2000).

**Soil Erosion**

Soil Erosion is often connected to crop production and associated ploughing (Montgomery, 2007, Pimentel et al., 1995). However, by itself, soil erosion is a natural process that results from natural land denudation and depends on the potential of 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., 2003, Novotny, 2003).

Soil erosion affects water bodies through increased sedimentation and through increased concentrations of chemical pollutants, such as pesticides and nutrients attached to soil particles. The deposition of sediment creates anaerobic conditions
that favour nitrate and ammonia production, which is particularly toxic to salmonoids. Sediment also reduces oxygen availability to fish eggs buried within the reeds which also contributes to mortality (National Research Council, 1993, Merrington et al., 2003). The soil loss from farmland to adjacent watercourses can have detrimental impact on water quality (Ockenden et al., 2012). Most authors classify these impacts as diffuse pollution, as major erosion events tend to affect large areas of land within a given catchment (Merrington et al., 2003). There are of course other human activities that are responsible for sediment deposition in the water bodies. For example, the soil loss from construction sites can reach a magnitude of over 100 tonnes per hectare affecting the habitat of different aquatic organisms. Forest harvesting operations can also increase the sediment level in surface water systems resulting in reduced in-stream photosynthesis and may destroy benthic organisms and fish eggs. Turbid water is also less aesthetically pleasing, which can make some water bodies unattractive for recreational uses (Boyd, 2000, Novotny, 2003).

As erosion is a natural process and sediment is moved as a result of natural denudation, it is impossible to eliminate sediment loss. However, it is possible to control excessive sediment loadings that result from land-use activities. Wade and Heady (1978) illustrate that controlling sediment loss from agriculture would come at an extra cost to farmers (Wade and Heady, 1978). However, they also cite their findings from 1973 where they calculated an annual damage caused by soil loss from agriculture in the USA of 1 billion US dollars (Wade and Heady, 1978). Another study by Ouyang et al., (2005) notes that the cost of sediment dredging projects of $20 million in the Great Lakes each year (Ouyang et al., 2005).

**PATHOGENS**

Another pollutant from agricultural activities is pathogens. Pathogens are pollutants that are harmful for humans that often come from contamination of water bodies by animal manure through leaching or runoff. Ritter and Shirmohammadi (2000) note that animal waste is a source of approximately 150 diseases, which include typhoid fever, gastro-intestinal disorders, cholera, tuberculosis, anthrax, and mastitis, hog cholera, foot and mouth disease, polio, respiratory disease, eye infections, protozoan parasites and cryptosporidium (Ritter and Shirmohammadi, 2000). Cryptosporidium
for example, is resistant to many of the procedures used to treat drinking water and is responsible for disease outbreaks (Callaghan et al., 2009, Graczyk et al., 2008, Merrington et al., 2003). Graczyk et al. (2008) report that in Ireland the most common source of bacteria in water is related to spreading of animal manure and sewage sludge on agricultural land. They also list the possible sources of pollution as wastewater discharges, sewage seepage to surface waters, operational deficiencies at wastewater treatment plants, septic tank malfunctioning, and the leisure industry discharging wastes overboard (Graczyk et al., 2008). The contamination of drinking water with cryptosporidium is associated with heavy rainfalls, stream flow and flooding (Ailes et al., 2013, Graczyk et al., 2008, IPA, 2009).

Thus, there are a number of pollutants from agricultural activities that can be lost to the aquatic environment and a number of regulations on international and national levels were introduced in attempt to reduce the loss of pollutants from agriculture and protect water resources.

II.4 Policies and regulations

HISTORY

Pollution is an economic externality caused by the failure of the market to include in the value of the economic output the cost of damage caused by disposal of pollutants into the environment. These concerns created the rationale for government intervention on national and international levels (Pretty et al., 2001, Shortle and Abler, 2001, Novotny, 2003, Tietenberg, 2006).

The level of water pollution that exists today and the growing concern regarding such levels of pollution signals that there is a market failure to determine the acceptable level of pollution. Novotny (2003) writes that environmental policies should not always be guided by economic principles but rather by common sense and a need to protect the health of humans as well as to preserve the environment and resources for future generations (Novotny, 2003). However, in times of economic crisis, limited resources and numerous mitigation options available for consideration, there is a growing need for efficient resource allocation based on economic principles.
Policy-makers have a number of options to choose from in designing a response to polluting activities: no action; moral persuasion and public pressure; court litigation; economic and government contributions in research, education, water body rehabilitation, and preservation (Novotny, 2003). Usually a mixture of all of these exist depending on the type and the level of pollution, existing levels of technology, and the political and institutional framework in a given country. The task of the policy-makers is a difficult one as environmental legislation is usually strongly opposed by lobbying groups (Kaika, 2003).

Despite the strong opposition from different lobbying groups a number of regulations regarding water pollution were developed on an international, EU, regional, country and local (within country) level. International agreements, declarations and resolutions govern water resource management in different dimensions and the most important of them is the UN Convention on the Law of the Non-navigational Uses of International Watercourses that was adopted by the UN General Assembly in resolution 51/229 of 21 May 1997. On the EU level, there are European directives that are legally binding, directly applicable in the Member States and have to be transposed into national law with specific deadlines. These have to be met or the country will be legally pursued in the European Court of Justice (Kissling-Näf and Kuks, 2004).

In the development of the European water protection legislation three distinct phases can be identified:

- (1973-1988) the focus is on water used for humanity and on determining water standards;
- (1988-1995) the focus is on specific measures introduced in a command-control fashion;
- (1995-present) the focus is on an integrated approach to water resource protection (Kissling-Näf and Kuks, 2004).

On the EU level there are European Directives, agreements of the United Nations, Economic Commission for Europe and the European Frontier Water Agreements. Although EU Member States may address water problems using internal legislative devices, there are 12 main EU Directives that form a legal framework concerning environmental regulation:
Directives of the European Parliament and of the Council include:


Directives of the European Economic Community include:


The Urban Waste Water Treatment Directive (91/676/EEC)

The Urban Waste Water Treatment Directive aims to protect the environment from the adverse effects of collection, discharge and treatment of urban waste water and water emitted by some industrial sectors. The 2011 report on implementation of this directive indicates that waste water treatment in EU is still not applied to 100 percent of waste water (in some countries as low as 48 percent) despite the improvements in implementation (European Commission, 2011). One of the main reasons of the poor implementation of the Directive is the extremely high cost of implementation.

The Drinking Water Directive (80/778/EEC)

The Drinking Water Directive aims to protect human health by setting the standards of Drinking Water (excluding natural mineral water and waters which are medicinal products). The standards are set according to the precautionary principle and the Member States are required to monitor the quality of drinking water on the regular basis and publish reports every three years. However, this Directive aims to regulate the end-pipe water quality instead of managing the quality of water all the way from the basin to the tap (Council Directive 80/778/EEC).

The EC Nitrates Directive (91/676/EEC)

The EC Nitrates Directive (ND) is aimed at the protection of water resources (drinking water as well as water for other needs) from pollution caused by the application and storage of inorganic fertilisers and manure in agriculture. ND highlights the importance of fertiliser usage but recognises agricultural emissions as the main source of diffuse pollution. This Directive requires Member states to
identify vulnerable zones ($\geq 50\text{mg NO}_3/l$) unless they are willing to apply ND programmes through their national territories, in which case there is no need to identify vulnerable zones; to establish and implement action programmes that encourage good agricultural practices; and set specific limits of farm land application of fertiliser and manure (Council Directive 91/676/EEC).


In spite of the growing number of legislative documents concerning water quality and protecting water bodies from polluting human activities, the persistence of the problem signalled the need for a comprehensive framework that would target all aspects of water quality. The WFD is intended to fulfil this need. It aims at maintaining and improving the aquatic environment in the European Community through protection of inland surface waters, transitional waters, coastal waters and ground water.

Under the requirements of the WFD countries are given stringent deadlines to set up policies to ensure the efficient and sustainable use of water resources through identifying river basins and river basin districts and putting in place appropriate administrative arrangements for their management. Member states are required to establish laws, regulations and administrative provisions in order to accommodate the process of meeting WFD standards.

The WFD does not only require regulation of quantity and quality of water, but it also obliges the member states to ensure recovery of costs for water services based on the polluter-pays principle and via pricing policies to provide adequate incentives for users to conserve water resources. The Directive identifies main protection areas and lists the main pollutants to be regulated. The countries are also given deadlines for reporting and the implementation of the Directive.

The WFD helped to attract the attention of many researchers with its broad cover of water-related issues. It is considered to be one of the most significant legislative documents in the water field that was introduced on an international level for many years (Dworak et al., 2005). There are of course those who are pessimistic about the Directive. For example, Lichtenstein (1998) termed the Directive as “fall of an integrated EU water policy”; Kallis and Butler (2001) criticise the protection of
groundwater in WFD. They point out that unlike the groundwater directive (Council Directive 80/68/EEC) that was mandated to “prevent direct and indirect input” of regulated substances into groundwater, within the WFD, member states are given an option to “limit” rather than to “prevent” input of these substances (Kallis and Butler, 2001). Moss (2008a) calls WFD a “political compromise”, he claims that provisionally it was established that most water bodies in the UK, Central and Eastern Europe would fail to meet good ecological status, it is the change in the water bodies topology classification that allowed to initially mitigate against the reality of ecological statuses and relax the definitions of classes for water bodies assessment. He points out two major flaws of the Directive: 1) the vagueness of definitions (for example, “good status” is defined as slightly different from high, without explaining what this “slightly” is); 2) the perception that ecological status can be measured in terms of simple ratios (number of fish etc.) (Moss, 2008a). Moss (2008a) also notes that the restoration of water bodies to their pristine conditions is only possible in some remote areas in Europe or, as he jokes, if most of people become extinct, thus, putting under question attainability of the WFD’s goals.

Another point of criticism of the WFD is the cost and enforceability of the Directive. It was expected from the very beginning that the cost of implementation of the Directive would be high. Kallis and Butler (2001) argue that cost-prohibitive implementation of WFD will be difficult especially for less well-off member states. The main concern of economic analysis from the beginning was to find the most cost effective measures of reaching WFD objectives. In November 2008 Goodbody Economic Consultants issued the “Guidance on Implementing Cost Effectiveness of Measures in the Context of Water Framework Directive” in which they gave a quick review of WFD and recommendations on cost-effectiveness analysis (Goodbody Economic Consultants, 2008).

There is a concern that most of the goals of the Directive are not legally enforceable and there are enough legislative loopholes that unwilling member states can exploit to avoid implementation and to reduce the unexpectedly high costs (e.g. member states will not be in breach of the directive if the objectives cannot be achieved for reasons of “technical feasibility” or “disproportionate cost”). On the positive side, the directive has enough legal capacity to establish important institutions that would
trigger the changes toward better water quality, at least there are concrete standards of no further deterioration for any water body (Kallis and Butler, 2001).

As has been already noted above, the Directive has economic implications as it calls for the application of economic principles for pricing water and using pricing mechanisms as economic incentives for more efficient use of water resources. Cost recovery for water consumption and services is a very sensitive issue. Economic theory requires that users are charged prices that reflect the costs at the margin of supplying this good or service (Joyce and Convery, 2009). WFD requires member states to charge on the basis of the polluter-pays principle. The main issue for many member states is to what extent to include environmental and social costs into pricing mechanisms. The dilemma that member states face is not only due to the heavy subsidised agricultural sector in many countries but due to the lack of comprehensive data and methodology needed to estimate social and environmental resource damage cost especially as a result of nonpoint pollution.

As many of the deadlines of WFD implementation approach, more and more scientists and economists are getting involved in research, primarily studying the aspects of nonpoint pollution focusing on agriculture. Some studies suggest that to meet WFD goals, management of agricultural practices will have to be changed drastically. It is suggested in the UK studies that the application of fertilisers to crops and grass may have to be reduced by 50 percent, sheep stocking rates may need to be halved and cattle stocking rates reduced by 25 percent (Bateman et al., 2006, Haygarth et al., 2003). Volk et al. (2009) developed a simulation model to examine the measures needed to achieve good ecological status. Their result showed that drastic measure are needed in the catchment of their study: 1) arable land reduction by over 30 percent, 2) increase in pasture land by 11 percent, 3) 11 percent increase in afforestation and 4) increase in protected wetland from 0 percent to 9 percent (Volk et al., 2009). These suggested restrictions immediately raise the question of the economic implications of possible future policies.

II.5 Conclusion

The fast growing population on planet Earth puts more and more pressure on the environment with pollution from human activities increasing and threatening the
wellbeing of future generations. These developments lead to the need to protect natural resources in general and water resources in particular. Water is one of the most important socio-economic resources, which is essential for the survival of all organisms, as well as being an important public good. However, until the 1970’s little effort was expended on the protection of this valuable resource. This signalled a market failure as markets failed to internalise the cost of the damage caused by pollution and through its mechanisms to balance pollution.

There are many sources of pollution which can be broadly divided into two groups – point source pollution and non-point source pollution. Pollution from point sources is easier to control once ambient or effluent (emission) standards are established. On the other hand, it is hard to control and regulate diffuse sources as by their nature it is hard to establish the direct link between the particular human activities and the water quality at a specific point. The most discussed sources of diffuse pollution in water are construction, street runoff, septic tanks and agriculture. All of these sources are known to have a potential to emit undesirable substances that can end up in the water bodies and cause negative effect. Agriculture is the area of the most concern because feeding the growing population demands the increase in agricultural production in Ireland by about 30 percent by 2020 (DAFM, 2010). Recently the added duty of agriculture became to produce raw material for biofuel industry. On the other hand, there is a pressure on agriculture to reduce loss of potential pollutants from its activities. The potential pollutants from agriculture include nutrients (nitrogen and phosphorus), pesticides, sediment and pathogens. It is difficult to regulate pollution from agricultural land for a number of reasons: first, it is hard to establish the direct link between activities on the farms and their effect on the water bodies in the adjacent area; second, farmers in Ireland operate on small margins and a reduction in agricultural activities may lead to a loss of these margins.

be brought to “good status” by 2015 and stringent penalties for countries for non-compliance. It is also the first document of the kind that recognises the need to take integrated approach of water resources protection, the need for holistic approach. The over-border cooperation is also recognised as an important element of the water protection. The Nitrates Directive is important document for agricultural sector.

In the face of limited economic and environmental resources there is a growing need for more sustainable management of natural resources and water is one of the most important of them. The costs have to be assigned on the polluter pays principle and the pollution mitigation targets should be achieved in the most cost efficient way. This requires knowledge about the sources of pollution, the cost of the damage and the cost of possible mitigation measures. There is still a gap in the literature and more research is needed urgently to close this gap in the research.
Chapter III.

AN EXPLANATORY ANALYSIS OF THE RELATIONSHIP BETWEEN HUMAN ACTIVITIES AND WATER QUALITY: A SPATIAL APPROACH

In this chapter an exploratory analysis of the relationship between the water quality and human activities upstream is conducted.

III.1 Introduction

This chapter is concerned with ambient water quality and status as defined by the Water Framework Directive (WFD) (Directive 2000/60/EC). One of the 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 (EPAI, 2014). This would require co-operation on a European and international level in cases where a river basin lies on the territories of more than one country.

Since the introduction of the WFD in 2000, a lot of effort has been made by the Member State countries to put together national level regulation on the protection of water resources. The standards have been set up, the targets identified and the catchment management plans put in place (Phillips, 2014). Despite this progress there is still an on-going debate about the sources of pollution in water streams and, thus, there is still no robust evidence to help policy-makers in their decision making. The point sources of pollution have been successfully dealt with and Environmental Protection Agency Ireland (EPAI) licence the industries discharging water into the streams (EPAI, 2011). However, the insufficiently slow improvement in the ambient water quality raises a lot of questions about the sources of diffuse pollution and how to mitigate them.

There are numerous sources of diffuse pollution in the water bodies that include: urban run-off, septic tanks, agricultural activities and soil erosion among others (Tietenberg, 2006). However, the main focus of research and regulatory discussions has been on agricultural activities (O’Shea and Wade, 2009, Blackstock et al., 2010, Kastner et al., 2011, Volk et al., 2009). In Ireland, as in many other countries around the world, farmers produce high quality food on very narrow profit margins. At the same time, they are the target of a number of environmental regulations that affects their activities. For example, the Nitrates Directive (Council Directive 91/676/EEC)
and the Good Agricultural Practices (European Communities, 2010) require a number of changes in the farming practices and impose a number of restrictions on the amount of organic and inorganic fertiliser that can be used by farmers; on the stocking rates and the methods/times when fertiliser can be spread. With the Food Harvest 2020 (DAFM, 2010) agenda dictating an increase in agricultural production to feed an exponentially growing Earth population, it is time to explore the polluting flows from all contributing sources. Moss (2008b) states that almost anything that happens on the catchment can affect the water quality. He compares the relationship between catchment and receiving water to that of a house and a waste bin (Moss, 2008b).

Many factors 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 (Donohue et al., 2006, Merrington et al., 2003, Novotny, 2003, Ritter and Shirmohammadi, 2000).

Besides agriculture, there is a wide variety of other human activities that can have a significant negative impact on water quality. Donohue et al. (2006) examined the effect of a broader variety of factors on river water quality. 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, Donohue et al. (2006) 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. A more recent Irish study by Curtis and Morgenroth (2013) examined the relationship between human activities and the water quality in 286 Irish lakes. They concluded that there is a statistically significant non-linear relationship between human activities, environmental conditions in the catchments and water quality in the relevant lakes. 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 fertiliser use were all negatively associated with water quality.

Urban pollution is one of the leading sources contributing to pollution in streams and the wider environment but for some reason is overlooked in the research on water pollution. The increasing population of the Earth is putting more pressure on ecosystems each year by directly polluting the environment or increasing demand for ecosystem services. 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 an 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 (2006) report that urban runoff may “prejudice” “good” ecological potential of the water bodies.

As a result of population growth there is a growing demand for construction services, these are one of the leading sources of soil erosion. Soil Erosion is a natural phenomenon, however, soil erosion and the resulting sediment pollution may have origins in certain land use activities such as construction and ploughing (Boyd, 2000, Merrington et al., 2003, Novotny, 2003). As has already been discussed in the previous chapters of this thesis, soil erosion causes a number of problems in water bodies.

Another important source of diffuse water pollution that is often over-looked in the environmental debates is septic tanks. The introduction of the EU Waste Framework Directive (Directive 2006/12/EC) is very relevant to Ireland as its main features and cultural identity is a large number of dispersed farms and rural dwellings. Usually urban households are not connected to sewage treatment systems and as a rule have individual septic tanks installed and some of these septic tanks are very old or installed and managed incorrectly (Naughton and Hynds, 2014). It is reported more and more often that septic tanks that are installed and managed incorrectly cause the ground water pollution resulting in a large number of untreated chemicals and bacteria entering the environment (Macintosh et al., 2011, Swarup et al., 1992,
Withers et al., 2011). Ireland has over 400,000 septic tanks throughout the territory, which is suspected to contribute to the problem of the water pollution (EC Ireland, 2011).

These are all known as diffuse sources of pollution. The diffuse sources of pollution are not only difficult to identify but also to control as they often have a spatial and temporal dimension (Donohue et al., 2006, Donohue et al., 2005). They are also dependent on environmental interactions between the elements, temperature and soil (Schulte et al., 2006) and as such they may be impossible to control. In addition to the sources of diffuse pollution, it is useful to note that diffuse 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 Donohue 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 were 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. Moreover, most studies do not take into consideration spatial factors like geographical location, temperature variation and soil quality that may contribute to geographic variation in pollution. Spatial studies may help to identify places where policies to mitigate pollution should be targeted. This chapter adds to the literature by combining a number of spatial datasets relating to agricultural activities and septic tanks density with environmental data 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 2010 Irish Census of Agriculture (COA) which provides spatial information relating to agricultural
activity, the 2011 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. The relationship between water quality, agricultural activities, septic tanks density and afforestation are the main focus of this study. While agricultural activities and septic tank densities are known to be negatively associated with water quality, afforestation is usually associated with ecosystem services and pollution mitigation, thus, to be positively associated with water quality. The analysis conducted in this chapter adds to the existing literature in an attempt to identify the contributing sources of pollution in Irish rivers controlling for spatial and environmental differences.

The rest of the chapter is structured as follows: the data sets employed in this chapter are described in section III.2; the methodology is explained in section III.2; and the results are presented in section III.4 and their discussion is given in section III.5.

III.2 Data

In order to model the relationship between water quality and upstream economic activity, data in relation to water quality, agricultural intensity, settlement intensity (specifically the use of septic tank based waste water treatment), environmental variability and economic intensity is required.

In this section the data used to conduct the analysis in this chapter 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 septic tank distribution data from the small area Census of Population (SAPS), levels of agricultural activity from the Census of Agriculture, forest land cover data from the Forest Service and environmental data.

THE EPA WATER QUALITY CLASSIFICATION SYSTEM

The WFD directs the Member States to conduct biological, hydro-morphological and chemical assessment of the water body status (EPAI, 2007). In addition to these indicators, 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 (physicochemical), and hydro-morphological 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 III.1) (EPAI, 2007).

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

<table>
<thead>
<tr>
<th>Biotic Index (Q)</th>
<th>Status</th>
</tr>
</thead>
<tbody>
<tr>
<td>Q5, Q4-5</td>
<td>High</td>
</tr>
<tr>
<td>Q4</td>
<td>Good</td>
</tr>
<tr>
<td>Q3-4</td>
<td>Moderate</td>
</tr>
<tr>
<td>Q3, Q2-3</td>
<td>Poor</td>
</tr>
<tr>
<td>Q2, Q1-2, Q1</td>
<td>Bad</td>
</tr>
</tbody>
</table>

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

In Ireland, the Quality Rating System has been used to monitor the ecological quality of streams and rivers since 1971 (Donohue et al., 2006, Flanagan and Toner, 1972, McGarrigle et al., 2002). Over 3000 sites on 13,200 km of main river channel are included in the current national survey and assessed using the Quality Rating System to characterise water quality (EPAI, 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 or excessive plant growth in Irish waters. 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 WFD (Donohue et al., 2005). The Q-values from a set of 2548 river sites that were monitored by the EPAI 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 in this chapter, 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. The shares of the Q-values for year 2011 are reported in Table III.2.

<table>
<thead>
<tr>
<th>Q-value</th>
<th>Share (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.2</td>
</tr>
<tr>
<td>2</td>
<td>2.0</td>
</tr>
<tr>
<td>3</td>
<td>11.4</td>
</tr>
<tr>
<td>4</td>
<td>68.3</td>
</tr>
<tr>
<td>5</td>
<td>18.1</td>
</tr>
</tbody>
</table>

Almost 90 per cent of Irish rivers in 2011 were given Q4 or higher, which still leaves over 13 percent falling short of the level required for the WFD Q4. This indicates that more effort is necessary to achieve high water quality in many river sites.

**THE IRISH CENSUS OF AGRICULTURE**

The second dataset used in this chapter is the COA. The objective of the census is to collect data related to agricultural activities on all farms within Ireland (CSO, 2002). This data is collected every ten years with the most recent data set available is for year 2010. 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) 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). In this chapter COA for 2010 is combined with the Census of Population for 2011.

The specific variables taken from the COA include the proportion of farmland in each ED under crops, the number of pigs per hectare in each ED and finally the 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 fertilisers and manure and urea deposited by grazing animals (Monteny et al., 2006). The figures for

---

2 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).
livestock density were combined with the EPAI conversion factors for different livestock types to produce an estimate of organic nitrogen produced per ED. The average organic N per hectare and associated Q values downstream for 2011 are reported in Table III.3. It is easy to see that the higher the water quality index, the lower the average organic N per hectare in the upstream EDs within the catchment.

Table III.3. Average Organic N per Q-Value 2011

<table>
<thead>
<tr>
<th>QV</th>
<th>Organic N per hectare (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>108.0</td>
</tr>
<tr>
<td>2</td>
<td>90.7</td>
</tr>
<tr>
<td>3</td>
<td>89.5</td>
</tr>
<tr>
<td>4</td>
<td>92.8</td>
</tr>
<tr>
<td>5</td>
<td>88.8</td>
</tr>
</tbody>
</table>

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. Table III.4 represents the share of the land under cereal production per Q-value in year 2011. As can be seen, there is no obvious relationship between the water quality and the share of land under production. However, it is still evident that the higher water quality points (Q4 and Q5) are associated with the areas where lower shares of land are under cereal production.

Table III.4. Average Cereal Land Use Share per Q-Value 2011

<table>
<thead>
<tr>
<th>QV</th>
<th>Share of crop land</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>5.3</td>
</tr>
<tr>
<td>2</td>
<td>6.3</td>
</tr>
<tr>
<td>3</td>
<td>9.2</td>
</tr>
<tr>
<td>4</td>
<td>6.1</td>
</tr>
<tr>
<td>5</td>
<td>1.9</td>
</tr>
</tbody>
</table>
**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 included in this analysis is a combined variable for mature forestry and immature forestry and scrub. This forest cover GIS data has been updated by (Farrelly, 2007) to reflect spatial changes in forestry cover in Ireland in recent years. The forest cover data used in this chapter is representative of forestry in Ireland in 2007. 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 (Novotny, 2003).

**CENSUS OF POPULATION, SMALL AREA POPULATION STATISTICS (SAPS)**

The Central Statistics Office (CSO) as part of the National Census of Population collects 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 facilities, the EPAI (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.

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.

**ENVIRONMENTAL DATA**

The dataset employed in the analysis described in this chapter has further been augmented by adding environmental variables – an amount of rain, elevation and
average temperature. These are known to affect the water quality though either increased runoff, volatilisation and/or accumulation of pollutants during dry/wet seasons (Huang et al., 2014, Novotny, 2003).

Median elevation per ED was generated from an Irish 20 m resolution digital elevation model (DEM) derived originally from Ordnance Survey Ireland 1:50,000 maps (Preston and Mills, 2002). Average annual rainfall and temperature per ED were derived from the models developed by Sweeney and Fealy (2003). These models employed monthly precipitation and temperature data from 650 and 70 stations, respectively, across the island of Ireland between the years 1961 and 1990 to develop average annual climactic maps at a 1km resolution. Zonal spatial statistics for elevation and the climactic variables were generated for each ED using ArcGIS 10.1.

II.3 Methodology

In this chapter a non-linear parametric approach - an ordered probit model - is used to estimate the relationship between water quality and human activities upstream controlling for spatial and environmental differences. This approach is chosen due to the fact that the dependent variable – the EPA Q-value system - is an index from 1 to 5 that expresses 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 highest water quality status followed by 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. For example, it may take a bigger change in the value of an independent variable to surpass the “first threshold” into the second category than it takes to surpass the “second threshold” to transfer from the second category into the third 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.

The parametric approach and regression analysis has been previously used in another Irish study on water quality. Curtis and Morgenroth (2013) used regression analysis to examine the effect of the human activities on the water quality in Irish lakes. However, unlike in this study, their study used linear and non-linear models. The use
of the concentrations of chlorophyll variable as an indicator of the water quality allowed them to utilise linear and non-linear regression analysis for estimations as this is a continuous variable. Similar to Curtis and Morgenroth (2013) characteristics such as physical land use, septic tank densities and economic (agricultural) activity of the river catchments were used as independent variables in this analysis.

Denoted $X_i$, these variables 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, $i = \{1, \ldots, n\}$. $Y_i$ is a scalar that takes the values of 1, 2, 3, 4 and 5. Larger values indicate higher water quality. $Y$ is an ($n \times l$) 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, $k = \{1, \ldots, K\}$. $X$ is an ($n \times k$) matrix summarizing each river catchments economic and land use characteristics. The $n^{th}$ row indicates the characteristics of the $n^{th}$ catchment. Therefore, it can be stated that:

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

Since the dependent variable is an ordered, qualitative variable, the relationship between $Y$ and $X$ with an ordinal response (an ordered probit) model is estimated. In ordered probit models, an underlying score is estimated as a linear function of the independent variables. A set of cut-off points are produced and indicate the category estimated for each monitoring point. The probability of observing outcome $i$ corresponds to the probability that the estimated linear function, plus random error, is within the range of the cut-off points estimated for the outcome (STATA.com).

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 ($k \times 1$), denoted $\beta$, and a disturbance term, $\varepsilon$, 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:

\[
\Pr(Y_i = 1) = \Phi(\mu_1 - \beta^\prime X)
\]

\[
\Pr(Y_i = 2) = \Phi(\mu_2 - \beta^\prime X) - \Phi(\mu_1 - \beta^\prime X)
\]

\[
\Pr(Y_i = 3) = \Phi(\mu_3 - \beta^\prime X) - \Phi(\mu_2 - \beta^\prime X)
\]

\[
\Pr(Y_i = 4) = \Phi(\mu_4 - \beta^\prime X) - \Phi(\mu_3 - \beta^\prime X)
\]

\[
\Pr(Y_i = 5) = 1 - \Phi(\mu_4 - \beta^\prime X)
\]

where the \(\mu\)'s are unknown threshold parameters (cut-points) to be estimated with \(\beta\), and the ranking depends on certain measurable factors \(x\) and certain unobservable factors \(\epsilon\). Since the disturbances are normally distributed, these probabilities are distributed according to the cumulative normal distribution. The ordered probit model is estimated using the method of maximum likelihood via the Newton-Raphson algorithm (Long, 1997). Two models are estimated for comparison:

- prob \((Q) = f\) (septic tank density, share of cereal farming, mean organic N per hectare, number of pigs per hectare, poultry number per hectare, forest cover, landfill within 3 km, county dummies, catchment area)

- prob \((Q) = f\) (septic tank density, share of cereal farming, mean organic N per hectare, number of pigs per hectare, poultry number per hectare, forest cover, county dummies, amount of rain, temperature, mean elevation, x coordinate, y coordinate).

The difference in the two models stated above is the environmental and spatial components that are added to the second model. The additional variables included in the second model are the amount of rain, temperature, slope and x and y coordinates to account for environmental and spatial variability.

The spatial scale of the analysis is an electoral district (ED). Independent variables in this analysis are the average of the five nearest EDs (weighted by an ED’s size relative to the total catchment area) that are located upstream from the EPA water quality points within the river catchments.
III.4 Results

In this section the results of the estimates are presented.

The results of the estimates for two models are presented in Table III.5. In the ordered probit models the sign of the coefficients tells one when \( X \) increases it is more likely to have a high value for \( Y \) if the coefficient is positive, or more likely to have a small \( Y \) if the coefficient is negative. The size of the coefficients themselves cannot be interpreted. It has to be noted that for ordered probit models one can get a marginal effects (ME) for the effect of the \( X \) on the probability of \( Y=1, Y=2, Y=..., \) etc., the ME depends on the values of the other \( X \)'s, thus, the MEs (which are highly informative in OLS models) for this model are reported in later this chapter.

**Table III.5. Ordered Probit Regressions**

<table>
<thead>
<tr>
<th></th>
<th>Model 1</th>
<th></th>
<th>Model 2</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Coef.</td>
<td>S.E.</td>
<td>Coef.</td>
<td>S.E.</td>
</tr>
<tr>
<td>Septic Tank Density</td>
<td>-0.0003*</td>
<td>0.0002</td>
<td>-0.0003*</td>
<td>0.0002</td>
</tr>
<tr>
<td>Cereal Share of Land Use</td>
<td>-1.6687***</td>
<td>0.2246</td>
<td>-1.2632***</td>
<td>0.3446</td>
</tr>
<tr>
<td>Organic N Density</td>
<td>-0.0014*</td>
<td>0.0008</td>
<td>-0.0044***</td>
<td>0.0012</td>
</tr>
<tr>
<td>Pigs per KM² (Top Quartile)</td>
<td>-0.0965</td>
<td>0.1190</td>
<td>-0.0303</td>
<td>0.1243</td>
</tr>
<tr>
<td>Poultry per KM²</td>
<td>0.00001***</td>
<td>0.00002</td>
<td>3.8x10⁻⁶</td>
<td>3.9x10⁻⁶</td>
</tr>
<tr>
<td>Cumulative Afforestation</td>
<td>0.0014***</td>
<td>0.0003</td>
<td>0.0014***</td>
<td>0.0003</td>
</tr>
<tr>
<td>Environmental Characteristics</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rainfall</td>
<td>0.0004</td>
<td>0.0003</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average Temperature</td>
<td>0.1424</td>
<td>0.0970</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Median Elevation</td>
<td>0.0035***</td>
<td>0.0006</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Landfill within 3km</td>
<td>-0.0425</td>
<td>0.2393</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coordinates</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>x_coord</td>
<td>-2.35x10⁻⁶</td>
<td>1.57x10⁻⁶</td>
<td></td>
<td></td>
</tr>
<tr>
<td>y_coord</td>
<td>2.19x10⁻⁶</td>
<td>1.40x10⁻⁶</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pseudo R²</td>
<td>0.0175</td>
<td>0.0921</td>
<td></td>
<td></td>
</tr>
<tr>
<td>N</td>
<td>2146</td>
<td>2139</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*** significant at 1% level; ** significant at 5% level; * significant at 10% level; NB County Dummies are not reported

Septic tank density proved to be significant at 1 percent level in both models, the coefficient stays the same even when the environmental variables are added. It has a negative sign on the estimator in both models indicating that the higher sceptic tank density is associated with lower water quality downstream. This is in line with the results of the estimates of Curtis and Morgenroth (2013) whose statistically significant estimates indicate the higher septic tank densities in a lake’s catchment are...
associated with higher chlorophyll amounts in water thus lowering its quality. The same is true with variables indicating an intensity of agricultural activities in the catchments – the share of the cereal land use and the amount of organic N per hectare – which are also statistically significant variables with the negative coefficients indicating that these activities have negative impact on water quality in the downstream areas, which is in line with findings of Curtis and Morgenroth (2013) and Donohue et al. (2006) who also established that agricultural activities in the catchment are associated with the lower water quality.

Presence of poultry farms yields a close to zero statistically significant coefficient in the first model estimated in this paper but becomes statistically insignificant when environmental variables are added to the model; while the coefficients on the pigs per hectare variable are negative but statistically insignificant in both models. Unsurprisingly, an afforestation has positive statistically significant at 1 percent level estimate signalling that areas of afforestation may have positive effect on the water quality in the downstream area according to this study. It has been long suggested that afforestation plays important role in delivering ecosystems services, which led to active policy efforts in promoting afforestation in Europe (Upton et al., 2014). This is in line with findings of Donohue et al. (2006). It is interesting that in Curtis and Morgenroth’s study the coefficient signs on the pigs and forest variables are opposite to those reported in this chapter, which is difficult to explain. There is some evidence that commercially grown for logging and timber forests may be a source of polluting activities rather than pollution mitigating activity (Novotny, 2003), however, it is hard to say if this is the case in their study.

Relative to the land-use variables, it is more difficult to capture an effect of environmental variables on water quality. This may be due to the fact that it is not the average frequency of an environmental event that matters most but rather its intensity that may be important. There is also a high variation of impacts of extreme weather events spatially and temporarily, which is hard to capture in the model presented in this chapter. The amount of the rainfall estimate is positive but statistically insignificant according to the model reported here. The same is true about the average temperature. These estimates are in line with the other Irish studies by Curtis and Morgenroth (2013) and Donohue et al. (2006). These results are in opposition with
the textbooks of Novotny (2003), Ritter and Shirmohammadi (2000) and similar studies that normally associate high rainfalls with runoff events and pollution. The intuitive explanation of the finding in this chapter’s model and other Irish studies is that high rainfall events in Ireland may lead to “dilution” of water in lakes and rivers, thus, reducing the concentrations of polluting substances. It is, however, harder to come up with an intuitive explanation for the sign of the elevation variable estimate. The elevation variable yields a significant but positive coefficient (in this study and studies by Donohue et al. (2006) and Curtis and Morgenroth (2013)) which is in opposition to existing literature which states that the higher the slope the higher the run-off volumes (Novotny, 2003). This estimate may be reflecting the fact that the higher elevated areas in Ireland are the areas of lesser human activities. Having the landfill within three kilometres upstream from the water sampling point proves to have a negative effect on the water quality; however, the result is statistically insignificant.

Both models estimated confirmed the hypothesis that agricultural activities are associated with lower water quality regardless of whether a particular model include or does not include environmental variables. It is however worth to examine MEs that are reported in Tables III.6 and III.7.

**Table III.6. Marginal Effects for Model 1**

<table>
<thead>
<tr>
<th>Variable</th>
<th>Q1</th>
<th>Q2</th>
<th>Q3</th>
<th>Q4</th>
<th>Q4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Septic Tank Density</td>
<td>4.87x10^{-7}</td>
<td>1.88x10^{-7}</td>
<td>4.96E x10^{-5}</td>
<td>-3.1 x10^{-5}</td>
<td>-8.19 x10^{-6}</td>
</tr>
<tr>
<td>Cereal Share of Land Use</td>
<td>0.0025</td>
<td>0.0098</td>
<td>0.2592</td>
<td>-0.1606</td>
<td>-0.0428</td>
</tr>
<tr>
<td>Organic N Density</td>
<td>2.18 x10^{-6}</td>
<td>8.41 x10^{-6}</td>
<td>0.0002</td>
<td>-0.0001</td>
<td>-3.7 x10^{-5}</td>
</tr>
<tr>
<td>Pigs per KM² (Top Quartile)</td>
<td>0.0002</td>
<td>0.0006</td>
<td>0.0156</td>
<td>-0.0110</td>
<td>-0.0022</td>
</tr>
<tr>
<td>Poultry per KM²</td>
<td>1.96 x10^{-8}</td>
<td>7.54 x10^{-8}</td>
<td>1.99 x10^{-6}</td>
<td>-1.24 x10^{-6}</td>
<td>-3.29 x10^{-7}</td>
</tr>
<tr>
<td>Cumulative Afforestation</td>
<td>-2.06 x10^{-6}</td>
<td>-7.96 x10^{-6}</td>
<td>-0.0002</td>
<td>0.0001</td>
<td>3.47x10^{-5}</td>
</tr>
</tbody>
</table>

The MEs in ordered probit models indicate the increase (if the sign is positive) or decrease (if the sign is negative) in probability of an outcome in a dependent variable when an independent variable is increase by one unit. For example, if organic N per hectare increases by 1kg, then a probability of a river to get a Q3 on assessment increases by 0.00062.
<table>
<thead>
<tr>
<th>Variable</th>
<th>Q1</th>
<th>Q2</th>
<th>Q3</th>
<th>Q4</th>
<th>Q5</th>
</tr>
</thead>
<tbody>
<tr>
<td>Septic Tank Density</td>
<td>1.73 x10^{-7}</td>
<td>8.87 x10^{-7}</td>
<td>4.43 x10^{-5}</td>
<td>-3.6 x10^{-5}</td>
<td>-4.16 x10^{-6}</td>
</tr>
<tr>
<td>Cereal Share of Land Use</td>
<td>0.0007</td>
<td>0.0035</td>
<td>0.1771</td>
<td>-0.1438</td>
<td>-0.0166</td>
</tr>
<tr>
<td>Organic N Density</td>
<td>2.41 x10^{-6}</td>
<td>1.24 x10^{-5}</td>
<td>0.0006</td>
<td>-0.0005</td>
<td>-5.81 x10^{-4}</td>
</tr>
<tr>
<td>Pigs per KM² (Top Quartile)</td>
<td>1.74 x10^{-5}</td>
<td>8.83 x10^{-5}</td>
<td>0.0043</td>
<td>-0.0036</td>
<td>-0.0004</td>
</tr>
<tr>
<td>Poultry per KM²</td>
<td>2.07 x10^{-9}</td>
<td>1.06 x10^{-8}</td>
<td>5.31 x10^{-7}</td>
<td>-4.31 x10^{-7}</td>
<td>-4.98 x10^{-8}</td>
</tr>
<tr>
<td>Cumulative Afforestation</td>
<td>-7.43 x10^{-7}</td>
<td>-3.82 x10^{-6}</td>
<td>-0.0002</td>
<td>0.0002</td>
<td>1.79 x10^{-5}</td>
</tr>
<tr>
<td>Landfill within 3km</td>
<td>2.51 x10^{-5}</td>
<td>0.0001</td>
<td>0.0061</td>
<td>-0.0052</td>
<td>-0.0005</td>
</tr>
<tr>
<td>Rainfall</td>
<td>-2.14 x10^{-7}</td>
<td>-1.10 x10^{-6}</td>
<td>-5.5 x10^{-5}</td>
<td>4.46 x10^{-5}</td>
<td>5.16 x10^{-5}</td>
</tr>
<tr>
<td>Average Temperature</td>
<td>-7.8 x10^{-5}</td>
<td>-0.0004</td>
<td>-0.0199</td>
<td>0.0162</td>
<td>0.0019</td>
</tr>
<tr>
<td>Median Elevation</td>
<td>-1.94 x10^{-6}</td>
<td>-9.97 x10^{-6}</td>
<td>-0.0005</td>
<td>0.0004</td>
<td>4.67 x10^{-5}</td>
</tr>
<tr>
<td>x_coord</td>
<td>1.28 x10^{-9}</td>
<td>6.59 x10^{-9}</td>
<td>3.29 x10^{-7}</td>
<td>-2.67 x10^{-7}</td>
<td>-3.09 x10^{-8}</td>
</tr>
<tr>
<td>y_coord</td>
<td>-1.20 x10^{-9}</td>
<td>-6.15 x10^{-9}</td>
<td>-3.07 x10^{-7}</td>
<td>2.50 x10^{-7}</td>
<td>2.88 x10^{-8}</td>
</tr>
</tbody>
</table>

Both models’ MEs indicated that increasing septic tank densities and agricultural activities in the catchments increase a probability of a river to be assigned a Q-value 3 or lower and decreasing a probability of this river to reach “good status” or higher. Having a landfill within 3 km of a river undermines its chances to be assigned a high Q-value. At the same time increasing afforestation, high amount of precipitation and being located in a higher area increases a probability of a river to be of “good status” or higher. Tillage farming seems to have a profound negative effect on water quality in rivers, which may need to be addressed if the targets of achieving of water quality no lower than Q4 are to be achieved.

Despite the fact that including environmental variables does not affect model’s estimations, it does affect MEs. However, the effect is not very profound. Including environmental and landfill variables reduced the probabilities associated with septic tank density and cereal share in each category. The intuitive explanation that comes to mind is that landfill variables are associated with septic tank density and when omitted, the latter would be capturing the variation. However, on closer examination it was discovered that that these two variables are very weakly correlated and the correlation coefficient sign is negative. Thus, the differences in the MEs between two models are hard to explain intuitively.

The slight differences in the values of the MEs between the two models, however, do not changes the conclusions that arrive from both models and confirm the conclusion.
that including environmental variable may not add a lot of information of these particular variables are not a focus of the analysis, especially if obtaining the data for these environmental variables comes at a high cost.

It is hard to base policy recommendation on MEs of the ordered probit model of this kind. The probabilities are higher around Q3 cells but this could be an indication that there are a lot of rivers of high quality in Ireland and very few of Q1 and Q2, and most of the water quality points are in Q3 and Q4 categories, thus, ultimately the policy efforts will be focused on this categories but either seeking to increase river status from Q3 to Q4 or not allowing to deteriorate it from Q4 to Q3.

**III.5 Discussion and Conclusion**

In this chapter, an exploratory data analysis concerned with determining the effect of both agricultural and non-agricultural activities on the ecological quality of water resources is undertaken. To achieve this aim, a number of spatial datasets relating to agricultural and residential 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. The degree of forest cover was found to be positively associated with water quality. Finally, the amount of rain and average temperature are positively associated with water quality downstream, but statistically insignificant in the models reported in this chapter. While elevation is statistically significant and positively related to water quality, it is hard to find an intuitive explanation to such result as it is in opposition with existing literature that normally links higher slopes with higher chances of pollution runoffs. It could be a reflection of the fact that there are fewer human activities in the higher elevated areas in Ireland.

Comparing the two models reveals that adding environmental variables does not affect the value of the estimates for the septic tank density variable or the other variables associated with agricultural intensity. It also shows the effect of the environmental variables on water quality is hard to capture in the statistical modelling framework as the relationships are too complex. The model results were compared
with the finding of the two other Irish studies by Donohue et al. (2006) and Curtis and Morgenroth (2013), who utilised different statistical techniques but achieved similar results to those reported in this chapter. In this chapter the ordered probit model was used for estimations while Donohue et al. (2006) used Spearman rank correlations and Curtis and Morgenroth (2013) used linear and nonlinear OLS models in their estimations.

The findings of the model reported in this chapter have important policy implications. In relation to the agricultural sector, given the strong association between agricultural activity and water quality it has been previously widely reported that the agricultural sector will need to undergo significant structural change if WFD requirements are to be met (Volk et al., 2009). Some of these suggested changes include reductions in the use of fertilisers and a reduction in sheep and cattle stocking rates and afforestation (Haygarth et al., 1998, Merrington et al., 2003, Volk et al., 2009). This study further highlights the fact that agricultural activities may need to be further reduced to meet water quality targets. Alternatively, further afforestation may prove to be beneficial in mitigating water pollution. 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. Thus, further studies on the economic and environmental benefits of afforestation may be useful.

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, however, unclear at this stage if it is the presence of the septic tanks per se that is causing an environmental problem or incorrect management of those. The further study in this regard would help to determine an appropriate policy response to mitigate pollution from septic tanks. If septic tanks per se are the problem, then more environmentally friendly waste treatment infrastructure would be needed. On the other hand, improper management of the septic tanks commands better enforcement of the septic tank management rules.
The results of this study as informative as they are do not include or cover the plethora of the diffuse pollution sources that contribute to water pollution problem. However, these results highlight the importance of the multi-dimensional and spatial analysis for designing policy response. Otherwise, targeting one particular sector will not yield satisfactory results and will result in failure to achieve environmental goals and targets. In addition, more research is needed on the role of spatial factors in determining the likelihood of different human activities to contribute to pollution due to spatial and environmental variations at a particular point in time.

To sum up, the analysis presented in this chapter highlights the important relationship between land use and water quality. In particular, the level of forestry, septic tank 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 the 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 the agricultural sector is not the only factor responsible for adverse water quality and in turn the solution will require addressing the multitude of factors affecting water quality. In this regard, the analysis presented in this chapter will help to understand the sources of water pollution and will assist in planning aimed at restoring and maintaining water quality as required by the WFD.
Chapter IV.

**Using a Farm Micro-simulation Model to Evaluate the Impact of the Nitrogen Reduction Mitigation Strategies.**

This chapter describes the development, extensions and sensitivity analysis of the microsimulation model that allows one to estimate the costs of the nitrogen (N) reduction on farms in Ireland.

**IV.1 Introduction**

Along with climate change and biodiversity loss, nitrogen (N) pollution is one of the most serious environmental challenges facing the planet (Giles, 2005). It has been estimated that nitrogen pollution from farms, vehicles, industry and waste treatment is costing the EU up to £280bn (320bn Euros) per year (Harrabin, 2001). In addition to problems in relation to drinking water quality, excessive N has been found to have a detrimental impact on biodiversity within the aquatic system. Some of the harmful environmental effects associated with N include problems with soil and water acidification, contamination of surface and groundwater resources, increased ozone depletion and greenhouse gas levels, and loss of biodiversity (Elser et al., 1990, Fraser and Chilvers, 1981, Sullivan et al., 1991, Yiridoe and Weersink, 1998).

In Ireland as in other EU countries, nitrate enrichment of watercourses is an important environmental issue. The Environmental Protection Agency Ireland (EPAI) reports that in 2008 out of 180 nitrogen monitoring river sites, five sites had the highest values of N concentration; 7 percent of groundwater monitoring sites failed to comply with the Irish N concentration Threshold Value in the same year and 1 percent failed to comply with the Drinking Water maximum allowable N concentration, which could be linked to areas with more intensive agricultural practices (John, 2009). Additionally, the EPAI classifies 21 percent of river channels as being slightly polluted, 10 percent as being moderately polluted and 0.5 percent as being seriously polluted (John, 2009). The main impact of excess nutrients in water bodies is eutrophication, which causes an increase in biological and chemical oxygen demand and an unpleasant odour from the water, as well as a loss of habitats and changes to the river bed that in turn affect ship/boat navigation and negatively impact on recreational usage (Johnson et al., 2010, Novotny, 2003). Thus, there are
significant socio-economic effects associated with nutrient enrichment in addition to the environmental effects.

At the same time, N is an important nutrient for the reproduction and growth of all organisms, however, in excess, N becomes a pollutant (Doole, 2012). Yet, current levels of agricultural output could not be achieved and maintained without the widespread use of both synthetic and organic forms of N fertilizers (Merrington et al., 2003). Thus, alleviating pressure on water quality is likely to impose costs on the agricultural sector.

The complexity of environmental interactions poses a problem for researchers in identifying the sources of pollution and establishing robust causal relationship between different human activities and the volume of pollutants in streams. Lally et al. (2009) state that emissions of organic and inorganic nitrogen cannot be observed at a reasonable cost. O’Donoghue et al. (2014a) studied the statistical relationship between water quality at over 3,000 monitoring sites in Ireland and human activities in the upstream areas and found that there was a significant statistical relationship between agricultural activities (in addition to other activities) and lower water quality in the downstream areas. This is in line with the findings of the DEHLG (2010), which states that intensively farmed agricultural land may be a source of excess nutrients in Irish waters.

A large volume of literature discusses the diffuse pollution from agricultural land. These studies identify a number of pollutants from agriculture that may present a potential problem to the wider environment and to water resources in particular, and conclude that the main pressure to water quality comes from nutrient enrichment (Donohue et al., 2006, Donohue et al., 2005, Merrington et al., 2003, O’Donoghue et al., 2014a, Ritter and Shirmohammadi, 2000, Novotny, 2003, Lovelock et al., 2009, Doole, 2012, Doole et al., 2013, Schulte et al., 2006). The N pathways and its transformations in the environment are very complex and it can be difficult to establish a direct link between the potential sources and the affected areas. A number of studies have attempted to link human activities and impaired water quality. Donohue et al. (2006) and O’Donoghue et al. (2014a) identify a number of factors including intensive agricultural activity and human settlement, as exhibiting a high correlation with downstream water quality in Ireland. This is in line with
international research (see (Merrington et al., 2003, Novotny, 2003, Zhang et al., 2012)) and has been discussed in detail in the previous chapters.

Increasing public awareness of and demand for environmental amenities is changing attitudes about agriculture and the agricultural industry’s implicit property rights (Batie, 1988). To address these issues, a number of policy mechanisms have been introduced to improve water quality. Perhaps the most comprehensive legislative document to date is the Water Framework Directive (WFD) (Directive 2000/60/EC), which not only protects water resources from deterioration but also demands improvement in water quality to ‘good ecological status’ by 2015. At the EU level, the WFD requires that (a) all waters are restored to at least “good” quality and that (b) water currently classified as “pristine” quality is maintained. Other EU legislation that aims to restrict pollution of water bodies and to protect their habitats (Habitat Directive (Council Directive 92/43/EEC), Freshwater Fish Directive (Council Directive 78/659/EEC), Birds Directive (Council Directive 79/409/EEC)), to protect the uses of the streams (Drinking Water Directive (Council Directive 80/778/EEC), Bathing Water Directive (Council Directive 76/160/EEC), Sewage Sludge Directive (Council Directive 86/278/EEC), Urban Waste Water Treatment Directive (Council Directive 91/271/EEC)), to restrict nitrogen and other pollutants’ loss to overland/ground waters (Nitrates Directive (Council Directive 91/676/EEC)) amongst others. One response to these legislative changes is a growing use of public policy options for mitigating agricultural pollution problems. Specifically farmers are encouraged and often required to adopt a range of practical farm management solutions such as livestock, manure and land management to reduce N pollution.

The Nitrates Directive (Council Directive 91/676/EEC) was introduced in 1991 to control N losses from agriculture. In Ireland the Good Agricultural Practice regulations (European Communities (Good Agricultural Practice for Protection of Waters) Regulations, 2010) were introduced to implement the Nitrates Directive. These regulations place restrictions on the period during which the application of fertilizer is allowed; the amount of manure and inorganic fertilizer that is applied per hectare; the distance to a water body for fertilizer application; ploughing activities; and also impose requirements on the minimum storage capacities for livestock.
manure. These restrictions apply on a whole farm basis and penalties can be applied if a breach is detected under cross-compliance regulations.

Codes of practice for nutrient management have also been implemented in Ireland under various Agri-Environment Schemes. There have also been substantial financial incentives for farmers in Ireland under a grant aided Farm Waste Management scheme to improve the storage of manure and waste water on farms resulting in expenditure of over €1.2 billion on mitigation measures since the scheme's introduction in 2001 (DAFM, 2011). Whilst the expectation is that the Nitrates regulations will lead to significant progress towards meeting the WFD water quality objectives nationally, additional efforts may be required at a local level.

There is some evidence, that despite the efforts of Irish farmers to reduce N loss from their land, the production processes used and the prevailing weather conditions, still lead to the loss of nutrients to the wider environment (Donohue et al., 2006, John, 2009). Recent evidence has shown that there have been improvements over time, possibly as a result of Agri-Environmental measures and improved nutrient management on farms (O’Donoghue et al., 2014a). However, more effort is still required to mitigate N pollution in Ireland.

Given the wide range of potential mitigation measures, a number of studies such as those by Cuttle et al. (2007) and DEFRA (2004), using representative farm types, have estimated the relative costs of various measures aimed at reducing diffuse water pollution from agriculture in the UK using linear programming. More recent work by Fezzi et al. (2010), Fezzi et al. (2008) have analysed the effect of farm management measures aimed at reducing N losses across a wide variety of individual farm types. Their analysis revealed high variability in impacts across different farm types. Mitigation measures, therefore, need to be targeted at the individual farm level since average national level analysis will not reflect the heterogeneity of impacts across individual farms.

This chapter explores the costs associated with a range of potential N mitigation options in the Irish context and provides decision-makers with an economic analysis to aid in the decision-making process concerning pollution mitigation policies for the agricultural sector. The cost-effectiveness of different farm level N mitigation
strategies is investigated for each N unit reduction for each of the measures. This allows the lowest cost to be identified. A number of economic analyses of possible mitigation measures in this area have been conducted to date (Cuttle et al., 2007, Fezzi et al., 2010, Fezzi et al., 2008, Hennessy et al., 2005). However, the modelling methodology utilised in this chapter differs from those previously used in similar studies. Cuttle et al. (2007) and Hennessy et al. (2005) used a linear programming approach to produce estimates based on a representative (average) farm. Fezzi et al. (2008) used a farm accounting approach to estimate the effect of nitrate reduction strategies on dairy farms in the UK. They extended the previous analysis to include the farms in the nationally representative sample. The methodology used in this chapter, however, differs considerably from previous work in a number of ways: the analysis is conducted using Irish data and more significantly, a novel microsimulation approach is applied to obtain the estimates for predicting changes. In line with previous research, the economic impact of the possible N mitigation strategies is estimated. This chapter adds to previous research by extending the analysis to Irish farms and by developing the methodology to simulate changes at an individual farm level and, thus, creating a framework for farmers to make farm management decisions as well as assessing the economic policy impacts at micro-level. The strategies are ranked according to their cost-effectiveness as captured by the cost per unit of N abated (CPUA). This ranking allows all measure to be easily compared. The ranking of the CPUAs for individual farms are also examined in order to establish if this ranking is homogeneous for all farms in the sample. The model is validated by comparing the results with two mitigation measures that have previously been assessed by Hennessy et al. (2005) and Fezzi et al. (2008) 1) a stocking rate reduction to achieve a maximum organic N of 170 kg per hectare; 2) a 20 percent livestock units (LU) reduction. The model is then extended to include five more mitigation measures consisting of seven measures in total: 1) a reduction of fertiliser application by 20 percent; 2) a reduction of livestock units to achieve 170 kg of organic N per hectare; 3) a reduction of livestock units by 20 percent; 4) a change of feed mix to reduce cows’ dietary N intake; 5) the fencing off of watercourses to introduce a buffer zone; 6) the introduction of higher performing dairy breeds to improve the genetic merit of dairy cows and 7) more efficient slurry application.
Finally, this chapter adds to the literature on the topic by examining the sensitivity of the results to changes in prices and quantities. By considering the impact of price volatility on the estimated costs associated with various farm management strategies aimed at reducing N pollution this study informs policy-makers of the possible implications of varying market conditions on the outcomes at a farm level. This is likely to be a fundamental issue in assessing the costs of implementing directives such as the WFD, since price volatility may mean that in addition to a need for targeting of measures at individual farm types these same measures may need to vary through time in response to prevailing market conditions. The scenarios chosen for analysis are among the most popular suggestions for farm management practices aimed at reducing N losses from agriculture.

The rest of this chapter is structured as follows: section IV.2 gives the background of the study; in section IV.3 farm management measures to reduce N pollution from agriculture are discussed; in section IV.4 of this chapter the methodology used is described; data sources are introduced in section IV.5, the results are presented in section IV.6 and discussion and conclusions are given in section IV.7.

IV.2 Background

There is an extensive literature around mitigation strategies to reduce N losses from agriculture discussing a wide range of mitigation options (Byström, 1998, Novotny, 2003, Ritter and Shirmohammadi, 2000, Cuttle et al., 2007, Velthof et al., 2009). The primary factors that encourage N leaching from agriculture are over-fertilisation, excessive livestock numbers, improper use of manure, and exposure of bare soil during drainage periods (Bateman et al., 2007, Fezzi et al., 2008). Measures that aim to reduce N leaching should, thus, be targeted at these factors. The appropriate choice of mitigation strategies is connected to the N cycle in the agricultural environment. Combinations of these measures would aim to impact particular stages of the N cycle. Thus, understanding N movement and transformations in the environment and its interaction with agricultural systems in particular is important.

The N cycle is complex and represents a network of different physical and biochemical pathways, and the pathways to surface and ground water are not well defined. There are three main pathways through which different forms of N and its
compounds circulate in the agricultural environment: inputs (into the soil), transformations (within the soil), and losses (out of the soil) (Merrington et al., 2003). Figure IV.1 illustrates an example of the N cycle (IFA, 2007).

**Figure IV.1. Nitrogen Cycle.**

Inputs of N into the environment occur through atmospheric depositions (rainfall), inorganic fertilizer application, organic manure application; mineralisation of the soil organic N; crop residue, and biological N fixation by legumes (National Research Council, 1993). Nitrogen exists in soil N in organic and inorganic forms. Most of the soil N is stored in the soil organic matter making it unavailable for plant uptake. Through the processes of mineralisation and nitrification, organic N in soil and crop residue is transformed into ammonium (NH$_4^+$) and nitrate (NO$_3^-$), the forms that are available for plant uptake. There are a number of pathways through which N is lost from agricultural soil: plant uptake, volatilisation and de-nitrification, as well as losses through surface runoff and leaching to watercourses. The most beneficial
pathway is plant uptake. However, nitrate is water soluble and excesses move readily in soil moisture (Merrington et al., 2003, National Research Council, 1993, Novotny, 2003, Ritter and Shirmohammadi, 2000). All stages of the N cycle are dependent on a number of factors including geographical location of an agricultural enterprise, climate, underlying geology, soil permeability, soil microclimate and the type of agricultural enterprise (National Research Council, 1993).

Fezzi et al. (2008) and Bateman et al. (2007) identify a number of key factors that encourage N losses including over-fertilisation, excessive livestock numbers, improper use of manure and exposure of bare soil during cultivation. N loss mitigation measures which address either excessive inputs or unwanted losses of N from agricultural activities to the wider aquatic environment are discussed extensively in the literature. Table IV.1 summarises the measures that are commonly proposed. Different issues and solutions are associated with each strategy.

As has been mentioned previously there are three main pathways through which different forms of N and its compounds circulate in the agricultural environment: inputs, transformations, and losses (Merrington et al., 2003). Mitigation strategies aim at controlling these pathways through either restricting excessive N input use or reducing N losses to the environment that are already in the system. Due to weather and other environmental variations it is easier to target the reduction of N inputs to the ecological system as when less N is introduced into the environment, less N can potentially be lost through undesirable pathways (Merrington et al., 2003, Novotny, 2003, Ritter and Shirmohammadi, 2000). The input of N into agricultural system comes from chemical fertiliser application, animal manure and crop residue (IFA, 2007).

Some work has explored the costs of mitigation measures: Cuttle et al. (2007), Hennessy et al. (2005) and Lally et al. (2009) used linear programming in their estimations, while Fezzi et al. (2008) used a farm accounting approach to find the possible cost of the mitigation measures. This chapter contributes to this nascent literature by introducing a model that allows the simulation of impacts associated with policy responses such as a change in N production, and resultant changes in farm income, at the farm level. The focus here is on seven measures related to a reduction in the N on farms, which are anticipated to lead to an associated reduction
in N losses. These specific measures were selected for investigation as they are among the most common approaches recommended by environmental scientists and are also particularly suited to the structure of the Irish agricultural sector.

**Table IV.1. N Loss Mitigation Strategies**

<table>
<thead>
<tr>
<th>Strategies</th>
<th>Related Issues</th>
<th>Solutions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Restricting Excessive Inputs</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Inorganic Fertilizer Reduction</td>
<td>Excess fertilizer applied to grassland can be lost to water through runoff and leaching.</td>
<td>Reduction of fertilizer application would help to avoid runoff &amp; leaching of N from fertilizer excess.</td>
</tr>
<tr>
<td>Organic Fertilizer Reduction</td>
<td>Excessive and untimely application of manure/slurry causes N losses via volatilisation and/or runoff/leaching.</td>
<td>Reduction in organic fertilizer deposited and careful application reduces undesirable N losses.</td>
</tr>
<tr>
<td>Livestock Numbers Reduction</td>
<td>Livestock produce manure that is directly deposited to the land by animals during grazing or by land spreading of manures produced during the housing period.</td>
<td>Reduction in livestock units would reduce manure deposited and spread over land.</td>
</tr>
<tr>
<td>Change of Feed Mix</td>
<td>70-80 percent of the ingested N is excreted by farm animals. The higher content of N in feed mix means higher N content in excreta.</td>
<td>Reduction of N in the diet reduces N in animal excreta.</td>
</tr>
<tr>
<td>Calibration of Spreading</td>
<td>More accurate and N efficient slurry application methods can improve the N fertilizer replacement value and decrease the farm N surplus by offsetting inorganic fertilizer N inputs.</td>
<td>Low ammonia emission application of slurry by optimising application timing and/or method.</td>
</tr>
<tr>
<td>Equipment/ injection vs overland spreading</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Soil Testing</td>
<td>High risk of over-fertilising without testing the soil for the level of nutrients.</td>
<td>Early season soil testing reduces the risk of over-fertilisation.</td>
</tr>
<tr>
<td>Higher Performing Cattle breeds</td>
<td>The lower the yield of the dairy cow, the higher the N emissions per unit of output produced.</td>
<td>Utilisation of higher yielding cattle allows for reduction of the size of herd (excreta produced) without affecting output thus reducing N emissions per unit of output.</td>
</tr>
<tr>
<td>Reduction of N Losses</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Livestock exclusion</td>
<td>Manure deposition near/into streams causes water pollution. Allowing animals to access streams also causes sediment deposition and river bank destabilisation.</td>
<td>Prohibiting livestock access to streams prevents deposition of faecal material, turbidity and denudation of the stream banks.</td>
</tr>
<tr>
<td>Wetland Development/ Restoration</td>
<td>Overland runoff from agricultural land carries sediment and nutrients to streams.</td>
<td>Provides a filter for pollutants originating from agricultural land.</td>
</tr>
<tr>
<td>Riparian Buffer Zones/ Filter Strips</td>
<td>Overland runoff from agricultural land carries sediment and nutrients to streams.</td>
<td>Slows over-land runoff, allowing infiltration; allows nutrient uptake by vegetative cover.</td>
</tr>
<tr>
<td>Cover crops/ minimising periods when the soil is left bare</td>
<td>Leaving soil bare during the winter months and at cultivation increases risk of soil erosion and nutrient loss through runoff/leaching.</td>
<td>Cover crops provide protection against erosion, &quot;green&quot; manure source and additional revenue for farmers.</td>
</tr>
<tr>
<td>Timing of Fertilizer Application</td>
<td>Fertilizer application during/prior/straight after precipitation events or during autumn and winter leads to overland runoff or leaching of nutrients.</td>
<td>Timely fertilizer application prevents runoff/leaching, allows uptake of fertilizers by crops/grass.</td>
</tr>
</tbody>
</table>

However, identifying the pressure points and the policy alternatives does not provide sufficient information for efficient decision-making. The information on the cost of
mitigation alternatives is needed to achieve environmental objectives in the most efficient manner. In addition, agriculture is a heavily subsidised sector in the EU, it generates narrow profit margins for many farmers so careful analysis is needed to assess the impact of alternative policy measures on farm incomes before any of them are transited into legislation.

The costs associated with the various options for reducing N losses can vary considerably and hence the use of cost-effectiveness analysis (as expressed in cost per unit abated - CPUA) is becoming increasingly popular as a means to choose between them (Domínguez et al., 2009, Blok et al., 2001a, Moran et al., 2008). Where a number of mitigation options exist, CPUAs of a number of alternatives in terms of their cost effectiveness and their ranking provide policymakers and other decision-makers with guidance as to the least cost policy option from a range of alternatives.

One issue that is frequently overlooked in CPUA analysis is the sensitivity of the results to changes in prices. According to the latest OECD/FAO medium term outlook projections, the prices of crops and of most livestock products will be higher in both real and nominal terms during the decade to 2019 than they were in the decade preceding the 2007/08 price spikes. Also, in the period since 2006 the agricultural sector has witnessed significant price volatility (Fao et al., 2011). To date, most of the concerns about price volatility are related to the potential negative effects on consumer welfare especially in poorer households as a result of rising food prices. Instead, here the effect of price volatility on CPUA results is explored. In particular, the study conducted in this chapter highlights that the assessment and in turn the ranking of mitigation measures will depend on the commodity and input prices prevailing in the particular period.

The differences in CPUAs and their ranking come from the recently observed increase in price volatility. A number of factors contribute to this increased price volatility in agricultural markets. Firstly, the demand for food and feed crops for the production of biofuels increased dramatically. For instance, during the period 2007-2009 bio-fuels accounted for a significant share of global use of several crops – 20 percent for sugar cane, 9 percent for vegetable oil and coarse grains and 4 percent for sugar beet. It is generally expected that bio-fuel production will continue to exert
considerable upward pressure on prices in the future. Secondly, agricultural commodity prices are becoming increasingly connected to oil prices through its effect on the price of fuel and fertiliser. Climatic factors have also placed a considerable upward pressure on agricultural prices in recent times. For instance, weather related low yields in important food exporting nations such as Australia, Canada, Russia and the US have brought strong market reactions and soaring prices. Finally increased affluence in developing countries and population growth are likely to increase demand for, and hence prices of, agricultural foodstuffs in the future (Fao et al., 2011).

IV.3 Farm management measures to reduce N pollution from agriculture

As a result of legislative requirements to reduce N losses into the environment, the effect of a range of agricultural and agronomic practices generally referred to as mitigation measures, have been investigated (see Stark and Richards (2008) for a useful review). Some of the plethora of farm management practices that have been found to reduce N pollution include: land use change (Lord and Anthony, 2000, Saggar et al., 2007), reducing stocking rates (Di and Cameron, 2002, Oenema et al., 1998), changes in the timing and form of manure application (Chambers et al., 2000), use of cover crops (Hansen et al., 2007, Hooker et al., 2008) and manipulation of animal diets (van Groenigen et al., 2008, Velthof et al., 1998, de Klein and Monaghan, 2011). Perhaps the most straightforward approach to reduce N emissions is the adoption of less intensive grassland systems, through for instance, lowering stocking rates (Cuttle et al., 2004, Oenema et al., 1998). Although requiring a greater land area to achieve the same agricultural output, this would result in less nitrate leaching per unit of production than intensively managed grasslands. Changing agricultural and land use practices from high to low emission systems by, for example, converting from arable to extensive grassland farming (Lord and Anthony, 2000) or by converting from conventional to organic dairy farming has also been shown to substantially reduce N loss from agriculture.

Within livestock enterprises, N loss can be mitigated by changes in manure storage and manure application strategies (Chambers et al., 2000, Lalor et al., 2011). 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
Reducing the duration of grazing per day and/or season can also significantly reduce the quantity of N excreted and deposited directly to the land (Luo et al., 2008). Finally the use of cover crops has been shown to be very effective in terms of reducing N losses (Hansen et al., 2007, Hooker et al., 2008).

When the use of chemical fertilisers exceeds the level required by crops, the excess is lost through groundwater, surface streams and/or via overland runoff. Reducing the use of chemical fertiliser is the most straightforward way to reduce nutrient inputs which in turn can lead to a proportional reduction of N introduced and subsequently lost to the environment.

The surplus N in the livestock diet is excreted (DEFRA, 2007). Reduction of the N content of feed without the loss of the output or compromising on animal health is viewed as an attractive N mitigation option. Wright and Mutsvangwa (2003) report that switching from one feed to another can potentially reduce protein fed by 15-20 percent yearly and will reduce N excreted by the animals proportionally.

Livestock, particularly dairy cows, produce large volumes of manure as part of their digestive process. This is either directly deposited to the land during grazing or is spread over the land after being stored and is known to be a cause of pollution. A dairy cow produces 5.3 m$^3$ of slurry in 16 weeks of housing (European Communities (Good Agricultural Practice for Protection of Waters) Regulations, 2010), which contains approximately 19 kg of N. This manure/slurry has to be spread overland or exported from the farm. Livestock also deposits manure/urea directly on fields during grazing periods. In Ireland the Good Agricultural Practice regulations (European Communities (Good Agricultural Practice for Protection of Waters) Regulations, 2010) place a restriction on the amount of manure and inorganic fertilizer that may be applied per hectare - presently the amount is capped at 170 kg of organic N per hectare, with a possibility to derogate to 250 kg of N per hectare. It has been suggested in the literature that in order to achieve the objectives of WFD, it would be necessary to introduce restrictions such as the introduction of a 50 percent reduction in the application of fertilisers to crops and grass, halving of sheep stocking rates and a reduction in cattle stocking rates of 20-25 percent (Bateman et al., 2006, Haygarth et al., 2003).
Excluding animals from the areas next to streams (fencing) can substantially reduce the deposition of faecal material, turbidity from in-stream trampling and denudation of the stream banks (Novotny, 2003). If the areas adjacent to the streams are excluded from agricultural production completely, then the environmental gains are greater still due to decreased fertiliser usage.

Another strategy that can potentially result in the decrease of chemical N usage on farms is increasing the N efficiency from applied cattle slurry by improving the timing and method of application. Lalor et al. (2010) report that the method of application and the timing of application both affect the utilisation of N by grass due to variation in N losses through NH$_3$ volatilisation. By optimising both the application method and timing, the N fertilizer replacement value (NFRV) can be increased, resulting in a reduction in the chemical N fertilizer requirement on the farm.

In Ireland over 95 percent of the national dairy herd is comprised of the Holstein-Fresian breed. However, within the breed there is a wide variation in milk output, total solids, body condition score and fertility rates. The variation may be attributed to a better breeding index within the best performing herds. There are a number of factors affecting milk output per cow including: breeding index, parity, season of calving, management factors (such as the choice of feed, milking intervals, milking frequency) and geographic location (Diskin, 2012). Improving the breeding index within dairy herds can significantly improve the milk output per cow allowing a reduction in herd size and consequently a reduction in N on the farm. Economic breeding index (EBI) of a dairy cow was developed by Teagasc and represents a monetary figure that is calculated on the basis of a number of dairy animal traits (for full discussion, see ICBF (2014)). However, in this chapter when referring to breeding index, an unobserved index$^3$ within the dairy breed that allows the herd to perform better in terms of milk output per animal is meant.

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$^3$ The unobserved breeding index here is measure of the prevalence of a variation within the predominantly HF breed in Ireland that accounts for higher milk output per animal. It should not be confused with EBI which includes a number of other traits besides milk production.
IV.4 Methodology

**Estimation of N produced on the farm.**

The aim of the analysis undertaken in this chapter is to develop a framework in which one can readily assess the impact of N reduction measures on farm N budget and on farm income within the context of implementation of the WFD. In doing so it is first necessary to decide how to estimate the farm’s N budget. Often researchers focus on modelling the run-off and undesirable losses of N from farm land. This approach leads to difficulties for modellers as it requires the development of a separate hydro-geological model that would permit the prediction of N losses through different pathways to be made. This in turn requires a lot of hydro-geo-ecological data in very high resolution. Such data seldom exists nationally. As an alternative approach, the model discussed in this chapter allows one to approximate N losses using a ‘reduce inputs’ approach. The assumption behind this approach is that if less N is introduced into the environment during the production process on the farm, then less is subsequently lost through undesirable pathways – via volatilisation, run-off, and/or leaching. A proportional reduction is assumed throughout.

The total N on a farm depends on the number of livestock units\(^4\) (organic N) and on the amount of chemical fertiliser used as a part of the production process (chemical N). Haygarth et al. (1998) and Merrington et al. (2003) report that 70-80 percent of N ingested by the animals during grazing and/or feeding on concentrates is subsequently excreted in manure. The level of organic N for an enterprise \(j\) is calculated by multiplying the number of LU of type \(k\) in that enterprise of the farm \((NLU_{kj})\) by the annual N excretion rate of that LU type \((E_k)\) and summing across the \(K\) types of LU. This is then added to the inorganic N for the enterprise \(j\) and summed over the \(J\) enterprises to obtain the total N on the farm, as is given by:

\[
N_j = \sum_j \left( \sum_k (E_k NLU_{kj}) + \text{Inorganic } N_j \right)
\]

(IV.1)

---

\(^4\) In the Teagasc National Farm Survey (NFS) a dairy cow is taken as the basic grazing livestock unit. All other grazing stock is given equivalents as follows: Dairy cows 1.0; Suckling cows 0.9; Heifers-in-calf 0.7; Calves under 6 mths. 0.2; Calves 6-12 months 0.4; Cattle 1-2 years 0.7; Cattle over 2 years 1.0; Stock bulls 1.0; Ewes and rams 0.20 (lowland) 0.14 (hill); Lambs to weaning 0.00 (lowland), 0.00 (hill); Lambs after weaning 0.12 (lowland), 0.10 (hill); Hoggets and wethers 0.15 (lowland), 0.10 (hill). For more details see Connolly et al. (2008).
The ‘annual nutrient excretion rates ($E_i$) for livestock’ tables is used as published in Good Agricultural Practice for Protection of Waters (European Communities (Good Agricultural Practice for Protection of Waters) Regulations, 2010) to determine the N produced by each animal on the farm – 85 kg of N per dairy LU, 65 kg of N per beef LU and 7 kg of N per sheep LU\(^5\). The number of kilograms of chemical fertiliser purchased by the farmers was used to determine the amount of chemical fertiliser used on the farms.

**Estimation of the Farm Profit.**

Farm profit $\pi_i$ is calculated as the farm’s gross output ($Y_i$) less farm’s direct costs ($C_i$) and fixed costs ($FC_i$). The level of farm gross output and direct costs determine a farm’s profit (equation IV.2):

$$\pi_i = Y_i - C_i - FC_i$$

(IV.2)

Animal numbers and chemical fertiliser each affect the gross output volume ($Y$) and the direct costs ($C$) on the farms (equations IV.3 and IV.4). Farms in Ireland usually engage in more than one enterprise. Each enterprise is modelled separately here due to the fact that only dairy farms are considered. Modelling each enterprise separately allows estimations to be done at an enterprise level, capturing the relationships within enterprises specific to each system and subsequently to simulate impacts that are likely to affect specific enterprises. However, when the farm system is not important adopting a ‘whole farm’ approach may yield better estimates. $X_{ij}$ is a vector of explanatory variables, where $i$ denotes individual farm and $j$ denotes each farm enterprise (dairy, beef or sheep), and variables include the size of farm, the volume of fertiliser and concentrate used, number of livestock units, forage area, etc. (see Table IV.6 for a full list of variables used in the model for each function estimation). These variables determine the level of $Y_{ij}$ and $C_{ij}$ in equations IV.3 and IV.4. When more LU are present on the farm, more output is produced, however more organic N is also produced on the farm and costs incurred by a farmer to feed and maintain animals are also greater.

\(^5\) An N excretion rate of 7kg per sheep livestock unit is used in this chapter, despite the fact that for lowland sheep the N excretion rate is 13kg. However, this excretion rate covers both the ewe and its lambs and would thus result in an over-estimate of N/ha on the farms and hence would subsequently produce a lower cost per unit of N abated in the second scenario.
Thus, through manipulating (reducing) the number of animals and the amount of chemical fertiliser used, farmers could reduce the N budget on the farm and hence reduce environmental pressures. A positive relationship is assumed between animal numbers, the amount of fertiliser and the value of gross output and costs.

Econometric techniques are utilised for determining the gross output and direct cost functions. Three production and three cost functions are estimated in this model: dairy gross output, dairy direct costs, cattle gross output, cattle direct costs, sheep gross output, and sheep direct costs (equations IV.2 and IV.3).

**Approaches to production analysis**

There are number of approaches that can be utilised in production modelling. They include general equilibrium models, econometric analysis and engineered functions (Arfini, 2012). The purpose of econometric production analysis is to establish the effect that changes in inputs have on output and costs. Econometric analysis can take two directions: 1) micro-level production analysis based on individual units (e.g., farms); 2) macro-level analysis in which agricultural output is an input to production at the macro level (Bonaccorsi and Daraio, 2005). In this chapter only the former is discussed. The main approaches used in a micro-level production analysis are the production function approach (O'Donoghue and Lennon, 2014) and production frontier analysis in the form of stochastic production frontier (Bonaccorsi and Daraio, 2005, Mawa et al., 2014). In the first instance, the estimations seek to establish an average relationship between inputs and outputs/costs by specifying a functional relationship that explains the observed data (Bonaccorsi and Daraio, 2005). This exercise is carried out using either parametric (where an underlying data functional forms specified), nonparametric (without reference to a specific functional form), or semi-parametric approaches (when part of the model is parameterised and another part is not) (Bonaccorsi and Daraio, 2005).
Stochastic production analysis is normally used in productivity analysis when the frontier that envelopes the data points is estimated instead. In this data analysis all data points on the frontier are considered to be “efficient” and the distance between the rest of the data points and the frontier is measured through estimation of farm-specific efficiency scores and the factors explaining efficiency differentials (Kumbhakar and Lovell, 2003, Green and O'Donoghue, 2013, Coelli et al., 2005, Cooper et al., 2013, Mawa et al., 2014).

In this chapter the production function approach is used throughout since production efficiency is not the focus of this research. Separate gross output and direct cost functions are estimated, despite the fact that the duality approach - first introduced by Shepard in 1953 (Shepard, 1953) - allows one to estimate a cost function based on a profit or revenue since they are all a special case of a restricted profit function (Fuss and McFadden, 1978).

The duality theory states that any production function has an underlying cost function or correspondence. Assuming fixed prices, farmers are assumed to be seeking to determine the combination of inputs that allow profit maximisation. If a farm’s output is \( y = f(x) \), its revenue is \( h(x) \) and this farm is facing costs \( c = g(x) \), then profit is maximised where \( h'(x) - g'(x) = 0 \), where \( h'(x) \) is marginal revenue, \( g'(x) \) is marginal cost and \( x \) is units of inputs. Thus, the production, revenue or cost function need to be known to build a model (Debertin, 2012). The Duality Theory helps in this case as it states that only one of these functions needs to be known as a cost function is inversely related to the production function and (providing prices are fixed) can be derived from it. For a production function \( y = f(x) \), the cost is \( x = f^{-1}(y) \) – an inverse function (Debertin, 2012).

A plethora of literature has discussed the duality between production, profit and/or cost functions (Quiggin, 2002, Chambers et al., 2000, Chambers and Quiggin, 1998, Lusk et al., 2002, Debertin, 2012).

The dual approach proves to be a useful tool in many analyses. However, it is not pursued in this chapter for the following reasons. The dual approach is normally used in optimisation models with a view to determine the optimal combination of inputs and outputs for profit maximization/cost minimisation. The model developed here is...
not a maximization/minimization problem as the purpose is not to explore conditions under which farmers would optimise their production, but rather to simulate the changes in “what if” scenarios based on the observed data. The focus is on the predictive ability of the model rather than on the coefficients per se and models based on estimation of simple relationships have been found to perform well in this regard (e.g. Fezzi et al. (2010), O'Donoghue and Lennon (2014)). O'Donoghue and Lennon (2014) used this approach in their farm income generation model in which they estimated farm level production and cost functions and validated their results by simulating data and comparing simulated results with the actual data. While they used a “whole farm” approach, here the impacts are assessed at an enterprise level.

Moreover, Thompson and Langworthy (1989) showed that identical primal and dual results cannot be achieved due to risk, stochastic error and/or the functional form chosen. Lusk et al. (2002) investigated the conditions under which estimated production, profit and cost functions yield the same results and discovered that it is possible only under perfect conditions when the functional form exactly matches the data and there is no measurement error in the data. Their findings suggest that a measurement error as small as 0.01 percent can lead to considerable differences between dual and primal estimates (Lusk et al., 2002). Due to the fact that enterprise data on the farm is not perfect and often farmers allocate expenses like labour costs to different enterprises based on approximation rather than actual data, estimating cost from production estimates yield results which can be far off the primal estimates.

Another reason for not using the dual approach here is that when functions with a large number of inputs are used, deriving the inverse function may be a practically impossible exercise. For example, for a two input Cobb-Douglas production function:

\[ Y = A x_1^{\beta_1} x_2^{\beta_2} \]

the dual (and the least cost) cost function is (Debertin, 2012):

\[ C = Y^{1/(\alpha_1+\alpha_2)} A^{1-1/(\alpha_1+\alpha_2)} (B_1^{-1} B_{21/2+1} v_1)^{1/(\alpha_1+\alpha_2)} (B_2^{-1} B_{12+2} v_2)^{1/(\alpha_1+\alpha_2)} \]

where \( A, \beta_1, \beta_2 \) are parameters, \( x_1 \) and \( x_2 \) are inputs, and \( v_1 \) and \( v_2 \) are the prices of \( x_1 \) and \( x_2 \) respectively. Furthermore, in this research, in the estimated functions, both value and volume variables are utilised to accommodate the simulation procedure and
since deriving a dual cost function is based on relative price elasticity, a dual approach is not suitable in this framework.

**Functional Form**

Choosing a functional form for parametric econometric analysis is another important task. There are a variety of functional forms one could use when estimating output or costs and since the true functional form cannot be known, the problem is to choose the form that best suits the task at hand (Griffin et al., 1987).

The most commonly used functional forms in production analysis are the Cobb-Douglas and trans-log (which is a generalised form of the Cobb-Douglas function), although other functional forms are sometimes used (Lusk et al., 2002, Flichman, 2011, Mawa et al., 2014). Lusk et al. (2002) state that from a theoretical point of view there is no advantage in using one functional form over another for estimating the true but unknown function underlying data.

Both trans-log and Cobb-Douglas functional forms have been used extensively in production analysis (a few examples include Švarc and Švarc (1989), O Neill and Matthews (2001), Liu (2002), Mawa et al. (2014)). Liu (2002) conducted a comparative analysis of the output studies for libraries and out of nine studies examined seven used Cobb-Douglas functional forms and two used trans-log. There are advantages and disadvantages in using both of these functional forms, they are both linear in parameters and, thus, can be estimated using the least squares method (Griffin et al., 1987, Mawa et al., 2014).

Trans-log is a flexible functional form, however it is hard to estimate, and difficult to interpret. It requires the estimation of a large number of parameters $(k+3+k(k+1)/2)$, where $k$ is the number of inputs, which with a small sample data can lead to a big loss of degrees of freedom$^6$ (Mawa et al., 2014). Cobb-Douglas, on the other hand, is an easy to estimate and interpret function, it requires estimation of much fewer parameters $(k+3)$ than the trans-log function and, thus, is very useful in production analysis.

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Romer (2011) states that the Cobb-Douglas function is “a good approximation to actual production functions”. In this chapter, production and cost functions are estimated using ordinary least squares (OLS) regressions. A similar parametric approach was used by Fezzi et al. (2010) who used a linear regression approach (assuming constant returns to scale) to estimate the change in farm gross margin that arises from different policy measures and by O'Donoghue and Lennon (2014) who used Cobb-Douglas functional forms for regression analysis in their Farm Income Generation model. In contrast to Fezzi et al. (2010), who estimated a single function – a change in farm gross margin resulting from policy change, here separate equations for gross output and direct costs are estimated and then a gross margin is calculated. O'Donoghue and Lennon (2014) have also estimated their output and cost functions separately.

Estimating functions separately allows the impact of shocks on these components to be explored and the simulation of changes in these components at a farm system level and, thus, maybe more useful for modelling purposes. Another reason for using the same functional forms and variables as O'Donoghue and Lennon (2014) is that their model is based on Irish farm data. They have also tested, calibrated and validated their model, thus, making it a ready to use tool for farm data analysis.

**DEVELOPING A MICRO-SIMULATION MODEL**

The model (as described in equations IV.1- IV.4) allows the simulation of changes in farm profits due to gross output or direct costs changes at an enterprise level. The impact of different measures to reduce N can differ in both the economic and in the environmental dimensions across farms, thus, the analysis should be carried out at a farm level. Microsimulation techniques allow a modeller to conduct analyses at this scale. Microsimulation has been used for many years and is an effective tool for evaluating the socio-economic impacts of different mitigation options where it is difficult or impossible to conduct a real life experiment (Merz, 1993, O'Donoghue et al., 2012). It has been widely used for income generation modelling, tax system evaluation and pension schemes evaluation inter alia (Mitton et al., 2000, O'Donoghue et al., 2012). Microsimulation can be carried out using various techniques, for example, linear programming (Hennessy et al., 2005), partial budgeting (Fezzi et al., 2008) or econometric regression analysis (Fezzi et al., 2010, 2012).
One of the examples of a successful microsimulation model is MAMBO – a microsimulation model developed by Kruseman et al. (2008a), Kruseman et al. (2008b) in the Netherlands as a framework to monitor pollution as well as to assess mitigation policy scenarios. They concluded that the micro-simulation approach was one of the most favourable in assessing the impact of non-point pollution mitigation policies. Ramilan et al. (2011) simulated alternative dairy farm pollution policies using meta-modelling techniques. Calibrating and linking two models for farm profit and nitrogen discharge calculations, they developed an integrated model to simulate an impact of the three policy interventions: a tax on nitrate emissions, a standard that limits the allowable level of emissions and a tax on emissions above the allowable standard (Ramilan et al., 2011). More recent work by Doole (2012) and Doole et al. (2013) developed a bio-economic model with 216 different cow herds to simulate effect of agri-environmental policies for reducing nitrate pollution from New Zealand dairy farms on farm incomes and nitrogen leaching loads.

**Figure IV.2. Simulation Model Flow Diagram**

![Simulation Model Flow Diagram](image-url)

Model Input: Farm Level Data

Step 1. Define Scenarios

Step 2. Use estimated Production and Cost Functions for dairy, beef and sheep enterprises. Save betas and error terms for microsimulation

Step 3. Change independent variables (X) according to scenarios.

Step 4. Predict new output and costs using changed Xs and betas & error terms from previous estimation.

Model Output: Simulated change in the farm GM
All of these techniques allow one to model changes at the farm level. The choice of a particular technique depends on the objective of the model. Hennessy et al. (2005) utilise the FAPRI-Ireland Farm Level Model, which is a dynamic gross margin maximising model and was first described by Breen and Hennessy (2003). The linear programming approach allows model optimization, however, in the model developed in this chapter the aim is not to optimize farm production but rather to understand how the farm system affects the costs of the mitigation measures. In the Fezzi et al. (2008) farm budget model the underlying assumption in the 20 percent LU reduction scenario is that the output and costs would be reduced by 20 percent as well. However, this assumption may not hold in reality as the relationships and dependencies between variables are more complex. In this chapter the Fezzi et al. (2010) approach is followed in adopting a regression framework. Regression analysis was chosen for this model as the most appropriate technique for estimations as it allows one to capture the marginal effect of changes in the variables of interest, e.g. the change in the number of livestock units. The schematic of the overall simulation procedure is depicted in Figure IV.2.

In order to validate the model as a case study two mitigation options that have previously been estimated by Hennessy et al. (2005) and Fezzi et al. (2010) are explored: 1) a stocking rate reduction to achieve maximum organic N of 170 kg per hectare; 2) a 20 percent LU reduction. These measures would lead to changes in the farm inputs and/or outputs through reductions in the dry stock, fertiliser usage, feed change etc. The model input is the farm level data which is described in the next section of this chapter.

The impact of the alternative mitigation measures on individual farm profit ($\pi$) is simulated using estimates of gross output and direct cost functions based on farm-level data (equations IV.2 and IV.3). The fixed costs are not affected by the scenarios in the simulations, thus, the changes in the farm profit are due to changes in farm gross margin (GM) (equation IV.5).

$$GM_i = Y_i - C_i$$  \hspace{1cm} (IV.5)

$$Y_{ij} = (X_{ij} \mid \beta_j, \epsilon_{ij})$$  \hspace{1cm} (IV.6)
The simulations are carried out by holding the regression coefficients ($\beta_j$, $\gamma_j$) and the error terms ($\varepsilon_{ij}^y$, $\varepsilon_{ij}^c$) constant and changing the explanatory variables ($X_{ij}$) according to the scenarios (in the case study scenarios here it is the number of LU that is altered). When the parameters of the model are estimated the new levels of gross output and direct cost are predicted for each enterprise (denoted as $C_{ij}^\circ$, $Y_{ij}^\circ$ in equations IV.6 and IV.7). The results are then aggregated to the farm level (equation IV.8). The impact of the simulated changes in the animal numbers and/or fertiliser is the difference between farm profit before ($\pi_i$) and after the change ($\pi_i^\circ$) (equation IV.9).

$$C_{ij}^\circ = (X_{ij}^\circ | \gamma_j, \varepsilon_{ij}^y) \quad \text{(IV.7)}$$

$$\pi_{ij}^\circ = \sum Y_{ij}^\circ - \sum C_{ij}^\circ \quad \text{(IV.8)}$$

$$\Delta \pi_i = \pi_{i}^\circ - \pi_i \quad \text{(IV.9)}$$

The simulations are carried out by holding the regression coefficients ($\beta_j$, $\gamma_j$) and the error terms ($\varepsilon_{ij}^y$, $\varepsilon_{ij}^c$) constant and changing the explanatory variables ($X_{ij}$) according to the scenarios (in the case study scenarios here it is the number of LU that is altered). When the parameters of the model are estimated the new levels of gross output and direct cost are predicted for each enterprise (denoted as $C_{ij}^\circ$, $Y_{ij}^\circ$ in equations IV.6 and IV.7). The results are then aggregated to the farm level (equation IV.8). The impact of the simulated changes in the animal numbers and/or fertiliser is the difference between farm profit before ($\pi_i$) and after the change ($\pi_i^\circ$) (equation IV.9).

$$N_i^\circ = \sum_j \left( \sum_k (E_k^{NLU_{ij}}) + \text{Inorganic } N_j \right) \quad \text{(IV.10)}$$

The changes in N come from the change in animal numbers according to the particular scenario. The adjusted number of livestock units is $NLU_{ij}^\circ$ (equation IV.10), where $k$ is a type of a LU – dairy, beef or sheep.

$$NLU_{ij}^\circ = (0.8 \times NLU_{ij}) \quad \text{(IV.11)}$$

In the second scenario the number of LU on each farm is reduced by 20 percent for each enterprise (equation IV.11) and the new $N^\circ$ on the farm is calculated as in equation IV.10.

The final change in N on the farm is the difference between the N level before the simulations and the level, $N^\circ$, which is simulated for the farm after the mitigation measure introduction (equation IV.12).

$$\Delta N_i = N_{i}^\circ - N_i \quad \text{(IV.12)}$$

$CPUA = \Delta \pi_i / \Delta N_i \quad \text{(IV.13)}$
Thus, this methodology allows one both to simulate the changes in farm profit and to simulate the change in N on the farm as a result of the mitigation measures. It can potentially be used by decision-makers in determining not only the level of abatement that can be achieved through different measures and the cost associated with them but also to compare the cost efficiency (expressed as CPUA) for each individual farm (equation IV.13).

**MODEL EXTENSIONS**

After the validation, the model is extended to include five more scenarios, thus, seven scenarios were simulated in the model altogether: 1) the reduction of inorganic fertiliser by 20 percent; 2) the reduction of Livestock Units to achieve 170 kg of N per hectare; 3) the reduction of LU by 20 percent; 4) a change in the feed mix; 5) the fencing off of adjacent streams (and subsequent de-intensification of production); 6) a change in the method and the time of slurry application; 7) an improvement in the breeding index of the dairy herd. In each of the simulated scenarios the level of N is adjusted in line with the assumptions of the scenario. The new level of $N^\hat{\alpha}$ is calculated for each scenario as shown in Table IV.2.

The CPUA for a scenario is then calculated as explained in equation IV.13 and the results for each measure on average and for each individual farm are compared. The estimation of farm-level CPUA requires knowledge of the effects of the proposed mitigation strategies on the gross output and direct cost functions of each farm as well as the amount of nutrient loss mitigated as a result of each strategy. The ranking of the mitigation strategies from the cheapest to the most expensive per unit of N potentially mitigated allows one to compare the costs of the measures.
Table IV.2. Implication of each policy scenarios for Nitrogen levels.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Level of Nitrogen in Scenario</th>
</tr>
</thead>
<tbody>
<tr>
<td>1) the reduction of inorganic fertiliser by 20 percent;</td>
<td>$N_{j}^{+} = \sum_{j} \left( \sum_{k} \left( E_{k} NLU_{kj}^{+} \right) + \text{Inorganic } N_{j}^{+} \right)$</td>
</tr>
<tr>
<td>2) the reduction of LU to achieve 170 kg of organic N per hectare;</td>
<td>Farms are assumed to reduce LU starting with the enterprise that has the lowest gross margin per LU to reach 170 kg of N per hectare. The adjusted number of livestock units is: $N_{j}^{+} = \sum_{j} \left( \sum_{k} \left( E_{k} NLU_{kj}^{+} \right) + \text{Inorganic } N_{j}^{+} \right)$</td>
</tr>
<tr>
<td>3) the reduction of LU by 20 percent;</td>
<td>$N_{j}^{+} = \sum_{j} \left( \sum_{k} \left( E_{k} NLU_{kj}^{+} \right) + \text{Inorganic } N_{j}^{+} \right)$</td>
</tr>
<tr>
<td>4) a change in the feed mix;</td>
<td>A reduction by 15 percent in the amount of N excreted by dairy cows is assumed; the excretion rate for other LU is unchanged. The adjusted excretion rate is $E_{j}^{<em>}$ $N_{j}^{</em>} = \sum_{j} \left( \sum_{k} \left( E_{k}^{<em>} NLU_{kj}^{</em>} \right) + \text{Inorganic } N_{j}^{+} \right)$</td>
</tr>
<tr>
<td>5) the fencing off of adjacent streams (and subsequent de-intensification of production)</td>
<td>$N_{j}^{<em>} = \left( \sum_{j} \left( \sum_{k} \left( E_{k} NLU_{kj}^{</em>} \right) + \text{Inorganic } N_{j}^{+} \right) \right) \times \frac{\text{NewFarmSize}}{\text{OriginalFarmSize}}$</td>
</tr>
<tr>
<td>6) a change in the method of slurry application from splash-plate to trailing shoe;</td>
<td>$N_{j}^{<em>} = \sum_{j} \left( \sum_{k} \left( E_{k} NLU_{kj}^{</em>} \right) + \text{Inorganic } N_{j}^{+} \right)$</td>
</tr>
<tr>
<td>7) an improvement in the breeding of the dairy herd.</td>
<td>Farms are assumed to reduce the number of dairy cows to maintain GO at its pre-existing level. Other livestock units remain unchanged. The adjusted number of livestock units is: $N_{j}^{<em>} = \sum_{j} \left( \sum_{k} \left( E_{k} NLU_{kj}^{</em>} \right) + \text{Inorganic } N_{j}^{+} \right)$</td>
</tr>
</tbody>
</table>

Assumptions behind the Mitigation Strategies

In order to formalise the model, certain assumptions had to be made within each measure. In the fertilizer reduction scenario the volume of chemical fertilizer used on dairy farm is reduced by 20 percent relative to the current level used on the farms. In Scenario 2, the farmers are assumed to drop the livestock with the lowest gross margin per LU. In Scenario 3, it is assumed that each type of livestock is reduced by 20 percent.

In Scenario 4, it is assumed that farmers make alterations to their feed mix to ensure a reduction by 15 percent in the amount of N excreted by dairy cows in line with the findings of Wright and Mutsvangwa (2003). Wright and Mutsvangwa (2003) suggest possible lowering of feed costs as a result of using more efficient N feed, in this
analysis it is assumed - in line with DEFRA (2007), Fezzi et al. (2008) - that there is an extra cost associated with this strategy. In the model described in this chapter a 25 percent increase in the cost of the new feed mix is assumed. This cost is associated with the higher price of the new feed and of protein monitoring.

In Scenario 5, it is assumed that adjacent streams are fenced off. The costs associated with this measure include the cost of fence construction and maintenance as well as a loss of grazing/production land. The cost of the fencing includes the actual cost of erecting the fence of €0.90 per metre and the cost associated with the reduction in productive land area. Ten meter zones are assumed to be fenced off. Two possible sub-scenarios are estimated: 1) a reduction in production intensity, where a farmer would choose to reduce the number of LU pro rata based on the reduction in farm size; and 2) the possible intensification of production where farmers would keep the existing livestock numbers despite the reduction in overall farm size. The land taken out of production is assumed to be pasture or forage land on the dairy farm. This scenario will lead to a cessation of fertilizer use on the land taken out of production. In the case of intensification (increase in a farm’s stocking rate) of production there will be an increase in costs associated with extra chemical fertilizer spread on pasture in order to ensure sufficient grass production. If the latter happens the change in N could potentially be in the opposite direction to environmental objectives as there will be an increase in both organic and chemical N per hectare. Increases in organic N per unit area will result from an increase in stocking rates due to a decrease in land available for spreading slurry. Increases in chemical N may result if more chemical fertilizer is used.

In Scenario 6, the effect of changes in the application of manure from the splash plate (SP) (spraying manure over the field) method to the trailing shoe (TS) method (depositing manure directly on the top layer of the soil) are examined. The TS method allows more precise manure/slurry application reducing losses of N through volatilization in the form of greenhouse gases so that less chemical fertiliser is required. The SP method of application allows 12 percent of N in manure to be available for plant uptake while the TS method allows 22 percent and, thus, decreases the amount of chemical fertiliser required to support sufficient grass cover (Lalor et al., 2011). Two slurry application sub-scenarios are estimated: 1) switching to slurry
application in spring vs summer without changing the method of application; 2) changing the method of application without changing the time of application. Farms are assumed to initially all use the splash plate and then shift to the trailing shoe method in scenario 6. The actual level of N reduction will differ depending on the stocking rate of the individual farms. Switching from summer to spring application has no extra cost. There is an extra cost of €0.77 per m$^3$ of slurry (Lalor et al., 2011) associated with using TS machinery instead of SP. This amounts to an extra cost of €4.08 per year per cow for TS application. The N fertilizer savings per cow are highly dependent on the stocking rate of the farm, and will vary depending on the fertilizer N advice for the farm, which is mainly influenced by stocking rate. Thus, the chemical N fertilizer requirement and cost associated with each strategy is calculated for each farm according to stocking rate. The change in costs is calculated as the difference between the chemical fertilizer reduction and extra cost associated with hiring TS machinery instead of SP machinery. It is reported that 34 percent of slurry is spread by farmers during spring and about 50 percent during the summer and the remainder (16 percent) is spread in autumn using mainly the SP method of application (Hyde et al., 2006). The availability of slurry N in summer and autumn is similar; hence it is assumed here that 66 percent of slurry is spread during the summer. Each tonne of slurry contains approximately 3.6 kg of N. However, the N availability from slurry differs depending on the timing of application and method of application (Table IV.3). The TS method of application increases N availability in slurry by 10 percentage points both in summer and in spring (Lalor et al., 2011).

### Table IV.3. N availability in slurry with respect to time & method of application

<table>
<thead>
<tr>
<th>Method</th>
<th>Splash-plate</th>
<th>Trailing Shoe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Timing</td>
<td>Summer</td>
<td>Spring</td>
</tr>
<tr>
<td>Total N content (kg/m$^3$)</td>
<td>3.6</td>
<td>3.6</td>
</tr>
<tr>
<td>NFRV (percent)</td>
<td>12</td>
<td>21</td>
</tr>
<tr>
<td>Available N in slurry (kg/m$^3$)</td>
<td>0.43</td>
<td>0.76</td>
</tr>
<tr>
<td>N chemical fertilizer advice per cow at stocking rate of 2 LU/ha (kg ha$^{-1}$)</td>
<td>100.5</td>
<td></td>
</tr>
<tr>
<td>Slurry production per cow (m$^3$ in a 16 week winter period)</td>
<td>5.3</td>
<td></td>
</tr>
</tbody>
</table>

A dairy cow produces 5.3 m$^3$ of slurry in 16 weeks of housing (European Communities, 2010), which contains approximately 19 kg of N. In a summer application only 12 per cent of this is available for crop/grass uptake, and in a spring
application 21 percent is available. The demand for chemical N fertilizer depends on the farm stocking rate, e.g. if a stocking rate on a farm is 2 LU per hectare, in the absence of slurry application, 100.5 kg/cow of chemical N fertilizer would be required as advised by Coulter and Lalor (2008). If 66 percent of slurry is spread during the summer and 34 percent in spring (at the aforementioned stocking rate) 97.63 kg per cow of chemical N fertilizer is required instead of 100.5 kg.

In Scenario 7 the effects of increasing the breeding index is examined. This can potentially allow achieving higher levels of performance from each animal in a herd, which in turn reduces the amount of N lost per unit of output (Walsh et al., 2008). It is assumed that farms maintain gross output at its current level and reduce their LU accordingly after moving to more productive breeds and by so doing, reduce the amount of N lost per unit of output. Data on breeding indices are currently not available within the NFS dataset. The effect of increasing milk yields by improving the breeding index is estimated by regressing the average yield per cow in the dairy herd, on the amount of concentrates, number of days grazed, early/late calving and predicting the error term. The error term captures the variation in milk output unexplained by the observables and is attributed to variation in the breeding index. The minimum breeding index (as captured by the error term) in the top yield per cow quintile is then taken as the target to be achieved. This error term is substituted instead of the error terms in the lower yield quintile regressions and the new milk yield per cow is predicted. Thus, it is assumed that farms in the lower quintiles are adjusting their breeding index to match the productivity of farms in the upper quintile. This increased yield allows for a reduction of the herd size without the loss of production. The size of the new reduced dairy herd is calculated and the effect of the reduction on direct costs, gross margin and N reduction is estimated through microsimulation as outlined in equations IV.1 to IV.8. There are extra costs associated with this strategy including additional feed for higher yield cows. This is assumed to be offset by the reduction in the herd size. The cost of increasing the breeding index (artificial insemination straws and labour), is generated by assuming 4.5 artificial insemination straws per cow at a total cost of €20 per straw (Diskin, 2012).
Two sensitivity analyses are conducted to assess whether prevailing market conditions, and farmers output supply/input demand responses influence the relative ranking of the CPUA of the mitigation measures. In the first sensitivity analysis all quantities (i.e. input and output) are held constant at their 2008 levels and the values of gross output and direct costs are updated by applying historic input and output price indices for the period 1997 to 2009, in the process generating a set of new baseline datasets (one for each year). The seven scenarios discussed earlier are then applied to each baseline dataset and new CPUA for each measure is calculated. This allows one to assess the sensitivity of CPUA ranking to short run changes in prices (i.e. before farms have an opportunity to adjust their decision making). Since input and output quantities are held constant, the change in N produced is not affected by price changes.

The second, more realistic, sensitivity analysis proceeds by using the actual input and output quantities and prices for each year from 1997 to 2009, capturing both the effects of price changes and the farmers’ responses. Once again the CPUA is calculated for the seven scenarios and the consistency of their ranking is explored. In this analysis the changes in quantities will also lead to changes in the level of N produced, in addition to changes due to the seven scenarios.

IV.5 Data

In order to simulate the changes at a farm-level, socio-economic data at the farm level is required. Teagasc - The Irish Agriculture and Food Development Authority- has conducted the National Farm Survey (NFS) on an annual basis since 1972 (Connolly et al., 2010). The resultant dataset contains information for a sample of approximately 1,200 farms per annum that are nationally representative of over 100,000 farms in Ireland. This sample, however, excludes pigs and poultry farms due to an inability to obtain a representative sample for these types of farms. It is also not representative of very small farms. The NFS dataset contains socio-economic information which allows analysis of the physical and economic performance of the different farming sectors in the Republic of Ireland to be conducted.
In this chapter NFS data for the year 2008 is used for model validation and model extension simulations and NFS 1997 to 2009 is used for sensitivity analysis. Farms in the NFS are assigned to one of six possible systems: specialist dairy; dairying other; cattle rearing; cattle other; mainly sheep; mainly tillage (Hennessy et al., 2010). The category assignment is based on the dominant enterprise, which is established based on the Standard Gross Margins (SGMs) under the EU FADN typology set out in the Commission Decision 78/463 (Hynes et al., 2008b, Kinsella and Connolly, 2004). Under this methodology SGM is assigned to each type of farm animal and each hectare of crops. Farms are then classified into groups called particular types and principal types, on the basis of the proportion of the total SGM of the farm which comes from the main enterprises (after which systems are named). This methodology was adapted to suit Irish conditions more closely (the reader is referred to Connolly et al. (2009) for further details). Farms in Ireland typically engage in more than one enterprise.

The number of farms in the NFS sample varies from year to year from 1,279 farms in 1994 to 1,054 in 2009, which reflects the decreasing number of farms in Ireland, however the farms are getting bigger in size and more specialised. National weights are applied to represent the population of farms in Ireland. National weights are produced by Teagasc on the basis of the Census of Agriculture tables produced by the Irish Central Statistics Office (CSO). All summary statistics and model results reported in this chapter are produced on the basis of weighted NFS data.

For the purpose of this research the focus is on farms that are identified in the NFS as ‘specialist dairy’ and ‘dairy and other activities’ (from now on referred to as dairy). There are two primary reasons for focusing on dairy farms in this research: 1) the relatively good economic performance of dairy farms in Ireland and 2) environmental pressures generally associated with intensive dairy farm systems.

In terms of economic significance, dairy farms in Ireland have gross margins that are high relative to other farm systems and dairy farms’ gross margins are growing at a faster rate. Gross Margin (GM) is a good indicator of farm performance because it represents the difference between Gross Output (GO) and Direct Costs (DC). Furthermore, movements in GO (Figure IV.3) and DC (Figure IV.4) provide useful information about the source of changes in GM (Figure IV.5).
Figure IV.3. Dynamics of Gross Output on farms in Ireland (1996-2008).

Figure IV.4. Dynamics of Direct Costs on farms in Ireland (1996-2008).
Figure IV.5. Dynamics of Gross Margin on farms in Ireland (1996-2008).

It can be seen in Figure IV.5 that dairy and dairy other farms have significantly higher GM than cattle, cattle rearing and mainly sheep systems. It is also evident that dairy GM is growing at a rate higher than in other systems during the period. This is due to the high growth rate of dairy output (Figure IV.3) despite the fact that for the dairy farms the value of direct costs was growing at the same time (Figure IV.4). The rapid growth in dairy farms’ GO was caused by both increased milk yield per animal and due to consolidation in the industry with fewer farms producing more milk.

Dairy and dairy other farms are not only leaders in terms of economic performance; they also have higher organic N production and chemical N use per hectare (Table IV.4) relative to other systems. The national average (which is lower than the non-derogation requirement of 170 kg N per hectare under European Communities (Good Agricultural Practice for Protection of Waters) Regulations (2010)) disguises the range of organic N application across Irish farms with 27 percent of the farms in Ireland producing more than 170 kg of organic N per hectare. At the moment farmers that are over the requirement of 170 kg of organic N per hectare can apply for derogation, but the regulation may become more stringent in the future.
Table IV.4. Mean N per hectare and Proportion of farms in N categories, 2008

<table>
<thead>
<tr>
<th>Farm System</th>
<th>Org. N (kg per hectare)</th>
<th>Chemical. N (kg per hectare)</th>
<th>Source: weighted NFS data, 2008</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>&lt;170</td>
<td>&gt;170</td>
<td>Mean</td>
</tr>
<tr>
<td>Dairy</td>
<td>0.79</td>
<td>0.21</td>
<td>142</td>
</tr>
<tr>
<td>Dairy other</td>
<td>0.96</td>
<td>0.04</td>
<td>82</td>
</tr>
<tr>
<td>Cattle</td>
<td>0.99</td>
<td>0.01</td>
<td>72</td>
</tr>
<tr>
<td>Cattle rearing</td>
<td>0.99</td>
<td>0.01</td>
<td>79</td>
</tr>
<tr>
<td>Sheep</td>
<td>1.00</td>
<td>0</td>
<td>36</td>
</tr>
<tr>
<td>Tillage</td>
<td>1.00</td>
<td>0</td>
<td>22</td>
</tr>
</tbody>
</table>

The dairy system turns out almost twice as much organic and chemical N per hectare as any other system. Dairy other farms, despite reducing N emissions over the previous few years, still report higher amounts than in other systems. Twenty one percent of the dairy farms and four percent of dairy other farms in Ireland in 2008 exceeded the limit of 170 kg N per hectare (Table IV.4). Additionally, 3 percent of dairy farms were found to have exceeded chemical N limit per hectare.

GIS Data

In order to estimate the effect of the stream fencing scenario, information on the number of farms that have streams within 500 meters of the farmhouse was collected. GIS was used to identify these data. From satellite imaging, using GIS the dairy farms that have streams within 500m from the centre of the farm were identified and the length of the stream within the farmland was added to the NFS data. Data on the proportion of farmland within a 10m buffer of the river were also obtained specifically for this analysis.

Data on Price indices

In order to account for changes in prices, information on price indices from the CSO is utilised in this chapter (Figures IV.6 and IV.7). To reflect changes in output prices a milk price index, a total cattle price index and a sheep price index from the CSO are used (see Figure IV.6). Changes in input prices for direct costs were captured using a ‘straight feeding stuffs’ index for concentrates; an energy price index for transport expenses; a veterinary expenses index for veterinary and medical expenses including AI; a labour index for all labour expenses on the farms and finally an “other products” agricultural CSO index was used for miscellaneous direct costs. Price indexes for winter forage and pasture expenses are not produced by the CSO, so a
price index for winter forage and pasture is calculated by taking a weighted sum of the indices for the components of winter forage and pasture, which include fertiliser, crop protection, seeds, machinery hire, transport costs and labour expenses.

As can be seen in Figure IV.6, all input prices exhibited an upward trend during the period 1997-2010. Fertiliser and energy prices increased most sharply, reflecting increasing oil prices. For dairy farms in Ireland, fertiliser on average contributed 55 percent of the total expenses for pasture and winter forage so it is not surprising that the prices for straight feeding stuffs follow a similar upward trend to that of fertiliser.

Figure IV.6. Agricultural input price indexes (1997-2010)

Figure IV.7 illustrates the evolution of agricultural output prices from 1997 to 2010. With the exception of sheep, product prices rose at a rate much lower than the consumer price index (CPI). Moreover the gap between the CPI and agricultural output prices widened during the period.
IV.6 Results

**MODEL ESTIMATES**

In Table IV.5 the summary statistics for the variables used in the model described earlier in this chapter are reported. These statistics are based on a sub-sample of the NFS farms weighted to be nationally representative. The numbers reported indicate that the mean age of the dairy farmers is 52 years with the youngest reported to be 27 years of age and the oldest 85. Most of them are Teagasc clients, however, only a small fraction of them participate in the Rural Environmental Protection Scheme (REPS). The average farm size is 50.8 hectares with on average 51 livestock units in their herds. These farms have an average GM of €87K per year, which mainly comes from the dairy enterprise (€61K), with a beef enterprise GM of about €11K per year and only €433 coming from a sheep enterprise. However, it can also be seen that the minimum reported cattle and sheep enterprise GMs are negative indicating that on some dairy farms these enterprises are operated at a loss.
Table IV.5. Summary Statistics (2008)

<table>
<thead>
<tr>
<th></th>
<th>min</th>
<th>mean</th>
<th>max</th>
</tr>
</thead>
<tbody>
<tr>
<td>Farmer's Age</td>
<td>27</td>
<td>52</td>
<td>85</td>
</tr>
<tr>
<td>Teagasc client</td>
<td>0</td>
<td>0.7</td>
<td>1</td>
</tr>
<tr>
<td>REPS</td>
<td>0</td>
<td>0.3</td>
<td>1</td>
</tr>
<tr>
<td>Farm Size (ha)</td>
<td>8.1</td>
<td>50.8</td>
<td>281</td>
</tr>
<tr>
<td>Farm GM (€)</td>
<td>8,102</td>
<td>87,272</td>
<td>589,288</td>
</tr>
<tr>
<td>Dairy GM (€)</td>
<td>2,859</td>
<td>61,548</td>
<td>375,108</td>
</tr>
<tr>
<td>Cattle GM (€)</td>
<td>-5,934</td>
<td>11,587</td>
<td>107,616</td>
</tr>
<tr>
<td>Sheep GM (€)</td>
<td>-1,514</td>
<td>433</td>
<td>15,541</td>
</tr>
<tr>
<td>Dairy LU</td>
<td>11</td>
<td>51.2</td>
<td>233</td>
</tr>
<tr>
<td>Cattle LU</td>
<td>0</td>
<td>37</td>
<td>272</td>
</tr>
<tr>
<td>Sheep LU</td>
<td>0</td>
<td>1.5</td>
<td>63</td>
</tr>
<tr>
<td>Dairy Labour per LU (€)</td>
<td>0</td>
<td>6.2</td>
<td>301</td>
</tr>
<tr>
<td>Cattle Labour per LU (€)</td>
<td>0</td>
<td>0.3</td>
<td>79</td>
</tr>
<tr>
<td>Sheep Labour per LU (€)</td>
<td>0</td>
<td>0.02</td>
<td>9</td>
</tr>
<tr>
<td>Dairy Fertiliser (kg/ha)</td>
<td>0</td>
<td>218</td>
<td>682</td>
</tr>
<tr>
<td>Cattle Fertiliser (kg/ha)</td>
<td>0</td>
<td>224</td>
<td>736</td>
</tr>
<tr>
<td>Sheep Fertiliser (kg/ha)</td>
<td>43</td>
<td>216</td>
<td>850</td>
</tr>
<tr>
<td>Dairy Concentrates per LU (€)</td>
<td>26</td>
<td>251</td>
<td>879</td>
</tr>
<tr>
<td>Cattle Concentrates per LU (€)</td>
<td>0</td>
<td>165</td>
<td>705</td>
</tr>
<tr>
<td>Sheep Concentrates (€)</td>
<td>0</td>
<td>179</td>
<td>1,470</td>
</tr>
</tbody>
</table>

Source: Weighted NFS data

The estimates for gross output and direct costs per LU for dairy, beef and sheep enterprises are reported in Table IV.6. The significance levels of the estimates and the standards error are also reported in Table IV.6. The farmer’s age is negatively associated with dairy and cattle gross output per LU but has a positive relationship with direct costs per LU. This indicated that younger farmers have higher GO per LU and lower costs per LU, which indicates that they may be able to manage farms more efficiently. Teagasc clients tend to have higher GO and lower DC per LU. This is not surprising as Teagasc clients have access to specialist advisory services. Teagasc delivers services to its clients via mass communication, office/phone consultation, visits to the farms, discussion groups as well as by organising event like farm walks, seminars and other public events (Teagasc, 2015b). There are also online services available via Teagasc website (Teagasc, 2015a). The results of the estimates indicate that those farms that participate in REPS also have higher GO/LU and lower DC/LU. Teagasc and REPS estimates are only statistically significant for dairy GO/LU.

Soil and region variables are included in the analysis to account for spatial heterogeneity attributed to soil and weather conditions that may affect the decisions that farmers make on the farms. In particular soil, temperature, the amount of rain
and the hours of sunshine affect the ruminant livestock production system in Ireland by accounting for annual dry matter production (which ranges from 11-12 tonnes per hectare in the Border Region to 15 tonnes per hectare in the South-West Region) and starting dates of the grazing season (which range from the March 25th in the South-West Region to April 15-20th in the Border Region) (O'Mara, 2008).

Table IV.6. Results for Dairy Farms Production Function Estimations

<table>
<thead>
<tr>
<th>VARIABLES</th>
<th>Dairy</th>
<th>Cattle</th>
<th>Sheep</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>GO/LU</td>
<td>DC/LU</td>
<td>GO/LU</td>
</tr>
<tr>
<td>ln(Farmer’s Age)</td>
<td>-0.110** (0.053)</td>
<td>0.0032 (0.105)</td>
<td>-0.035 (0.085)</td>
</tr>
<tr>
<td>Teagasc client</td>
<td>0.0507* (0.026)</td>
<td>-0.0595 (0.052)</td>
<td>0.0182 (0.042)</td>
</tr>
<tr>
<td>REPS</td>
<td>0.053** (0.026)</td>
<td>-0.024 (0.052)</td>
<td>0.047 (0.042)</td>
</tr>
<tr>
<td>Soil 2</td>
<td>-0.045* (0.026)</td>
<td>-0.015 (0.051)</td>
<td>-0.131*** (0.042)</td>
</tr>
<tr>
<td>Soil 3</td>
<td>-0.124** (0.052)</td>
<td>0.0786 (0.102)</td>
<td>-0.0798 (0.081)</td>
</tr>
<tr>
<td>Western Region†</td>
<td>0.183 (0.150)</td>
<td>0.121 (0.293)</td>
<td>-0.131 (0.244)</td>
</tr>
<tr>
<td>Midlands Region</td>
<td>0.185*** (0.049)</td>
<td>-0.014 (0.095)</td>
<td>-0.076 (0.076)</td>
</tr>
<tr>
<td>Mid-East Region</td>
<td>0.149*** (0.052)</td>
<td>-0.356*** (0.102)</td>
<td>-0.00717 (0.083)</td>
</tr>
<tr>
<td>Dublin Region</td>
<td>0.111** (0.052)</td>
<td>-0.472*** (0.098)</td>
<td>-0.115 (0.078)</td>
</tr>
<tr>
<td>South-East Region</td>
<td>0.119*** (0.044)</td>
<td>-0.365*** (0.083)</td>
<td>-0.033 (0.066)</td>
</tr>
<tr>
<td>South-West Region</td>
<td>0.123*** (0.040)</td>
<td>-0.284*** (0.075)</td>
<td>-0.140** (0.060)</td>
</tr>
<tr>
<td>Mid-West Region</td>
<td>-0.040 (0.0840)</td>
<td>-0.272* (0.164)</td>
<td>0.043 (0.118)</td>
</tr>
<tr>
<td>ln(LU)††</td>
<td>0.028 (0.036)</td>
<td>0.155** (0.069)</td>
<td>-0.171*** (0.031)</td>
</tr>
<tr>
<td>ln(Farm Size)</td>
<td>0.0875*** (0.033)</td>
<td>-0.0883 (0.0655)</td>
<td>0.221*** (0.049)</td>
</tr>
<tr>
<td>ln(Labour/LU)††</td>
<td>0.0190** (0.009)</td>
<td>0.107* (0.064)</td>
<td>-0.0004 (0.314)</td>
</tr>
<tr>
<td>ln(Concentrates/LU)††</td>
<td>0.188*** (0.022)</td>
<td>0.121*** (0.020)</td>
<td>0.023 (0.061)</td>
</tr>
<tr>
<td>ln(Fertiliser/ha)††</td>
<td>0.129*** (0.027)</td>
<td>0.054 (0.052)</td>
<td>0.181*** (0.032)</td>
</tr>
<tr>
<td>Constant</td>
<td>5.527*** (0.296)</td>
<td>5.955*** (0.550)</td>
<td>4.884*** (0.427)</td>
</tr>
<tr>
<td>Observations</td>
<td>326</td>
<td>326</td>
<td>351</td>
</tr>
<tr>
<td>R-squared</td>
<td>0.457</td>
<td>0.176</td>
<td>0.285</td>
</tr>
<tr>
<td>F statistic</td>
<td>15.25</td>
<td>4.42</td>
<td>7.79</td>
</tr>
</tbody>
</table>

*** significant at 1 percent level; ** significant at 5 percent level; *significant at 10 percent level
† Boarder Region is a reference region in the regressions; †† enterprise specific

The Border Region is the reference region in the regression. The results indicate that farms located in Midlands, Mid-East, Dublin, South-East and South-West regions...
have on average higher dairy GO/LU and lower DC/LU comparing to the reference region.

The number of LU and size of farm are included in the regression analysis to capture economies of scale with the bigger farms normally displaying higher outputs and lower costs per LU indicating possibly higher efficiency in production.

Labour, concentrates and fertiliser used per LU are driving GO per LU up, these are the main inputs into the production on the farm. The results indicate that those farmers that input more labour benefit from the higher GO/LU and lower costs. This is explained by the fact that in dairy production animal health conditions that stem from proper nutrition and care as well as tight calving patterns that require observing animal for a prolonged periods of time and manual insemination are determinants of the output per animal (Inchaisri et al., 2010).

The amount of fertiliser used on the farm is positively associated with gross output per animal. This is not surprising as farmers apply fertiliser to support sufficient grass growth on pasture land. Fertiliser usage is also positively associated with other direct costs on the farm, which indicates that when farmers use more fertiliser per LU they also tend to use higher quantities of other inputs per LU.

**Model Validation Results.**

The results of the validation scenarios are reported in Table IV.7. The analysis is focused on the farm GM because it changes in the short run while fixed costs are only adjusted in the long term. Table IV.7 presents the farm GM and the enterprise specific GM, GO and DC (with a prefix D representing dairy, C for cattle and S for sheep enterprises) that are anticipated to result under each mitigation scenario. Baseline figures, which reflect the average farm gross margins, gross output and direct costs on the affected farms before simulations, are presented in parentheses. The simulated outcomes suggest that farm gross margin would decline significantly following a reduction of LU by 20 percent, decreasing from €63,867 down to €49,635 – a loss of around €14K on average per farm. Gross margins decline on average across all enterprises, but beef enterprise will be mostly affected. Fezzi et al. (2008), using a farm budgeting model, which is based on similar UK farm data for
year 2005, reports an average loss of £166/ha or approximately €244/ha on dairy farms due to this measure, which is broadly consistent with the findings reported here, which is €294 per hectare (17 percent difference with Fezzi et al.).

Table IV.7. Farm and enterprise GM, GO, DC under Each Scenario

<table>
<thead>
<tr>
<th>Scenario</th>
<th>FGM</th>
<th>DGM</th>
<th>DGO</th>
<th>DDC</th>
<th>CGM</th>
<th>CGO</th>
<th>CDC</th>
<th>SGM</th>
<th>SGO</th>
<th>SDC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduce LU 170kg</td>
<td>75,116*</td>
<td>72,962</td>
<td>120,997</td>
<td>48,035</td>
<td>1,983</td>
<td>15,757</td>
<td>13,774</td>
<td>171</td>
<td>246</td>
<td>75</td>
</tr>
<tr>
<td></td>
<td>(81,259)†</td>
<td>(73,964)</td>
<td>(122,776)</td>
<td>(48,813)</td>
<td>(7,124)</td>
<td>(28,623)</td>
<td>(21,498)</td>
<td>(171)</td>
<td>(246)</td>
<td>(75)</td>
</tr>
<tr>
<td>Reduce LU -20percent</td>
<td>49,636**</td>
<td>43,878</td>
<td>71,460</td>
<td>27,581</td>
<td>5,497</td>
<td>20,572</td>
<td>15,075</td>
<td>260</td>
<td>553</td>
<td>293</td>
</tr>
<tr>
<td></td>
<td>(6,3867)†</td>
<td>(55,864)</td>
<td>(89,880)</td>
<td>(34,016)</td>
<td>(7,695)</td>
<td>(24,750)</td>
<td>(17,054)</td>
<td>(306)</td>
<td>(676)</td>
<td>(370)</td>
</tr>
</tbody>
</table>

*Equivalent per hectare figures – 1687 (1848); **Equivalent per hectare figures – 1021 (1315)
†The baseline amounts are reported in the brackets, the averages are produced for affected farms only (for example, “reduce LU 170kg affects only farms that produce more than 170 kg of organic N per hectare”)

When the mitigation approach is instead to reduce organic N on the farm to a maximum of 170 kg of organic N per hectare, the GM on the affected farms would decline on average by €6K, or by €161 per hectare. This measure is more likely to affect farms engaged in relatively intensive production with stocking rates close to or over 2 LU per hectare and a higher farm GM too (Table IV.7). This measure affects beef enterprise leading to a loss in beef enterprise GM of €5,141 on average for the dairy and dairy other farms. The underlying assumption here is that the farmers drop the livestock with the lowest GM per animal. Results from the NFS sample in 2008 indicate that beef LU attract on average low GM returns on dairy and dairy other farms in Ireland and hence this enterprise is most affected. One might expect that sheep would be affected, however, as the analysis revealed – only 5 farms out of 323 dairy and dairy other farms in the sample have sheep. Beef LU attract lower GM on these farms. Hennessy et al. (2005) simulated the effect of this measure on 2002 data using representative NFS farm for 2008 and their result was a loss of between €8,000 and €3,496 in farm profit depending on how much N a farm would need to mitigate to comply with the 170 kg N/ha limit. In the model reported here this measure results in the change of FGM of €6,143 (which is comparable to change in farm profit as stated in equation 4), which falls roughly in the middle of their estimates. Hennessy et al, simulated their results into the future which may lead to differences in estimations, however, the results are quite comparable otherwise.

7 In 2005, £1 was approximately equivalent to €1.47 (http://www.x-rates.com/average/?from=GBP&to=EUR&year=2005)
Table IV.8. Percentage Change in Farm and Enterprise GM, GO, DC under Different Scenarios

<table>
<thead>
<tr>
<th>Scenario</th>
<th>FGM</th>
<th>DGM</th>
<th>DGO</th>
<th>DDC</th>
<th>CGM</th>
<th>CGO</th>
<th>CDC</th>
<th>SGM</th>
<th>SGO</th>
<th>SDC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduce LU 170kg</td>
<td>-8</td>
<td>-1</td>
<td>-1</td>
<td>-2</td>
<td>-72</td>
<td>-45</td>
<td>-36</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Reduce LU - 20%</td>
<td>-22</td>
<td>-21</td>
<td>-20</td>
<td>-19</td>
<td>-29</td>
<td>-17</td>
<td>-12</td>
<td>-15</td>
<td>-18</td>
<td>-21</td>
</tr>
</tbody>
</table>

The percentage change in GM, GO and DC as a result of the simulated policy scenarios is reported in Table IV.8. Reducing LU to achieve 170 kg organic N per hectare on average yields a decrease in farm GM of 8 percent on affected farms—mostly due to fall in GM from the beef and dairy enterprises (Table IV.8). This is despite an associated fall in costs. The results also reveal that not all farms that exceed the 170 kg of organic N per hectare threshold are equally affected.

Twenty five percent of dairy and dairy other farms (approximately 4,242 farms) exceed the limit in 2008 based on the weighted NFS data. If these farms were to reduce their emissions to comply with the stated limit, approximately 90.5 percent of these farms would have a reduction in GM and 9.5 percent would have a gain in GM due to the fact that on some farms the GM from beef cattle is zero or even negative. In Hennessy et al.’s (2005) simulations the projection is that approximately 80 percent of farms would be negatively affected by this measure by 2007 and 88 percent by 2012, thus, their projections slightly underestimated the impact.

If the farmers in Ireland were to reduce their livestock units by 20 percent, their GM per hectare would on average decrease by 22 percent. This measure would negatively affect all farm enterprises (dairy, cattle and sheep) on dairy farms. It is interesting to note here that Fezzi et al. (2008) assumed the 20 percent cost decrease due to this measure in their calculations. They based their assumption on different research papers and expert opinion. Thus, the results of the model developed here confirm their assumptions. It has to be noted that reducing LU by 20 percent would not only lead to a loss of output squeezing already narrow farm margins, but would also be inconsistent with the Food Harvest 2020 agenda, an Irish policy, which requires the growth of agricultural output by about 33 percent (DAFM).
Farm N implications under each model validation scenario

The potential N reduction that would result from the mitigation measures would have important environmental implications. Both measures offer the potential for N reduction on the farms in the form of organic N reductions (i.e. less manure). Table IV.9 summarises the amount of organic N on the farms under the two case study scenarios and the percentage changes that would be anticipated. A relatively high organic N reduction (20 percent) can be achieved by reducing the number of LU by 20 percent on Irish dairy and dairy other farms; under the LU reduction to achieve 170 kg organic N per hectare on average 18 percent of organic N can be mitigated on affected farms, or 5 percent on average across all dairy and dairy other farms (Table IV.9).

### Table IV.9. Changes in N per hectare under Different Scenarios

<table>
<thead>
<tr>
<th>Scenario</th>
<th>OrgN Kg/ha</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduce LU to 170kg N Per hectare</td>
<td>164 (200)*</td>
<td>-18</td>
</tr>
<tr>
<td>Reduce LU by 20%</td>
<td>115 (144)*</td>
<td>-20</td>
</tr>
</tbody>
</table>

*the baseline amounts of organic N per hectare

### Table IV.10. The average cost of mitigation measures

<table>
<thead>
<tr>
<th>Scenario</th>
<th>€/N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduce LU to 170kg N Per hectare</td>
<td>4.5</td>
</tr>
<tr>
<td>Reduce LU by 20%</td>
<td>10</td>
</tr>
</tbody>
</table>

In order to compare the measures, the average cost per unit of N abated through each measure is presented in Table IV.10. The average cost per unit N reduced in the scenario reducing LU by 20 percent is €10 while the cost of complying with the organic N limits is €4.5 per unit of N abated. However, the latter offers relatively small opportunities for N mitigation (20 percent versus 5 percent). These reductions translate into 26,162 tonnes of organic N abated at a cost of approximately €261 million for the scenario with LU reduction by 20 percent and 5,740 tonnes of N mitigated at a cost of almost €26 million if the target of no more than 170 kg of organic N per hectare was enforced on the dairy and dairy other farms in Ireland. Thus, if specific targets for N reductions were to be introduced, farmers may need to introduce a combination of different measures in order to achieve the targets. The
costs of a combination of methods could potentially be higher and are more difficult to assess.

**MODEL EXTENSIONS RESULTS**

Estimates for higher yield dairy cows scenario

The breeding index of the dairy herds is not collected as part of the NFS data, thus, it had to be estimated using an OLS (Ordinary Least Squares) regression as discussed above, where the breeding index constitutes part of the residual. The results of the OLS estimations for the higher yielding dairy scenario are reported in Table IV.11. The dependent variable is the log of milk output in litres of milk per dairy cow, produced on the farm. The independent variables used (constrained by data availability) include kilograms of concentrates per dairy cow and its square term; an early calving dummy variable (which takes a value of 1 for farms that have 80 percent or more calves born in January, February and March and a value of 0 for the rest of farms) and calves per cow and its square term. The number of calves per cow proved to be statistically insignificant but was retained in the regression as a proxy for cow fertility.

**Table IV.11. Results of “Higher yielding dairy cows breeding index” estimations.**

<table>
<thead>
<tr>
<th>Ln(milk output/cow)</th>
<th>β</th>
</tr>
</thead>
<tbody>
<tr>
<td>Concentrates/cow</td>
<td>0.0003***</td>
</tr>
<tr>
<td></td>
<td>(0.00003)</td>
</tr>
<tr>
<td>(Concentrates/cow)^2</td>
<td>-1.96x10^-8</td>
</tr>
<tr>
<td></td>
<td>(2.32x10^-9)**</td>
</tr>
<tr>
<td>Early Calving</td>
<td>0.0721***</td>
</tr>
<tr>
<td></td>
<td>(0.02532)</td>
</tr>
<tr>
<td>Calves/cow</td>
<td>0.4322</td>
</tr>
<tr>
<td></td>
<td>(0.2836)</td>
</tr>
<tr>
<td>(Calves/cow)^2</td>
<td>-0.1711</td>
</tr>
<tr>
<td></td>
<td>(0.1113)</td>
</tr>
<tr>
<td>Constant</td>
<td>7.9735***</td>
</tr>
<tr>
<td></td>
<td>(0.1752)</td>
</tr>
</tbody>
</table>

N=331; R^2 = 0.29; F( 5, 325) = 26.60

*** significant at 1 percent level; ** significant at 5 percent level; * significant at 10 percent level
Simulated Results

**Gross margin and net margin analysis of mitigation strategies**

Results for changes in the farm level gross margin per hectare are reported in Table IV.12. Gross margin per hectare declines on average for the subset of affected farms under most of the strategies with the exception of the improved breeding index scenario for which the GM per hectare increases from €1,203 to €1,294 (7.6 percent) and the two slurry application scenarios under which farm GM increases from €1,315 to €1,319, €1,315.13 per hectare (0.3 percent, 0.01 percent) respectively. The overall trend reflects the fact that the proposed mitigation measures will generally affect either the scale of production or will increase the costs of production resulting in lower gross margins on dairy farms. Only the strategies that increase production and/or cost efficiency yield positive economic results.

The highest negative impact is observed under the reduce LU by 20 percent scenario, with fencing-off (de-intensification) scenario yielding second highest negative result (projected loss is 22 and 8.7 percent respectively) (Table IV.12).

<table>
<thead>
<tr>
<th>Scenario</th>
<th>FGM, €/ha</th>
<th>ΔFGM,%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>1315</td>
<td></td>
</tr>
<tr>
<td>Fert-20%</td>
<td>1265</td>
<td>-3.8</td>
</tr>
<tr>
<td>ReduceLU-20%</td>
<td>1021</td>
<td>-22.3</td>
</tr>
<tr>
<td>Feed change</td>
<td>1256</td>
<td>-4.5</td>
</tr>
<tr>
<td>Slurry 1*</td>
<td>1319</td>
<td>0.3</td>
</tr>
<tr>
<td>Slurry 2**</td>
<td>1315</td>
<td>0.01</td>
</tr>
<tr>
<td>Baseline</td>
<td>1848</td>
<td></td>
</tr>
<tr>
<td>Reduce LU 170kg</td>
<td>1688</td>
<td>-8.7</td>
</tr>
<tr>
<td>Baseline</td>
<td>1373</td>
<td></td>
</tr>
<tr>
<td>Fencing-off (intensification)</td>
<td>1313</td>
<td>-4.4</td>
</tr>
<tr>
<td>Fencing-off (deintensification)</td>
<td>1269</td>
<td>-7.6</td>
</tr>
<tr>
<td>Breeding Index</td>
<td>1203</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1294</td>
<td>7.6</td>
</tr>
</tbody>
</table>

*change in the timing of slurry application to spring;  
**change the method of slurry application from SP to TS
A change in feed composition leads on average to a loss of €59 (4.5 percent reduction) per hectare this is due to assumed increases in feed cost which increase the direct costs on the farms. Reducing fertiliser by 20 percent leads to a loss of €50 per hectare (3.8 percent). Fezzi et al. (2008) also explored the cost of reducing the amount of fertiliser by 20 percent. Their results show a reduction of GM on dairy farms by £132 per hectare (approximately €194). This result is higher than in the model presented here. However, in their paper Fezzi et al. (2008) state that in another UK study by Cuttle et al. (2007) the cost of this measure is much lower - £35 per hectare (approximately €51), which is very close to the result reported in Table IV.12. Fezzi et al. (2008) attribute the differences between their results and that of Cuttle et al. (2007) to the differences in assumptions used in the two models. This could also be the case here; however, the differences could also be attributed to the differences in the farm data and prices. Fencing off streams results in a loss of €104 (7.6 percent reduction) per hectare on affected farms if the farmers reduce stocking rates proportionally to the reduction in land farmed. However, if the farmers decide to keep their stocking numbers and intensify the production on the remaining land the loss can be mostly offset – reducing to €60 per hectare (4.4 percent reduction). However, this may mean that environmental objectives are not then being met.

Farmers may be more concerned with how the mitigation measures would affect Farm Net Margin (FNM) rather than GM. FNM is GM less overhead costs, which can be substantial on the farms. The effect of the measures on the FNM is reported in Table IV.13. The effect on FNM and FNM per hectare is more dramatic than that on the GM. The reduction in LU by 20 percent potentially leads to FNM reduction from €441 per hectare to €147 per hectare, which translates into 63 percent reduction in FNM/ha due to this measure. Reducing LU to achieve 170 kg of N per hectare leads to a FNM/ha loss of almost 24 percent and fencing off stream leads to a loss of 23 percent of FNM/ha on average per farm. Changing the breeding index leads to gains in FNM, however, the initial investment could be high and is not taken into account here due to uncertainty regarding the costs.
Table IV.13. Effect of the N Mitigation Measures on Farm Net Margin and Farm Net Margin per hectare

<table>
<thead>
<tr>
<th>Scenario</th>
<th>FNM/ha</th>
<th>FNM</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>441</td>
<td>21,066</td>
</tr>
<tr>
<td>Fert -20%</td>
<td>391</td>
<td>18,701</td>
</tr>
<tr>
<td>Reduce LU -20%</td>
<td>147</td>
<td>6,835</td>
</tr>
<tr>
<td>Feed change</td>
<td>381</td>
<td>18,364</td>
</tr>
<tr>
<td>Slurry 1*</td>
<td>444</td>
<td>21,232</td>
</tr>
<tr>
<td>Slurry 2**</td>
<td>441</td>
<td>21,072</td>
</tr>
<tr>
<td>Baseline</td>
<td>679</td>
<td>28,754</td>
</tr>
<tr>
<td>Reduce LU 170kg</td>
<td>519</td>
<td>22,611</td>
</tr>
<tr>
<td>Baseline</td>
<td>458</td>
<td>22,080</td>
</tr>
<tr>
<td>Fencing-off (intensification)</td>
<td>398</td>
<td>19,871</td>
</tr>
<tr>
<td>Fencing-off (de-intensification)</td>
<td>354</td>
<td>17,768</td>
</tr>
<tr>
<td>Baseline</td>
<td>377</td>
<td>16,715</td>
</tr>
<tr>
<td>Breeding Index</td>
<td>548</td>
<td>23,591</td>
</tr>
</tbody>
</table>

*change in the timing of slurry application to spring; **change the method of slurry application from SP to TS

Farm N implications of mitigation strategies

Table IV.14 summarises the amount of organic, chemical and total N per hectare on the affected farms under each scenario and the percentage changes from the baseline. All scenarios produce a reduction in total N (except for the case of intensification of production after fencing off streams, when N per hectare on the farms is expected to increase and, thus, cannot be considered a mitigation option). However, some options result in larger total N per hectare reduction than others. Relatively high total N reduction can be achieved by reducing chemical N fertilizer application by 20 percent (12 percent N reduction) and decreasing the number of LU by 20 percent (7.9 percent N reduction). Increasing the breeding index of dairy animals would allow farmers to abate 3.9 percent of total N on affected farms. Under the LU reduction to 170kg N per hectare strategy, on average 17.7 percent organic N can be mitigated. However, the cost of abating N under the different strategies varies across farms. This is discussed in the following sections.
Table IV.14. Average farm N under different scenarios (kg/ha;%)  

<table>
<thead>
<tr>
<th>Scenario</th>
<th>OrgN</th>
<th>ChemN</th>
<th>TotalN</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>kg/ha</td>
<td>%</td>
<td>kg/ha</td>
</tr>
<tr>
<td>Baseline</td>
<td>143.8</td>
<td>-</td>
<td>220.1</td>
</tr>
<tr>
<td>Fert-20%</td>
<td>143.8</td>
<td>0</td>
<td>176.1</td>
</tr>
<tr>
<td>Reduce LU 170kg</td>
<td>136</td>
<td>-5.4</td>
<td>220.1</td>
</tr>
<tr>
<td>Reduce LU -20%</td>
<td>115</td>
<td>-20</td>
<td>220.1</td>
</tr>
<tr>
<td>Slurry 1*</td>
<td>143.8</td>
<td>0</td>
<td>218.7</td>
</tr>
<tr>
<td>Slurry 2**</td>
<td>143.8</td>
<td>0</td>
<td>217</td>
</tr>
<tr>
<td>Feed change</td>
<td>122.8</td>
<td>-14.6</td>
<td>220.1</td>
</tr>
<tr>
<td>Baseline</td>
<td>199.3</td>
<td>-</td>
<td>307</td>
</tr>
<tr>
<td>Reduce LU 170kg</td>
<td>163.9</td>
<td>-17.7</td>
<td>307</td>
</tr>
<tr>
<td>Baseline</td>
<td>142.6</td>
<td>-</td>
<td>223.4</td>
</tr>
<tr>
<td>Fencing-off (de-intensif)</td>
<td>139.3</td>
<td>-2.3</td>
<td>218.5</td>
</tr>
<tr>
<td>Baseline</td>
<td>142.3</td>
<td>-</td>
<td>208.6</td>
</tr>
<tr>
<td>Breeding Index†</td>
<td>128.6</td>
<td>-9.6</td>
<td>208.6</td>
</tr>
</tbody>
</table>

*change in the timing of slurry application to spring  
**change the method of slurry application from SP to TS  
†reduction in dairy herd without a loss of farm output through improving genetics of herd to achieve higher milk output per a dairy cow

Cost per Unit Abated Results

The changes in farms’ income and/or N per hectare at farm level as reported in Tables IV.12 and IV.14 do not in themselves allow for cost-efficiency comparisons. CPUA for each strategy are calculated (except for intensification of farming due to fencing off streams) and reported in Table IV.15 for comparison. The results represent the cost in € per kg of N abated per hectare, ranging from a cost of over €10.5 per kg N for the fencing-off (de-intensification) strategy to a saving of €5.5 per kg N for the improved cow breeding index strategy.

Table IV.15. Cost per Unit Abated by scenario

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Cost (€ per kg N Ha(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fert-20%</td>
<td>1.22</td>
</tr>
<tr>
<td>Reduce LU 170kg</td>
<td>4.5</td>
</tr>
<tr>
<td>Reduce LU -20%</td>
<td>10</td>
</tr>
<tr>
<td>Feed change</td>
<td>4</td>
</tr>
<tr>
<td>Fencing-off (de-intensif)</td>
<td>10.5</td>
</tr>
<tr>
<td>Slurry 1*</td>
<td>-2.5</td>
</tr>
<tr>
<td>Slurry 2**</td>
<td>-0.03</td>
</tr>
<tr>
<td>Breeding Index†</td>
<td>-5.5</td>
</tr>
</tbody>
</table>

*change in the timing of slurry application to spring  
**change the method of slurry application from SP to TS  
†reduction in dairy herd without a loss of farm output through improving genetics of herd to achieve higher milk output per a dairy cow
The higher the CPUA result, the more expensive it is to abate each unit of total N for farmers. The negative CPUA for the slurry scenarios and the improved breeding index scenario, indicate that these strategies on average would not only produce a decrease in total N introduced into the environment but would also lead to a reduction of costs on farms.

**Figure IV.8. Ranking of the CPUAs for dairy farms (€/kg N)**

The ranking of the CPUAs of each mitigation strategy is presented in Figure IV.8. Changing the timing or method of slurry application and increasing breeding index of the dairy herd are the most cost-efficient results for N abatement. However, further investigation reveals that country average CPUA numbers hide the diversity of impacts that each strategy would have across individual farms. Figure IV.9 shows CPUAs for each strategy for all the individual dairy farms in the NFS sample with farms ranked by the most cost-effective aggregate CPUA – improved breeding index.

On Figure IV.9 on axis X the farms are ranked starting with the farm that has the lowest CPUA under the breeding index scenario and finishing with the farm that has the highest. The costs of the rest of the scenarios are either above or below the breeding index cost.

---

*Note that in Figure IV.9 some scenarios only affect a subset of farms (e.g. fencing-off) and hence there are breaks in the lines representing these scenarios.*
If the ranking of mitigation strategies was the same for all the farms in the dataset, the lines would be parallel and would not cross. However, as can be seen in Figure IV.9, the lines cross a number of times indicating different magnitude and ranking of the CPUAs for individual farms. There is no strategy that is strictly dominant for all dairy farms in the sample. This has very important policy implications and suggests that any policy measure introduced in a rigid manner will not produce the economically most efficient result across individual farms.

**Sensitivity Analysis Results**

Figure IV.10 displays the results for the first sensitivity analysis, showing the average CPUA for dairy farms for each of the seven policy scenarios while holding quantities constant at 2008 levels and applying prices from the years 1997 to 2009. As can be seen from Figure IV.10, the magnitudes of the CPUAs are broadly stable over time up to 2008 and the ranking for many of the scenarios is generally consistent. However, in a small number of cases the curves do cross suggesting that price changes that have been experienced in the recent past would have altered the policy recommendation resulting from the CPUA analysis. The relative stability of the CPUAs up to 2008 may offer some comfort to decision-makers regarding the
robustness of CPUA analysis to temporary short term price changes which do not result in farmers altering behaviour.⁹

Figure IV.10. Average CPUAs with the Price changes holding quantities constant at 2008 levels

However in 2009 the large drop in milk output prices leads to substantially lower CPUAs for a number of the scenarios – particularly those involving reductions in livestock numbers. Thus, whilst the CPUA displays some evidence of robustness, nonetheless results may be sensitive to large changes in price levels even when assuming there is no alteration in farmer behaviour (a fairly strict assumption).

When the quantities are kept constant, Scenario 7 (breeding index) emerges as the most cost efficient, averaging across all dairy farms. It is possible to say that, on average, two measures are clearly the most "expensive" during this period – Scenario 5 (fencing off the adjacent streams) and Scenario 3 (reducing LU by 20 percent). The fluctuations of the curves seem to follow the trends in the relevant input/output prices, the cost of the reduction in the dairy LU follows the trend in milk and cattle output prices – going down when these prices go down and going up then these prices go up. Reduction of LU to achieve 170 kg of N per hectare follows the cattle

⁹The values of the CPUA are displayed in Appendix 1.
output price trend as this measure would lead to reductions in cattle numbers. The cost of the more efficient slurry application measure seems to slightly fluctuate in the opposite direction to fertiliser prices as would be expected and the improved breeding index scenario reflects the prices on the feeds.

Figure IV.11. Changes in CPUA (1997-2009)

Figure IV.11 displays the results for the second sensitivity analysis, showing the average CPUA for dairy farms for each of the seven policy scenarios taking account of changes in production decisions due to price changes. It is immediately evident that the CPUA curves are more volatile as are the ranking of the seven policy scenarios once changes in input and output quantities are accounted for.  

Land and fertiliser are the most important inputs into dairy production, hence if these are reduced considerable loss of production and perhaps profits is to be expected. From Figure IV.11 it is clear that the most cost efficient measures are the slurry application measures and the increase in higher yielding cows measure (Scenarios 6 and 7). These measures do not affect the volume of production but make the use of inputs more efficient. Since these measures do not affect the volume of production they may yield multiple-dividends - reduced production costs and reduced negative environmental impacts, i.e. less GHGs emission and less N loss to water streams.

The values of the CPUA are displayed in Appendix 2.
However, these approaches tend to lead to relatively small percentage reductions in total N – 3.9 percent from increasing the breeding index and 0.4 percent (slurry 1) from more efficient slurry application suggesting additional measures may be necessary if the N reduction targeted is larger.

From 2002 onwards changing the breeding index offers the lowest CPUA, however between 1997 and 2001 this is not the case. Also the actual CPUA of the two lowest cost abatement approaches (Scenarios 6 and 7) also exhibit considerable volatility during the period which has implications for inferences regarding the total cost of abatement. Thus, not only is the optimal policy instrument likely to fluctuate according to prevailing market conditions and farmers’ responses but so too is the cost of abatement. The ranking of the most expensive abatement options (Scenarios 3 and 5) also fluctuates over time although the changes in the level of CPUA are less dramatic. The results support the hypothesis that the ranking of the CPUA is sensitive to changes in prices and to farmers’ production decisions and thus, while the CPUA methodology has proven to be a helpful tool in informing policy making, a degree of caution is advisable when presented with a CPUA analysis based on currently prevailing market conditions. Sensitivity analysis based on historic farm behaviour and/or anticipated future market conditions should be routinely carried out to assess whether the policy choice is robust to other market conditions or that the policy itself is designed to be flexible in the face of changing conditions.

The results of the analysis reported in this chapter are consistent with the previous literature and lead to similar conclusions. There are a number of microsimulation models concluding that flexible mitigation approaches that allow abatement efforts to vary from producer to producer yield more favourable economic results than those that are enforced uniformly across all the producers. For example, (Doole et al., 2013) and (Doole, 2012) concluded that uniform policies are at least three times more expensive than differentiated approaches, while a threshold policies that require farmers to emit below a certain threshold can be up to four times more expensive.

**IV.7 Discussion and Conclusion**

Of all human-derived N emissions, agriculture is the most significant single cause for current high N levels in the biosphere. Unfortunately the impressive advance in
agricultural productivity has come at a high price in terms of environmental degradation. For instance, N losses from the agricultural sector have been held as a contributory factor behind soil and water acidification, contamination of surface and groundwater resources and loss of biodiversity (Knudsen et al., 2006). Truly sustainable agricultural production will require countries to achieve the twin goals of meeting the growing demand for agricultural produce while also reducing negative environmental impacts from the agricultural production process. The initial policy objective to protect drinking water quality had to be extended over recent decades to account for the damage caused to the environment by excess N compounds. This is reflected in new thresholds for ground- and surface water N concentrations, which have evolved from simple parameter limits to more complex water quality parameters which reflect hydrological, hydro-geological, hydro-morphological and biological properties as enshrined in the EU Water Framework Directive (Directive 2000/60/EC). There are now a wide range of farm management measures which can be employed to reduce N pollution. However, the impact of these measures must be considered before any policy recommendations can be made.

From an environmental point of view a wide range of N mitigation options are available to decision-makers in designing rational economic responses to the continuous pressure to reduce N losses from farmed land. A plethora of research exists which describe different mitigation options to reduce N losses to watercourses from agricultural land (Byström, 1998, Cuttle et al., 2007, Novotny, 2003, Ritter and Shirmohammadi, 2000, Velthof et al., 2009). However, there is a great deal of uncertainty regarding the economic impacts that these measures would have on individual economic agents and on farm incomes in particular. In this chapter a microsimulation model that would help to asses such impacts is developed. Initially, a case study analysis of two mitigation measures is explored namely: 1) a stocking rate reduction to achieve a maximum level of organic N of 170 kg per hectare; 2) a 20 percent livestock reduction. Both measures discussed could potentially lead to a reduction in N loss from agricultural land. These measures were chosen as they have been the basis of previous studies using other microsimulation models, thus, are suitable for assessing the consistency of the model specification with the existing research literature. The results are compared in detail to the results by Fezzi et al. (2008) for a 20 percent LU reduction and Hennessy et al. (2005) for a LU reduction
to achieve 170kg of organic N per hectare scenario; however, where appropriate, other similar studies are also mentioned.

The results of the model described in this chapter are consistent with those previously obtained by Lally et al. (2009), Fezzi et al. (2008) and Hennessy et al. (2005) and confirm that the measures would lead to a reduction in farm gross margins if introduced. In addition the model developed in this chapter allows the volume of N mitigated to be assessed and hence an average cost per unit of N mitigated to be calculated.

The model is then further extended to seven potential strategies for N reduction at the farm scale. All of the mitigation options discussed in this chapter could potentially lead to reduction in N losses (keeping other factors constant) from agricultural land. In addition, in some instances a double-dividend may be achieved through a reduction in N application without affecting output. However, it should be noted that this analysis explored N reduction at a farm scale and the subsequent cost to the affected farmer and does not explore the environmental benefit of these measures. It does not follow that all reductions in N at the farm level would be fully realized as benefits to the local aquatic environment. The relationship between the inputs and losses of N may not be linear and may differ spatially, seasonally and be weather dependent; also N losses may happen due to poor management rather than the actual quantities of N used on the farm. Notwithstanding this, it is anticipated that a proportional reduction of losses would happen as N that is not introduced to the environment cannot be lost. The environmental effectiveness of the measures would be dependent on a range of local bio-physical conditions including soils and hydrological pathways (Jordan et al., 2012). This chapter concentrated on the cost of seven potential N reduction strategies but an overall environmental effectiveness of these measures would require an integrated economic and hydrological approach (Fezzi et al., 2010).

There are a number of microsimulation models that assess the impact of the nitrogen mitigation models on farm income, however, most of them are either use hypothetical or average farm data (Doole, 2012, Doole et al., 2013, Hennessy et al., 2005, Ramilan et al., 2011) or limit their study to a particular catchment (Fezzi et al., 2010). In this chapter the simulation is carried out using real farm Irish data. The methodology used
in this chapter extends existing research in the area and offers an additional tool for decision-makers for efficient policy design. Results from this study indicate (on average) that farm GM per hectare declines under all scenarios except in the higher yielding dairy cows (increase of 7.6 percent) and the two slurry scenarios (increases of 0.3 percent, 0.01 percent), which allow efficiency gain on the farms. These findings are consistent with those of Hennessy et al., (2005) and Fezzi et al. (2008), whose studies also found that the measures that would require livestock reduction would lead to a loss in farm gross margins. These three cost reducing measures can potentially represent a win-win scenario of increased returns to production while reduced risk of N loss. The smallest loss (3.8 percent) in the farm GM per hectare was produced by the 20 percent fertiliser reduction scenario. The largest decline in farm GM per hectare is observed under the 20 percent reduction in livestock numbers scenario at 22 percent. If farmers were to use a lower N feed composition in an effort to decrease N excreted by the dairy cattle, they would experience on average 4.5 percent decline in the dairy enterprise GM, which would be due to a 25 percent increase in feed costs, thus the cost of this measure is dependent on the feed prices prevailing on the market.

Increasing the breeding index of dairy cows can potentially allow the achievement of higher milk yields per animal and, thus, reducing the herd size without the loss of output. This would produce a 7.6 percent increase in the enterprise GM due to the direct cost reduction associated with the reduction in animal numbers. However, it has to be noted that due to data limitations a number of assumptions were made in modelling this scenario. Extending this scenario to a bio-economic model would allow better predictions of the impacts of this measure on the farm incomes.

All the mitigation measures produce a reduction in total N, however, some allow for higher total N reduction than others. Relatively high total N reduction can be achieved by reducing fertilizer application by 20 percent and decreasing the number of LU by 20 percent - 12 percent and 7.9 percent respectively. Increasing the breeding index of dairy animals would allow farmers to abate 3.9 percent of total N.

CPUA analysis indicates that a number of the mitigation measures (changing season or method of slurry application and increasing the breeding index) have negative signs which means that these mitigation strategies on average would not only result
in a decrease in total N introduced into the environment but would reduce costs at farm level. The mitigation measures with the largest average CPUA were LU reduction by 20 percent and fencing of watercourses, caused by the large abatement costs associated with these measures as the higher the positive CPUA the more expensive it is to abate each unit of total N for farmers.

The analysis of CPUAs ranking at individual farm level indicates that no strategy is strictly dominant for all farms across the sample. Individual farms have their own CPUA ranking of the measures. Therefore the mitigation strategy that is the most cost-effective at the aggregate level may not be the most cost-effective at the micro (farm) level. Efficient policy should reflect this and allow flexibility and innovation at farm level to respond to any policy objective in this area. However, there may be increased transaction costs to establish the most efficient mitigation strategies at farm level.

CPUA analysis is now widely used to assess the merits or otherwise of various policy options for reducing harmful emissions from agriculture both for human health and the environment (Blok et al., 2001b, Dominic et al., 2009, Moran et al., 2008). Growing populations and incomes in emerging and developing countries will add significantly to the demand for agricultural commodities in the coming decades (Fao et al., 2011). At this stage, it is unclear how elastic the supply response from the agricultural sector will be in response to this increase in demand but it seems highly probable that there will be some impact on price. Biofuel demand could also affect the supply of agricultural goods and hence increase price. On the cost side, there are a number of external factors that could significantly impact farm production and hence the price of agricultural commodities. For instance, agricultural commodity prices are becoming increasingly connected with oil prices and furthermore climate change could also lead to more extreme weather events such as droughts, heat waves and floods which can threaten supply and in turn affect food prices. One issue that is frequently overlooked in CPUA analysis is the sensitivity of the results to changes in input and output prices. The research conducted in this chapter has demonstrated that given farm heterogeneity it is important to tailor packages of mitigation measures at the individual farmer. Using microsimulation techniques the current study examined the sensitivity of CPUA ranking of seven N mitigation measures to changes in
agricultural input and output prices and finds that CPUA curves and their ranking across the difference scenarios are sensitive to input and output prices.

A major limitation of the model described in this chapter is that it does not presently allow for a combination of the mitigation measures to be considered - this may be needed if specific N reduction targets are to be introduced. As a static model, it does not allow for dynamics in farmers’ behaviour. Thus, further extensions to the model are necessary to improve the model’s capabilities.

The results of the case study scenarios reported in this chapter should be interpreted with care as the results of the model are conditional on the validity of the assumptions underlined. The presented results are average results for all dairy farms in the country and hence may obscure differences in the impacts of the considered N mitigation measures for individual farms. Notwithstanding these cautionary remarks, the model represents a considerable advance in determining the costs and other impacts of the mitigation measures.

Given the dynamic nature of the agricultural sector flexibility will be crucial when it comes to formulating farm management actions to reduce N pollution. Policy should not only be targeted at the individual farm level to account for farm heterogeneity, but should also be designed in a flexible manner in recognition of the impacts of changes in agricultural input/output prices. Thus, policy-makers who must choose a particular policy response based on an CPUA analysis should be cognizant of the extent to which changes in commodity prices can alter the conclusions of their analysis and hence the optimal policy instrument. This may in fact require that policy instruments be flexible as rigid regulations may lead to an inefficient allocation of resources and loss of income for the farming community. A more flexible set of policy instruments may increase efficiency while maximising the achievable pollution reduction.
### IV.8 Appendix 1.

**Table IV.16. CPUA of N reduction measures (1997-2009) Constant Q**

<table>
<thead>
<tr>
<th>Year</th>
<th>fert-20%</th>
<th>reduce LU -20%</th>
<th>feed change</th>
<th>slurry 1*</th>
<th>slurry 2**</th>
<th>reduce LU 170kg</th>
<th>fencing-off (de-intensif)</th>
<th>breeding Index</th>
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### IV.9 Appendix 2.

**Table IV.17. CPUA of N reduction measures (1997-2009)**

<table>
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<th>reduce LU -20%</th>
<th>feed change</th>
<th>slurry 1*</th>
<th>slurry 2**</th>
<th>reduce LU 170kg</th>
<th>fencing-off (de-intensif)</th>
<th>breeding Index</th>
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Chapter V.

**Using a Spatial Microsimulation Model to Estimate the Potential Economic Impact on Agriculture of Possible Freshwater Pearl Mussel Protection Strategies**

V.1 Introduction

As has been highlighted in the previous chapters, the protection of water resources is an important topic on the agenda of many policy-makers around the world and in the EU Member States in particular. The need to protect water bodies was originally driven by the odour and visual pollution which had become apparent and barely tolerable (Novotny, 2003). As a result of research and increased social awareness of these issues, demand for protection of water bodies across Europe developed. Protection not only involves a physical cleaning of the water in the rivers and streams but also a more holistic approach to water body management, where the water, flora, fauna and morphological structure are all treated as being of a high value and importance. One result of this new understanding is an introduction of new policies such as the Water Framework Directive (WFD) (Directive 2000/60/EC). This directive recognises the need for an integrated holistic approach to water protection and demands cooperation at the EU level in order to achieve the targeted improvements in water quality. It also prohibits any further deterioration of the water bodies and sets penalties for non-compliance (Directive 2000/60/EC).

A number of studies have sought to estimate the cost of the WFD implementation internationally. Most of this research focuses on non-point pollution sources and on agricultural activities in particular; however some research is of a general nature, like Del Saz-Salazar et al. (2009), who estimated costs and benefits of restoring water quality. In the UK Fezzi et al. (2010) conducted an econometric analysis to estimate the cost of nitrate reduction measures. In Belgium, Cools et al. (2011) coupled a hydrological and economic optimisation model to create a framework to evaluate the cost-effectiveness of nitrogen emission reduction measures. Gómez-limón and Martin-Ortega (2013) conducted an economic analysis of the WFD implementation in Spain and outlined the weaknesses in the existing economic methods used to estimate the economic costs. Lescot et al. (2013) conducted a spatially-distributed cost-effectiveness analysis framework comparing various agro-environmental
measures to control pesticide pollution in surface waters in France. In an Irish context, a number of studies have investigated the costs of the possible WFD measures that would potentially reduce pollution from agricultural activities (Doody et al., 2012, Lally et al., 2009). As has been shown in chapter IV mitigating pollution from agricultural activities will lead to a loss of income for many farmers. This chapter builds on the model developed in chapter VI and the spatial microsimulation model described by O'Donoghue et al. (2012), O'Donoghue et al. (2014a), O'Donoghue et al. (2014b), O'Donoghue et al. (2014c) and estimates the cost of the proposed measure to protect specific species of the fresh water pearl mussel (FWPM). Microsimulation models present an invaluable alternative where conducting a real-life data collection is time or cost-prohibitive. Furthermore, spatial microsimulation models contain geographically referenced information that links micro-data to specific location (O'Donoghue et al., 2012).

In this chapter the costs associated with protecting the endangered species of FWPM Margaritifera Margaritifera (MM) and Margaritifera Duravensis (MD) are estimated by combining a microsimulation model developed in chapter IV with spatial microsimulation model developed by O'Donoghue et al. (2012). The decline of these mussels coincides with intensification of agricultural activities in the second half of the XXth century. Thus, protecting this mollusc would require mitigation of non-point pollution from agricultural activities.

These mussels are protected under the Habitats Directive (Council Directive 92/43/EEC), the Wildlife Acts (Wildlife Act No 39 of 1976) and the Water Framework Directive (Directive 2000/60/EC). These species have very complicated reproduction cycles and require very high water quality for successful reproduction, particularly for the survival of juvenile mussels (Bauer, 1987, DEHLG, 2010). The presence of these pearl mussels in their natural habitat of fresh water rivers has been used as an indicator of water quality and the declining numbers of once abundant FWPM indicates declining water quality and the need for habitat protection policy in Ireland. This mollusc is not only a very sensitive organism that signals the presence of water pollution problems but is also a unique and endangered species that has to be preserved for future generations in its own right.
Ireland has 46 percent of the individual EU FWPM (DEHLG, 2010). This species is currently in decline throughout Ireland and the rest of Europe. Sedimentation, turbidity and nutrient enrichment from a variety of anthropogenic activities have contributed significantly to this decline (Österling et al., 2010, Ostrovsky and Popov, 2011, Cooksley et al., 2012). Under the WFD, water quality requirements within FWPM special areas of conservation must be met by 2015. In order to meet the requirements of the Habitats Directive (Council Directive 92/43/EEC) and the WFD (Directive 2000/60/EC) associated with the FWPM, Sub-Basin Management Plans (SBMPs) have been established for each of the FWPM sites. These plans deal with impacts on FWPM arising from all land uses and activities in the FWPM catchments and discussed in more detail in a later section.

Agriculture has been identified in the literature as one of the many sources of non-point pollution (O’Donoghue et al., 2010, Schulte et al., 2006, Buckley and Carney, 2013, Buckley et al., 2012) and the SBMPs also cite agriculture as a sector putting environmental pressure on the catchments' ecology (DEHLG, 2010). Irrespective of the size of this possible contribution from agricultural activities, farmers are required to participate in efforts to reduce environmental pressure on FWPM, along with other sectors. The measures that are needed to protect the pearl mussel species will impact on the agricultural sector since research indicates that the FWPM is sensitive to nutrient enrichment and sediment pollution which are known to originate from agricultural activities (DEHLG, 2010, Merrington et al., 2003, Novotny, 2003, Ritter and Shirmohammadi, 2000, Novotny et al., 2005).

There are other human influences that contribute significantly to environmental pressures on FWPM – such as the use of septic tanks and sewage on site treatment plants (Cooksley et al., 2012, Ni Chathaín, 2009). However these non-agricultural influences are not considered in this chapter notwithstanding the fact that they may significantly impact the ecology of water streams.

In this chapter the cost of introducing the measures to protect the Irish populations of the freshwater pearl mussels is estimated in a spatial microsimulation framework. In face of more and more limited economic resources there is a need for efficient resources allocation and, thus, the cost of each project needs to be known before the allocation decisions can be made. The costs of protecting FWPM are likely to be
localised in character and confined to the areas where these organisms are present. The use of spatial microsimulation models in different research areas is well documented and these models can be used to generate data as well as providing a framework for spatial policy impact analysis. O'Donoghue et al. (2012) conducted a comprehensive review of these models and their applications. These models have been originally applied to evaluate taxation systems in the US in 1960s (O’Donoghue, 2001). Later spatial microsimulation models found their way into the areas such as healthcare (Edwards and Clarke, 2009, Procter et al., 2008, Smith et al., 2006), transport (Bradley and Bowman, 2006, De Palma and Marchal, 2002, Liu et al., 1995), housing and labour markets’ (Clarke, 1996, Hooimeijer and Oskamp, 2000), and finally in environmental and climate change analysis (Kruseman et al., 2008a, Kruseman et al., 2008b, Hynes et al., 2009b, Svoray and Benenson, 2009).

There is still a gap in the literature regarding the costs of protecting FWPM despite the fact that there is a comprehensive literature on its biology and ecology. The focus is on agricultural activities because existing policy responses to protect the FWPM will impact heavily on the farms located in the FWPM catchments. In Ireland, many farmers are motivated by a desire to preserve the culture and tradition of farming in addition to profit motives, reflected in the relatively high proportion of farming that is loss-making. Thus, regulations that impact on the agricultural sector may further erode the position of farming leading to welfare implications for farming communities. Another consideration which must be borne in mind when determining optimal policy responses to endangerment of FWPM is the necessity of increasing food production in light of the projected increases in the population of Earth. This projected increase in the population will lead to an increase in demand for agricultural outputs. Food Harvest 2020 agenda includes increase in the value of agricultural output by of 33 percent by 2020 (DAFM, 2010). In light of the implications that many FWPM measures are anticipated to have for agricultural production, this chapter explores the effects of these protective measures on the income of farmers in the catchments where the populations of FWPM are present. Moreover, unlike the air pollution, the water pollution impacts are often localised, thus, the measures to mitigate such pollution and as a result the economic impacts will also be localised. Thus, spatial microsimulation models need to be utilised in the
assessment of the FWPM protection measures that will be localised to the catchments where these organisms are present.

Following the recommendations put forward in the SBMPs, five measures that would reduce farming activities in the FWPM catchments are considered: 1) the reduction of fertiliser by 20 percent; 2) the reduction of livestock units (LU) to achieve 170kg of organic N per hectare; 3) a 20 percent LU reduction; 4) switching from tillage to beef production and 5) the fencing off of water body banks by 10m, 25m and 50m. The economic impacts of each measure are modelled using the microsimulation approaches described in detail in chapter IV of this thesis. The microsimulation model produces a farm-level estimate of the economic impact for farms contained in the National Farm Survey (NFS). In order to simulate the effect that these measures have on farm gross margin for all farms in Ireland a spatial microsimulation model, the Simulation Model of the Irish Local Economy (SMILE) as described by O'Donoghue et al. (2012) is also used. This further allows one to isolate the economic effects of the measures on farms located in the FWPM catchments. Costs are compared on the basis of cost in Euro per unit of Nitrogen abated per hectare or CPUA (as defined in chapter IV).

The chapter is structured as follows: section V.2 provides some background information on the ecology of the FWPM and the main problems associated with the environment of these endangered species. Section V.3 then briefly discusses the sub-basin management plans and the measures for controlling pollution from agricultural land that would reduce environmental pressures on FWPM habitats. In section V.4 the methodology is described. Section V.5 introduces the datasets that were utilised for the analysis in this chapter. The results are presented in section V.6. Finally, section V.7 contains the discussion of the results and conclusions are drawn.

V.2 Ecology of the Freshwater pearl mussel and environmental issues associated with it.

The measures that have been proposed in the literature to protect the FWPMs are based on the ecology of these species. The FWPM (MM and MD) is a type of mollusc that is known to be present in a small number of places in the world. MM has reportedly been found in Austria, Belgium, Czech Republic (critically endangered),
Denmark, Estonia, France, Germany, Lithuania and Poland (believed to be extinct), Portugal, Spain, Scandinavia, Canada, USA, UK and Russia. Ireland is known to have 46 percent of all populations of MM in Europe (DEHLG, 2010, IUCN, 2009, Makhrov, 2011). MD is a unique type of freshwater pearl mussel that can be found only in the Nore catchment in Ireland. Thus, Ireland is an important region for efforts to preserve FWPM.

Experts now estimate that over 90 percent of all Margaritifera individuals died out during the 20th century and that there are now only small ageing populations left across the EU and other parts of the world (DEHLG, 2010). The declining numbers of FWPM species are attributed to ecological and environmental factors that make reproduction and survival of this mollusc more and more difficult.

In Ireland, Magaritifera is a protected species under the 1976 Wildlife Act and it is also listed in the Habitats Directive (Council Directive 92/43/EEC). It is also protected by the International Union of Conservation of Nature (IUCN) red data book as endangered worldwide (Baillie, 1996, WFD Ireland, 2005, IUCN, 2009). FWPMs are valuable to the environment due to their ability to filter water from pollutants that naturally occur in the environment. One Margaritifera can filter up to 50 litres of water a day (Ziuganov and Nezlin, 1988). However, pearl mussels are sensitive to man-made pollutants and require very high quality of water with low levels of nutrients/sediment and clean river beds for survival and for sustainable reproduction. It is believed that the condition of the water in the river needs to be very close to its natural state (unimpacted by human activities) to allow FWPM reproduction and survival (DEHLG, 2010).

FWPM can live up to 130 years in temperate zones of Europe and up to 200 years in the sub-Arctic in rivers with soft water with low levels of calcium (Geist, 2010, Hartmut and Gerstmann, 2007). It has a somewhat complicated method of reproduction - female species release high numbers (between 1 individual and 28 million (Bauer, 1987, Ross, 1988)) of glochidia during two days sometime between July and September; the glochidia then has only 24 hour to find a salmonoid host. Young and Williams (1983), Young (1991) report that in the wild glochidia fail to find a host in 99.9996 percent cases and even in the event of finding a host the same authors report the loss of 95 percent of glochidia while attached to the host. Those
that survive, develop into young mussels, fall off the fish and are buried for 5-10 year in the river-bottom gravel. Only 5 percent of the FWPM that survive to this stage reach the age of 5-6 years (Arvidsson et al., 2012, DEHLG, 2010, Geist et al., 2006).

There are four main threats to the sustainability of the FWPM population in Ireland, 1) siltation; 2) nutrient enrichment; 3) acidification; 4) toxic pollution (DEHLG, 2010). These threats make reproduction, development or survival of the FWPM difficult. These are also the main threats to FWPM populations around the world (Cooksley et al., 2012, Österling et al., 2010, Ostrovsky and Popov, 2011).

Siltation can lead to the death of up to 100 percent of juvenile (under 5 years old) mussels and can cause considerable damage to adult species (Österling et al., 2010). For their survival, FWPMs need clean gravel river beds with clean below the pea-sized gravel as adult mussels are two thirds buried in the substrate and juveniles are completely buried. Once the substrate is clogged with fine sediment, oxygen movement to mussels "buried" in the gravel is prevented which leads to the death of all mussels under 5 years of age (DEHLG, 2010, Novotny, 2003, Österling et al., 2010, Arvidsson et al., 2012). Turbidity causes adult mussels to clamp up as the ingestion of fine particles can lead to rapid death. If mussels remain closed for prolonged periods of time (a few days) they may die due to a lack of oxygen. Even if the FWPM survive, the stress caused by oxygen deprivation may take some time to recover from (DEHLG, 2010). Adult FWPMs can survive despite siltation, but in order to excrete pollutant particles and mud they must continuously expend energy, which decreases their life expectancy. In addition, with strong mud contamination pregnant females eject their mussel larvae at an immature stage (NATURA, 2000). One of the objectives of the European Communities Environmental (Freshwater Pearl Mussel) Regulations, S.I. 296 of 2009 (European Communities, 2009), is to ensure that there is no artificially elevated levels of siltation present at the sites where pearl mussel populations are present.

Nutrient enrichment is another serious threat to the welfare of all aquatic organisms with some organisms more prone to damage than others. Eutrophication and silt may adversely affect mussels in a variety of ways ranging from negative effects on the salmonoid host fish to increases in BOD associated with algal overgrowth which in turn may cause problems to both juvenile and adult mussels which require constant
oxygen flow (Arvidsson et al., 2012). A shortage of oxygen may lead to stress and suffocation of the pearl mussels. Sewage from scattered settlements, pond waste waters and fertilizer losses from agriculture are cited as the main sources of nutrients in water bodies (NATURA, 2000, Merrington et al., 2003, Novotny, 2003, Ritter and Shirmohammadi, 2000).

Acidification of the water in rivers inhabited by pearl mussel populations also poses a direct threat to the FWPM by destroying the mussel's calcareous shell and the reproductive organs of adult mussels causing infertility (DEHLG, 2010). A low water pH level affects the habitat of the streams and has a negative effect on host fish indirectly affecting reproduction of the pearl mussel (Österling et al., 2010, Skinner et al., 2003).

As filtering organisms, mussels are endangered by pollutants which tend to be adsorbed and enriched in the particles; this also forms their major food supply and the substrate on which they live. The effect of toxic pollutants (such as heavy metals and pesticides) on aquatic organisms has been well established. Amongst the most dangerous pollutants in regard to FWPM are: 1) lime; 2) heavy metals and 3) pesticides (DEHLG, 2010). These toxic pollutants may have both long and short-term adverse effects on pearl mussel populations. Lime can lead to reductions in the growth of mussels and shorten their life expectancy leading to the loss of reproductive years. In some cases lime may also lead to the immediate death of pearl mussels. Hartmut and Gerstmann (2007) studied freshwater pearl mussel populations from Germany and Finland and found highly elevated levels of heavy metals in tissue of the sampled molluscs. Their results indicated that the number of persistent organic pollutants and some heavy metals may cause calcium deficiency in FWPM. Low doses of pesticides like sheep dip are reported to be lethal during the glochidial stages of the FWPM lifecycle (Bringolf et al., 2007a, Bringolf et al., 2007b, Bringolf et al., 2007c, Hartmut and Gerstmann, 2007).

Agriculture has been consistently cited as one of the sources of nutrients in water bodies (Ghosh and Sarkar, 2014, Ouyang et al., 2010, Parris, 2011). A decline in the numbers of FWPM in Ireland and other countries coincides with intensification of agricultural practices and increase in the use of phosphorus and nitrogen fertilisers as well as increases in sheep numbers and the number of cattle drinking directly from
It is thus important to alleviate the pressures that come from agricultural activities. However, this must be done in a cost effective manner.

The problems discussed above not only affect the FWPM populations but can also lead to changes to the whole habitat of watercourses. Thus, measures taken to protect FWPM are likely to lead to improved environmental conditions for a number of other flora and fauna species.

**V.3 Mitigation Measures**

As discussed in the previous section a number of threats to the sustainability of FWPM exist. The focus here is on measures aimed at reducing threats from activities associated with agricultural production. To reduce nutrient enrichment (eutrophication) from runoff and leaching that originate on agricultural land, reduction of fertiliser usage and/or reductions in the number of livestock units could be pursued. A reduction in chemical fertiliser application would result in less risk of runoff and/or leaching to the ground waters, while a reduction in the number of livestock units would lead to reductions in organic fertiliser (in the form of urea and/or manure) directly deposited during grazing and would reduce the volume of slurry accumulated during the winter and later spread on the land (NATURA, 2000; Ritter, 2001; Merrington et al., 2002; Novotny, 2003).

The most commonly cited sources of sediment in rivers and streams are construction (which is beyond the scope of this chapter) and ploughing (Merrington et al., 2002; Novotny, 2003). The soil channel erosion of tillage land as well the destabilisation of river banks causes siltation of the gravel substrate and increases the level of suspended sediment in streams. Measures that restrict the ploughing of land in proximity to streams as well restricting livestock access to the streams through fencing would deal with these problems (DEHLG, 2010).

Sub-basin Management Plans (SBMPs), which are part of the WFD river basin programme of measures, were developed in Ireland by the Department of Environment, Heritage and Local Government (DEHLG) to address the location specific issues related to FWPM in an attempt to develop favourable conditions for the sustainable reproduction of these species.
There are 96 populations of FWPM in Ireland, the 27 catchments where these populations are present fall into 19 special areas of conservation. MM is found in 26 river catchments (see Figure V.1) and MD is found only in the Nore catchment (DEHLG 2010). Thus, 27 SBMPs were developed with the aim of providing a mechanism to address the threats to the survival of FWPM on a catchment-level scale. The SBMPs develop a strategy to implement measures that will bring the catchment, and by extension the FWPM populations, back to a favourable condition. This is done by defining a programme of measures for each sub-basin. Under the WFD requirements the measures contained in these programmes had to be operational by the 22nd of December 2012 (DEHLG, 2010).

**Figure V.1. Map of the FWPM**

As part of this process, each SBMP lists potential pressures on the FWPM populations and suggests both general and specific measures to reduce these
pressures. These measures are grouped into two broad categories: measures to reduce pressure at the source and measures to remediate pressure along the pathway (DEHLG, 2010). Table V.1 lists these measures.

**Table V.1. List of measures to reduce pressure on the ecology of FWPM.**

<table>
<thead>
<tr>
<th>Measures to remediate pressures at the source:</th>
<th>Measures to remediate pressures along the pathway:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reductions/cessation of fertiliser use</td>
<td>Establishment of an appropriate, site-specific buffer zones of native woodland or semi-natural</td>
</tr>
<tr>
<td>Reductions/cessation of slurry application</td>
<td>Vegetation around drains, streams, rivers and lakes</td>
</tr>
<tr>
<td>Implementation of nutrient management plans</td>
<td>Floodplain restoration</td>
</tr>
<tr>
<td>Reductions/cessation of ploughing</td>
<td>Wetland restoration</td>
</tr>
<tr>
<td>Reductions/cessation of drainage and drainage maintenance</td>
<td>Creation of artificial wetlands or filter beds</td>
</tr>
<tr>
<td>Reductions in grazing intensity/livestock units</td>
<td>Installation of appropriately-sized sediment traps</td>
</tr>
<tr>
<td>Other reductions in land use intensity, e.g. conversion to native woodland</td>
<td>Other measures to increase infiltration or slow/divert surface run-off, or flow in drains.</td>
</tr>
<tr>
<td>No liming of land in sensitive areas</td>
<td>Reducing or eliminating extraction within the identified catchment.</td>
</tr>
<tr>
<td>Fencing off drains, streams or rivers where there is significant bed or bank erosion.</td>
<td></td>
</tr>
</tbody>
</table>

In addition to the measures listed above, emergency measures may be employed in “highly sensitive” areas. However, the definition of highly sensitive areas is unclear and the “emergency” measures are not currently specified in the SBMPs. It is proposed that agri-environmental schemes are to be developed to reward/compensate the farmers that operate in a more environmentally friendly fashion. This scheme would insure that farms within the 27 catchments would have on-farm plans that may include any of the measures listed if they are deemed necessary after an on-farm survey and assessment has been conducted (DEHLG, 2010).

To achieve the mitigation objectives, and the associated reduction in pressures on FWPM streams, would require either reductions in the level of agricultural production or increases in farms’ production efficiency or some combination of both.
This chapter estimates the costs associated with five mitigation measures cited in the SBMPs that have the potential to reduce the pressure on FWPM at source since they may mitigate problems associated with nutrient enrichment, siltation, toxic pollution etc. However, the cost of the measures needs to be known to fully inform the decision-makers. The five mitigation measures are:

- the reduction of inorganic fertiliser by 20 percent;
- the reduction of LU to achieve 170 kg of organic N per hectare\(^{11}\);
- the reduction of LU by 20 percent;
- switching from tillage to beef production and
- the fencing off of adjacent streams (and an associated de-intensification of production).

V.4 Methodology

A cost-benefit analysis would be required to fully inform the stakeholders of the economic implications of the various mitigation measures. Such analyses very often demand detailed spatial data on all the costs and benefits (both tangible and intangible) of the proposed measures. In this chapter only the costs associated with five mitigation measures are explored, as the anticipated benefits of the measures (in terms of increases in the FWPM populations) have not been quantified in the relevant scientific literature. A number of different approaches were used for calculations in this chapter. The costs for each mitigation measure were calculated in the following manner.

**Fertiliser and Livestock Reduction Scenarios**

The economic impacts of the 20 percent fertiliser reduction scenario and of the reductions in the number of livestock units were estimated using microsimulation methods similar to those described in chapter IV of this thesis. Such microsimulation techniques are useful in conducting counterfactual analyses such as the outcome of

\(^{11}\) As defined by SI 378 of 2006 the maximum allowable production of organic N on the farms is 170 kg of N per hectare. Farmers at the moment can apply for derogation up to 250 kg N per hectare – this scenario assumes that the limit of 170 kg of N per hectare becomes binding. http://www.environ.ie/en/Legislation/Environment/Water/FileDownload,1573,en.doc
different policy scenarios at a household, firm/farm or other micro-unit levels. Microsimulation approaches have been used extensively to evaluate taxation and pension systems (Merz, 1993, Mitton et al., 2000, Spadaro, 2007) and are increasingly being applied in other contexts (O'Donoghue et al., 2012). The application of simulation approaches within an environmental context is growing. For instance, Hennessy et al. (2005) use a linear programming approach to simulate the effects of complying with a limit of 170 kg of organic N per hectare on farm income.

In this chapter an econometric technique (described in detail in chapter IV of this thesis) is used to simulate the impact of the five alternative policy measures on individual farm profits ($\pi_i$). While the model in chapter IV is applied only to dairy farms in Ireland; the current chapter applies the model to all farm systems represented in the NFS data. Farms in Ireland are generally engaged in multiple enterprises. In chapter IV only 3 enterprises are considered, while in the current context four - dairy, beef, sheep and crops enterprises – are considered:

Similarly to chapter IV, farm profit ($\pi_i$) is defined as the value of gross output ($GO_i$) less direct costs ($DC_i$) and fixed costs ($FC_i$) (equation V.1), where $i$ denotes an individual farm. However, since FC are not affected by the changes in GO and DC that are simulated within the model, changes in $\pi_i$ are driven by changes in only $GO_i$ and $DC_i$ i.e. by changes in Gross Margin ($GM_i = GO_i - DC_i$) (equation V.2). In Ireland many farms engage in more than one farm enterprise so to reflect this this model operates at the farm enterprise level (dairy, sheep, beef and crops enterprises) rather than at farm level. Thus, unlike in chapter IV, eight instead of six functions are estimated in this chapter. Each enterprise provides different gross margins per unit of output. Individual enterprises also have different gross output and direct cost functions. Thus, GO and DC are simulated for each enterprise separately (equations V.3 and V.4) and the results are then aggregated together to give a farm level total for GO and DC. A subscript $j$, is used to denote a particular enterprise.

Thus, the micro simulation model can be represented by:

$$\pi_i = GO_i - DC_i - FC_i$$  \hspace{1cm} (V.1)

$$GM_i = GO_i - DC_i$$  \hspace{1cm} (V.2)
Where $X_{ij}$ is a vector of explanatory variables; e.g. livestock units, farm size, fertiliser usage and concentrates$^{12}$. $X_{ij}$ determine the level of each enterprise $GO_j$ and $DC_j$, where $j$ denotes dairy, cattle, sheep or crop enterprise on the farm.

The simulation procedure is the same as described in chapter IV and is carried out by holding the regression coefficients ($\beta$, $\gamma$) and the error terms ($\epsilon_{ij}^{GO}$, $\epsilon_{ij}^{DC}$) constant and changing the explanatory variables ($X$) to reflect the introduction of a particular measure. This involves changes in the amount of fertiliser used or the number of livestock units in the model. When the parameters of the model are estimated the new production and costs are simulated (denoted as $GO^\circ$ and $DC^\circ$ in equations V.5 and V.6).

$$GO_{ij}^\circ = (X_{ij}^\circ \beta_j, \epsilon_{ij}^{GO})$$ \hspace{1cm} (V.5)

$$DC_{ij}^\circ = (X_{ij}^\circ \gamma_j, \epsilon_{ij}^{DC})$$ \hspace{1cm} (V.6)

The simulated impact of the measure is the difference between farm profit before ($\pi$) and after the change ($\pi^\circ$) (equations V.7 and V.8).

$$\pi^\circ_{ij} = GO_{ij}^\circ - DC_{ij}^\circ$$ \hspace{1cm} (V.7)

$$\Delta \pi_{ij} = \pi^\circ_{ij} - \pi_{ij}$$ \hspace{1cm} (V.8)

This is the approach taken for the first three measures considered in this chapter: 1) the reduction of fertiliser by 20 percent; 2) the reduction of livestock units (LU) to achieve 170 kg of organic N per hectare; 3) a 20 percent LU reduction. For each measure the livestock numbers or fertiliser usage were adjusted according to the scenario considered and the change in profits was simulated.

$^{12}$ The models are estimated using a log-polynomial functional form using ordinary least squares (OLS) regression.
SWITCHING FROM TILLAGE TO BEEF PRODUCTION

It was assumed that if farmers were not allowed to plough in the FWPM catchments they would switch from tillage to beef production. The farms are divided into three groups: 1. the farms that do not engage in the crop production; 2. the farms that have crop enterprises and have at least 20 percent of land designated to beef production; 3. crop farms that do not have beef enterprise. The first type of farms is unaffected by the measure.

It is assumed that after switching from tillage to beef production the second type of farms will derive the same gross output per hectare and will incur the same direct costs per hectare as their existing beef enterprise. It is assumed that the third type of farms will derive the same gross output per hectare and will incur the same direct costs per hectare as an average beef enterprise in the same region. The effect of the measure is the change in the farm profit (equation V.8). This measure does not take into consideration the initial investment that may be required for extending the beef enterprise (e.g. buying more cattle, investing in buildings etc.).

SPATIAL MICROSIMULATION

Although the NFS dataset is nationally representative, it cannot be used to estimate aggregate effects at a local level since it is not spatially representative, nor would the sample size permit detailed local analysis even if it were spatially representative (see Green and O'Donoghue (2013)). In order to estimate the effects of each measure on all farms located in the FWPM catchment, a spatially representative dataset is required. One potential source is to simulate the data utilising a spatial microsimulation approach (O’Donoghue et al., 2014c). Hynes et al. (2006) outline three main benefits of using synthetic data: the ability to create micro data from aggregated macro data at different spatial resolution; the ability to retain a number of characteristics of micro-units within the data and facilitate a multivariate analysis and the ability to assess the impact of different policies on particular groups within the population within spatial unit.

The SMILE-FARM model simulates spatially representative households and farms at an electoral district (ED) level using a number of data sets: the NFS, the Census of Population and the Census of Agriculture (COA) amongst others (O’Donoghue et al., 2014c).
The data simulation process involves the sampling of farms from the micro dataset containing detailed farm level data from the Teagasc NFS to make it consistent with the COA. The constraint variables used include, farm size, farm system, soil type and stocking rate; variables that are strongly associated with farm level outcomes in Ireland.

SMILE was developed as a policy simulation tool to evaluate and provide evidence in relation to the impact of public policies that have a spatial dimension, particularly in relation to policies that affect rural populations. The model comprises two linked components: a household model and a farm enterprise model. The household model contains a database of households in each of the 3400+ electoral districts of the country with detailed data on individuals within households and their respective demographic characteristics, labour market, income and expenditures. The farm level model that is linked to the farm households in the household model contains 140k farms each located within their district and containing all of the land use, output, subsidy, direct cost and indirect cost variables at the individual enterprise level, amounting to 1,500 variables per farm. The objective of the model is to assess how the types of policies introduced by the government may affect households and farm enterprises within the spatial locations of where they live and work and to be able to perform disaggregated analyses by the characteristics of these units such as age, income category, employment status, farm type etc. (O'Donoghue et al., 2012).

The SMILE-FARM model was developed by Hynes et al. (2008b), Hynes et al. (2006), Hynes et al. (2009b), Hynes et al. (2009a) who used a Simulated Annealing (SA) method to enhance the household SMILE model with a view to examining the impact of EU Common Agricultural Policy in Ireland. They used NFS data for the year 2008 and the COA for the year 2000 to develop a data set that would represent the population of farms in Ireland. Although their methods proved to be robust, its limitation was that SA took weeks or even months to run. There were also some challenges in relation to the spatial representativeness of stocking rates, which is very important given the importance of animal based agricultural systems in Ireland. O’Donoghue et al. (2014b), O’Donoghue et al. (2014c) have further improved the methodology of SMILE-FARM by trying 16 different methods of simulation on NFS 2010 and COA 2010 and data validation. They utilised a new method for data
generation, known as Quota Sampling (QS) which was found to be the most efficient approach to simulate the data with a minor loss of convergence but large gains in time-efficiency, which took only hours to run (for further discussion see O’Donoghue et al., 2014b, 2014c).

SA and QS methods are very similar, they select observations at random and consider whether these observations are suitable for selection for a given spatial unit based on conformance with aggregate totals for this spatial unit. Unlike SA, QS only assigns units (in this case farms) that conform to aggregate constraint totals and once a unit is selected, it is not replaced; which is the main reason for computational improvement (O’Donoghue et al., 2014, Hynes et al., 2006).

The spatial unit aggregate totals for each constraining variable are required as to determine ‘quotas’, or running totals for each constrained variable, which are recalculated once a unit is admitted to a small area population. The method randomly sorts the population of farms and allocates one unit at a time, subject to a number of constraints. If the unit sum of each constraining characteristic (e.g. a Dairy Specialist Farm) is less than or equal to each small area total (e.g. 10 Dairy Specialist Farms in the small area), the unit is assigned to the small area population. Once a unit is selected for a given small area, quota counts are amended, reduced by the sum of the characteristics of the assigned unit(s). This procedure continues until the total number of simulated units is equal to the small area population aggregates (i.e. all quotas have been filled) (O’Donoghue et al., 2014).

This mechanism of sampling without replacement avoids the repeated sampling procedure of SA and is fundamental to the efficiency gains of the quota sampling procedure relative to other methods. However, this method of improving efficiency does present a number of convergence issues. Disparities in population distributions between census and survey totals may create a number of problems for unit-based microsimulation procedures. This is because survey micro-data are representative at the national level, whereas small area census data are representative at the district level. This poses little difficulty in simulating small areas that have a population distribution similar to that of the national distribution, but areas that differ from the national distribution may lead to some demographic groups consistently being
underrepresented in a given district. These differences may cause some districts to consistently fail in reaching adequate convergence (O'Donoghue et al., 2014).

Also, the use of sampling without replacement in quota sampling results in quota counts becoming increasingly more restrictive as the simulation progresses. As quota counts reach their target, the search space is continuously refined in accordance with concurrent quotas, whereby all units no longer eligible given updated quota totals are removed from the subset and the procedure is repeated. When each constraint allocation reaches its target quota, all individuals of that characteristic are removed from the candidate search space. These mechanisms cumulate to offer a continuously diminishing search space and may prohibit convergence, whereby no unit is able to satisfy all concurrent quota counts (O'Donoghue et al., 2014).

The SMILE has been applied in a number of contexts including for rural healthcare services evaluation (Morrissey et al., 2008), travel cost modelling (Cullinan et al., 2008, Cullinan, 2011) and proved to be a very useful tool in evaluating agricultural policy analysis. For example, Hynes et al. (2009b) used SMILE-FARM to evaluate methane emissions on Irish farms. In another study by Hynes et al. (2008a) this model was used to estimate the probability of Irish farmers to participate in REPS.

The SMILE model is used in this chapter to create a spatial microsimulation model and enhance the micro-simulation model described in chapter IV. There are a number of advantages in using spatial microsimulation models over the traditional macro-models that include the potential to link different data sets that contain just one attribute in common with flexibility in spatial resolution of the study that can vary in scale. Moreover, such models store data more efficiently and can be updated with new information becoming available (Hynes et al., 2006). The disadvantages include the difficulty in results validation (Hynes et al., 2006) and the cost of updating the models (O'Donoghue et al., 2012).

The changes in the farm practices and incomes are simulated at a farm level and the over-all results are at a catchment scale. As a result of merging the two models the final model allowed the aggregation of the simulated results for the farms contained within the FWPM catchment to obtain a total cost of implementing a particular
measure in the catchment area and, thus, to assess the impacts of the mitigation measures at a catchment scale.

**FENCING OFF STREAMS**

The cost of the fencing off measure cannot be estimated in the same manner as the other four measures due to data limitations. Since the NFS does not contain spatial information it is not possible to say which farms would have streams passing through them. There are two costs associated with introducing this measure – 1) the cost of erecting the fence on both sides of the streams in the pearl mussel catchments and 2) the opportunity cost of the agricultural land taken out of production in the buffer zone fenced off. The cost of erecting the fence is calculated on the basis of €0.9 per meter of the fence (equation V.9),

\[ C_{\text{fence}} = m \times 0.9 \times 2 \]  

(V.9)

where \( m \) is the total length of streams in the pearl mussel catchments. The opportunity cost of the land fenced off is calculated by multiplying the average farm gross margin per hectare \( (GM_{\text{av/ha}}) \) in the FWPM catchment by the total area of agricultural land fenced off in the catchment \( (n_{\text{ha}}) \) (equation V.10).

\[ GM_{\text{loss}} = n_{\text{ha}} \times GM_{\text{av/ha}} \]  

(V.10)

**THE COST PER UNIT OF NITROGEN**

In order to make the economic costs of the three simulated (livestock and fertiliser reduction) measures comparable, a cost per unit of Nitrogen abated due to the measures is calculated. To do so first, the total amount of organic and chemical N per hectare that is produced on the farms in the status quo scenario is calculated, then, assuming one of the three policy measures is introduced the new level of N is calculated. The level of organic N used in enterprise \( j \) is calculated by multiplying the number of LU of type \( k \) on a farm by the annual N excretion rate of that LU type \( (E_k) \) and summing across the \( k \) types of LU, which is then added to the level of Inorganic
N for enterprise type to give a total N for each enterprise type. The total N for farm \( i \) is obtained by summing over each of the enterprises (equation V.11).

\[
N_i = \sum_j \left( \sum_k \left( E_i^{NLU_{kj}} \right) \right) + \text{Inorganic } N_i
\]  

(V.11)

The change in N is then calculated for each of the measures. The cost per unit of N abated (CPUA) is then calculated by dividing the total cost by the change in N to give a cost per unit of N abated (equation V.12).

\[
\text{CPUA} = \frac{\Delta \pi_i}{\Delta N_i}
\]

(V.12)

**V.5 Data**

Since the impact of the measures, if introduced, would be very localised, the estimated costs should apply only to the impacted areas. To isolate the affected farms, spatially referenced farm level data is required which is representative at the local level. Since such rich data is not available, instead farm level data is utilised to estimate costs and then a spatial micro-simulation approach is used to determine the costs at a local level. Data from the Teagasc NFS data is used to estimate the effect of each of the five FWPM protection measures on farm gross output and direct costs. In order to estimate the effects of each measure on all farms located in the FWPM catchment, a spatial microsimulated dataset – produced using the Spatial Microsimulation of the Irish Local Economy (SMILE) model was used to isolate farms in the catchment area (O'Donoghue et al., 2013). Information on the catchments is taken from the SBMP reports (DEHLG, 2010).

**NFS DATA AND THE CATCHMENTS’ STATISTICS**

As has been described in the previous chapters, the NFS data is collected by surveying a sample of farms in Ireland. The survey has been conducted by Teagasc on an annual basis since 1972. It contains detailed socio-economic information about approximately 1,200 farms each year that represent over 100,000 farms in Ireland (Connolly et al., 2010). On the basis of statistical information from the Census of Agriculture which is reported by the Central Statistics Office (CSO), Teagasc
develops national and regional weights every four years to ensure the representativeness of results obtained using the NFS.

The SBMPs report the length of the rivers in catchments where populations of the FWPM are present (DEHLG, 2010). Out of the 27 catchments in Ireland where FWPM can be found only nine are considered to be under ecological pressure from agricultural activities: Munster Blackwater, Nore and Leannan, Dereen, Mountain Aughnabrisky, Ballymurphy, Clodlagh, Bandon and Caha and finally Cloon.

V.6 Results

The results of the analysis are presented and discussed in this section. Table V.2 presents summary statistics on the catchments. These statistics show that the catchments under consideration not only differ in size but in the average GM per hectare that farmers earn in different catchments. While in Blackwater, Brandan and Caha and average GM/ha is about €1000, in Nore it is about €850. Blackwater catchment falls into dairying agricultural area, Nore falls into livestock and arable region (please, see map in Appendix 1).

<table>
<thead>
<tr>
<th>Catchment Name</th>
<th>Size (ha)</th>
<th>Length of river (km)</th>
<th>Proportion of catchment under agri use</th>
<th>Land in agri production (ha)</th>
<th>Average GM** € per ha</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blackwater</td>
<td>233,300</td>
<td>2240</td>
<td>0.81</td>
<td>189,155</td>
<td>1,003</td>
</tr>
<tr>
<td>Nore</td>
<td>105,890</td>
<td>970</td>
<td>0.80</td>
<td>84,985</td>
<td>853</td>
</tr>
<tr>
<td>Leannan</td>
<td>23,765</td>
<td>460</td>
<td>0.70</td>
<td>16,752</td>
<td>872</td>
</tr>
<tr>
<td>Dereen</td>
<td>20,118</td>
<td>253</td>
<td>0.78</td>
<td>15,714</td>
<td>947</td>
</tr>
<tr>
<td>Mount Aughnabrisky*</td>
<td>10,316</td>
<td>130</td>
<td>0.78</td>
<td>8,058</td>
<td>975</td>
</tr>
<tr>
<td>Ballymurphy*</td>
<td>3,242</td>
<td>41</td>
<td>0.78</td>
<td>2,532</td>
<td>975</td>
</tr>
<tr>
<td>Clodiagh*</td>
<td>12,303</td>
<td>154</td>
<td>0.78</td>
<td>9,610</td>
<td>975</td>
</tr>
<tr>
<td>Bandan &amp; Caha*</td>
<td>15,821</td>
<td>199</td>
<td>0.78</td>
<td>12,358</td>
<td>1,003</td>
</tr>
<tr>
<td>Cloon*</td>
<td>5,900</td>
<td>74</td>
<td>0.78</td>
<td>4,608</td>
<td>891</td>
</tr>
</tbody>
</table>

*Only the total length of the streams for these catchments was available, the length was calculated as a proportion of the catchment size. **GM (farm gross margin is calculated as a difference between gross output and direct costs)

The Blackwater and Nore are the two largest catchments with significant agricultural activities. The NFS survey for 2008 was used to calculate the average gross margins (GMs) reported in Table V.2. The Gross Margin figures for each catchment were
calculated based on the average farm Gross Margin in the relevant NUTSIII region weighted to represent the full population of farms at a NUTSIII level\textsuperscript{14}.

The results of the analysis confirm that measures proposed in the SBMPs to reduce pressure on the freshwater pearl mussel ecology would lead to a significant reduction in agricultural production and a loss of income for farmers in the affected catchments.

**THE COST OF FERTILISER AND LIVESTOCK REDUCTION**

The total impact of each measure and the percentage change in GM are reported in Table V.3. The measure that leads to the largest negative impact in absolute terms would be a livestock reduction by 20 percent measure. This measure would result in the loss of estimated €48 million per year for farmers located in the FWPM catchments. This represents a decrease in the total GM for all farms in the catchments of over 12 percent. A reduction in fertiliser application of 20 percent would lead to a total estimated loss of over €5 million per year.

<table>
<thead>
<tr>
<th>Measure</th>
<th>Total impact, € ’000</th>
<th>Total Impact, percent GM lost</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fertiliser reduction by 20 percent</td>
<td>-5,252</td>
<td>-1.32</td>
</tr>
<tr>
<td>LU reduction down to 170gk N/ha</td>
<td>-2,690</td>
<td>-0.68</td>
</tr>
<tr>
<td>LU reduction by 20 percent</td>
<td>-48,000</td>
<td>-12.07</td>
</tr>
</tbody>
</table>

The total costs of the measures differ for these measures. This is due to the difference in the impact that individual measures would have on individual farms and on farming in the pearl mussel catchments. If all the farms in the FWPM catchments were to comply with the organic N limit of 170 kg of organic N per hectare they would incur an estimated total cost of €2,690K per year. This represents a total loss of GM of approximately 0.68 percent per year. The reason for the lower cost for this measure is that this measure would affect only 7 percent of the farms in the FWPM catchments – 6.5 percent would experience a loss of beef enterprise gross margin and 0.5 percent would benefit from the measure since they currently have cattle that

\textsuperscript{14} The Nomenclature of Territorial Units for Statistics (NUTS) classification has three levels: NUTS1 covers the whole Ireland; NUTSII divides Ireland into Boarder Midlands, Eastern Region (BMW) and Southern, Eastern (SE) region; NUTSIII divides Ireland into eight regions: West, Boarder, Midlands, Mid-East, Dublin, Mid-West, South-East, and South- West IRO. 2014. \textit{The Irish Regions Office} [Online]. Available: http://www.iro.ie/irish_regions.html.
attract a negative gross margin and the assumption is that these cattle would not be maintained under the measure.

**THE COST OF SWITCHING FROM TILLAGE TO BEEF PRODUCTION.**

The analysis shows that if the farmers in the FWPM catchments were to transfer the land that they are using for tillage production to beef production, there would be a total loss of €1,738K per year for all affected farms. This amount does not include the initial investment that may be required to switch from one system to another which may be considerable. The farms with some crop production that have over 20 percent of their land designated to beef production the loss would be €273K and on farms that have less than 20 percent of land designated to beef production (predominantly tillage farms) the loss is estimated to be €1,471K per year for all affected farms.

**THE COST OF FENCING OFF STREAMS**

The cost of the fencing off measure will be comprised of (a) the costs of erecting the fence around the streams and (b) the costs associated with the agricultural land in the buffer zone taken out of production. The cost of fencing is assumed to be 0.9 Euros per meter. The total cost of fencing all riverbanks in the catchments in this study comes to €8,136K. If the fence is assumed to last approximately 5 years and maintenance costs are not included, the annual cost of erecting fencing is €1,627K.

The introduction of buffer strips between the fence and waterway is often a necessary measure. A 10m buffer strip is normally sufficient where the slope of the land does not exceed 10 percent and where no major polluting activities take place. However wider buffers may be needed where the slope is steeper than 10 percent and where there is poor soil quality. The actual buffer width would need to be decided on a case by case basis. However to give an indication of the associated costs (i.e. the GM foregone) three buffer widths (10m, 25m and 50m) are considered. Table V.4 presents the costs associated with fencing and with the buffer zones for each of the catchments.

Fencing streams off is one of the most cited measures in relation to the streams protection (Collins et al., 2007, McDowell, 2008, Bryan and Kandulu, 2009, McDowell and Nash, 2012) and the FWPM protection in particular (DEHLG, 2010).
It seems to be one of the most straight-forward ways to mitigate pollution to water streams, especially from animal production. However, if introduced, this measure would not only lead to an expense in the form of erecting a fence, but to a loss of income to farms located in the areas.

The total GM lost across all catchments as a result of the fencing off of 10m buffer strips would amount to almost €7 million per year. If 25m was fenced off to further reduce the ecological pressure on FWPM populations, the gross margin loss would amount to almost €17 million per year and almost €34 million per year would be lost by farmers if 50m buffers were required. Thus the total costs of erecting fencing and creating buffer zones is between almost €8.4 and €35.6 million per year. With environmental benefits of such measure un-quantified at the moment, the large cost associated with this measure calls into question its implementation due to the disproportionally high cost. Moreover, the spatial analysis highlights the heterogeneity in the impacts that this measure would have on farms in different catchments. If the farms in Blackwater are expected to lose 2% of the GM, then in Leannan this number is double of that, thus, again stressing the need to consider spatial heterogeneity while formulating mitigation policies.

**Table V.4. Gross Margin lost due to fencing off land used in agriculture**

<table>
<thead>
<tr>
<th>Catchment Name</th>
<th>Length of streams (km)</th>
<th>Fencing Costs €’000</th>
<th>GM lost in 10m buffer €’000</th>
<th>GM lost in 25m buffer €’000</th>
<th>GM lost in 50m buffer €’000</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blackwater</td>
<td>2,240</td>
<td>806</td>
<td>3,640 2</td>
<td>9,099 5</td>
<td>18,198 10</td>
</tr>
<tr>
<td>Nore</td>
<td>970</td>
<td>349</td>
<td>1,324 2</td>
<td>3,310 5</td>
<td>6,619 9</td>
</tr>
<tr>
<td>Leannan</td>
<td>460</td>
<td>166</td>
<td>562 4</td>
<td>1,405 10</td>
<td>2,809 19</td>
</tr>
<tr>
<td>Derreen</td>
<td>252.6</td>
<td>91</td>
<td>374 3</td>
<td>933 6</td>
<td>1,867 13</td>
</tr>
<tr>
<td>Mount Aughnabriskiy</td>
<td>129.5</td>
<td>47</td>
<td>197 3</td>
<td>492 6</td>
<td>985 13</td>
</tr>
<tr>
<td>Ballymurphy</td>
<td>40.7</td>
<td>15</td>
<td>62 3</td>
<td>156 6</td>
<td>310 13</td>
</tr>
<tr>
<td>Clodiagh</td>
<td>154.5</td>
<td>56</td>
<td>235 3</td>
<td>588 6</td>
<td>1,175 13</td>
</tr>
<tr>
<td>Bandan &amp; Caha</td>
<td>198.6</td>
<td>72</td>
<td>311 3</td>
<td>777 6</td>
<td>1,554 13</td>
</tr>
<tr>
<td>Cloon</td>
<td>74.1</td>
<td>27</td>
<td>103 3</td>
<td>258 6</td>
<td>515 13</td>
</tr>
<tr>
<td>Total</td>
<td>4,520</td>
<td>1,627</td>
<td>6,807 2</td>
<td>17,017 5</td>
<td>34,033 10</td>
</tr>
</tbody>
</table>

*percentage of a total GM in each pearl mussel catchment

**THE CPUA RESULTS**

In order to compare the relative cost of the measures, the CPUAs were calculated for measures 1-3. The results, reported in Table V.5, suggest that while the cost per unit
of N abated are similar for the measures reducing fertiliser application and for reducing LU to reach a level of 170kg of N per hectare, the costs associated with an across the board reduction in LU of 20 percent are prohibitive at over six times the magnitude of the preceding measures.

Table V.5. CPUA results

<table>
<thead>
<tr>
<th>Measure</th>
<th>CPUA, €/N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fertiliser reduction by 20 percent</td>
<td>0.41</td>
</tr>
<tr>
<td>LU reduction down to 170kg N/ha</td>
<td>0.24</td>
</tr>
<tr>
<td>LU reduction by 20 percent</td>
<td>4.85</td>
</tr>
</tbody>
</table>

The analysis revealed that switching from the tillage to beef production, a measure that aims to reduce the loss of sediment due to ploughing, leads to an increase in the application of N per hectare on 86 percent of the farms in the catchments. Thus, this measure can potentially reduce the loss of sediment in the FWPM catchments but at the expense of increasing nutrient loss from the same land. Thus, the relative risks associated with nutrient loss and sediment loss must be carefully assessed before seeking to implement this measure.

V.7 Conclusion and Discussion

Spatial heterogeneity has long been recognised as an important element in policy assessment and formulation. This fact is reflected by a large number of the spatial microsimulation models developed and applied in different areas of economic analysis. Originally formulated for assessing the impact of the taxation on the population income, it is now used a variety of analysis like healthcare, environment, transport and other areas of economic analysis (O'Donoghue et al., 2012).

In this chapter the spatial microsimulation model is utilised in conducting the case study if the cost of the FWPM protection. In most countries within the EU, the FWPM is either almost extinct or exists only in small senescent populations and may become extinct before future generations arrive if a major hydro-ecological recovery in the catchments where they are present is not achieved (Araujo and Ramos, 2001; Geist, 2006; DEHLG, 2010). Thus, there is an increasing demand to reduce ecological pressures on pearl mussel habitats in order to ensure the survival and reproduction of these critically endangered species.
In line with the requirements of the Water Framework Directive (Directive 2000/60/EC), SBMPs for 27 river catchments in Ireland were developed (DEHGL, 2010). In these management plans a list of possible measures to reduce environmental pressures on the FWPM at their source and along the pathways are cited. Measures that aim to reduce the pressures at their source would lead to a reduction in agricultural output due to a reduction in stocking densities, in livestock numbers or through a reduction in fertiliser usage. In this chapter the likely impact of such measures is explored using microsimulation techniques applied to the Teagasc NFS data and a synthetic spatial microsimulated population using SMILE.

The results of this analysis suggest that reducing pressures on the FWPM at their source (as specified in the SBMPs) will come at a considerable cost to the agricultural sector in these regions. Moreover, the analysis reveals that some mitigation measures may potentially lead to increasing pressure in another environmental dimension if the farmers were to switch to different systems of production in response to a policy measure. This leads to the conclusion that the current plan to mitigate the pressure on the FWPM may lead to disproportionally high costs to the farming communities with the benefits of the measure somewhat ambiguous. Finally, the analysis highlighted the heterogeneity of impact that mitigation measure would have in different catchments.

However, these results should be interpreted with caution. The analysis is static and does not take into account the possibility that farmers alter their behaviour nor is the analysis a cost-benefit analysis since it does not look at the holistic costs nor seek to quantify or value the potential benefits of the measures. The list of the measures estimated in this chapter does not cover all the measures listed in the SBMPs. Future work in this area could model the economic impact of the other measures in order to provide full information for policy-makers. A full analysis is currently not possible due to data limitations. Data availability is one of the main obstacles for conducting research of this kind, especially in situations where the impacts of the measures are to be localised and spatial data is needed. Simulation models like SMILE help to overcome data limitations in some instances, however, even these types of models still do not contain all the spatial data needed for comprehensive economic-ecological assessment. Despite the limitations outlined, this chapter offers the insight into the
magnitude of the potential costs that agriculture may need to face in order to protect the FWPM populations in Ireland.

The overall conclusion of the chapter is, thus, that protective measures as currently envisioned may lead to significant costs on the agriculture sector, while alternative measures which focus on increasing farm efficiencies possibly via the introduction of new technologies may be a superior means to protect the FWPM. Such an approach would also be consistent the objectives of Agenda 2020, which envisions an increase of agricultural output to meet food demand of growing population, while the current measures are inconsistent with this objective.

The policy recommendations as outlined in the SBMPs will lead to significant costs to the agricultural sector and the wider economy. However, there is a paucity of robust scientific literature to indicate what measures are necessary and sufficient to preserve the existing populations of the FWPM in Ireland and to reinstate a successful reproduction process. Until such information is available it is not possible to conduct a cost benefit analysis on which to base policy recommendations. Thus, further research will be dependent on the advances in the biological research on ecology of the FWPM. However, in the times of the limited economic resources, the results of this study inform about the differences that the protective measure will have and, thus, may inform policy decisions if those could be isolated to specific catchments to minimise the losses.
Appendix 1. Agricultural Regions In Ireland

Chapter VI.

THESIS CONCLUSIONS

VI.1 General Conclusions

The fast growing population on planet Earth, urbanisation and industrialisation put an increasing pressure on the environment with pollution from human activities increasing and threatening the wellbeing of wildlife and future generations. There are a number of natural resources that have deteriorated as a result of human activities and water is one of them. The negative impact of such activities has reached a level that cannot be ignored anymore and many water bodies are already lost for beneficial human use and as a habitat for flora and fauna. Considering the fact that all known forms of life depend on water for survival, these developments created a rationale for government intervention and protection of natural resources and water resources in particular. In order to mitigate this pollution, a number of regulations were put in place; and water quality is now protected under a number of EU Directives, namely the Urban Waste Water Treatment Directive (Council Directive 91/271/EEC), Drinking Water Directive (Council Directive 80/778/EEC), Nitrates Directive (Council Directive 91/676/EEC) and Water Framework Directive (Directive 2000/60/EC). The most important legislative document, in terms of its comprehensiveness, is the Water Framework Directive (WFD), which demands all water bodies to be brought to “good status” by 2015, with countries penalised for non-compliance. It is also the first document of its kind to recognise the need to take an integrated and holistic approach towards water resources protection where cross-border cooperation is recognised as an essential element.

Despite the fact that water is arguably one of the most important socio-economic resources, until the 1970’s there has been little effort to protect it (Kissling-Näf and Kuks, 2004). This signals that market mechanisms fail to internalise the cost of the damages caused by pollution and indicates the need to balance this through mechanisms for pollution control. A clean environment is also a public good. Thus, there is a strong rationale for government intervention.

There are many sources of pollution in water bodies. They can be broadly divided into two groups – point sources and diffuse sources. Due to the fact that the origins of
the pollution can be identified with some certainty, the pollution from the point sources is easier to control once ambient or effluent (emission) standards are established. The pollution from the diffuse sources is harder to control and regulate as by their nature establishing the direct link between the particular human activities and impaired water quality at a specific point presents a considerable challenge.

Since the introduction of the WFD diffuse pollution sources have been the focus of researchers and policy-makers. The most discussed sources of diffuse water pollution are construction, street runoff, septic tanks and agriculture. With the population growth these activities intensified and also extensified. All of these sources are known to have the potential to emit undesirable substances that can end up in the water bodies and have harmful effects on the aquatic ecology. Agriculture is an area of a special concern as feeding the growing population demands increasing agricultural production. In Ireland the targeted increase in agricultural production is 30 percent by 2020 (Food Harvest 2020). The growth of agricultural output can be even higher with the lifting of the US embargo on beef imports from the EU and full access being granted to Irish beef. Recently the added “duty” of agriculture includes the production of raw material for the biofuel sector. At the same time, there is pressure on agriculture to reduce losses of potential pollutants to water. The potential pollutants from agriculture include nutrients (nitrogen and phosphorus), pesticides, sediment and pathogens. In high quantities these substances not only cause deterioration of water bodies but can also pose a threat to human wellbeing. The regulation of pollution from agriculture poses a number of challenges: first, it is hard to establish the direct link between activities on farms and their effect on the water bodies in the adjacent area; second, farmers in Ireland operate on small margins and reduction in agricultural activities may lead to a loss of these margins making farming either economically unviable or forcing food prices up affecting people living on the margin of the poverty line. Thus, careful consideration has to be given to the topic of agricultural pollution mitigation strategies.

In the face of limited economic and environmental resources there is a growing need for more efficient and sustainable management of resources and water is one of the most important of them. The WFD stipulates that the costs have to be assigned on the polluter pays principle and the pollution mitigation targets should be achieved in the
most cost efficient way. This requires knowledge about the sources of pollution, the cost of the damage and the cost of possible mitigation measures. There is still a gap in the literature (environmental and economic) and more research is needed to close the information gap in this area. Thus, research conducted in this thesis presents a contribution relevant in contemporary context of environmental economics and agro-environmental regulations. This thesis contributes to existing research by developing a modelling framework useful in assessing the possible impact of water pollution mitigation measures for Irish farmers. This model can also be used as a platform for assessment of other environmental measures, for example, GHGs emissions mitigation measures from Irish farms.

VI.2 Research Outcomes and Contributions.

The analysis undertaken in this thesis resulted in a number of outcomes and contributions to existing research, which are outlined for each chapter separately. In Chapter III of this thesis an exploratory data analysis considering the spatial statistical relationship between human activities (including agricultural and non-agricultural activities) and water quality in Ireland was undertaken. This analysis is designed to improve our understanding about the potential sources of pollution and the impact they can have on water in Irish river systems. This work extends earlier analyses conducted in Ireland by Donohue et al. (2006) who carried out a multivariate analysis of the relationship between ecological status of rivers and human activities, urbanisation and other catchment characteristics; and Curtis and Morgenroth (2013) who used linear and non-linear regression analysis to explore the effect of human activities and population densities on water quality in Irish lakes. Extending this research, the model developed in chapter III uses a different methodological approach that is based on spatially referenced data combined with an ordered probit statistical model and explores the relationship between agricultural and non-agricultural variables and water quality outcomes in river systems in Ireland.

As a result of this analysis it has been established that septic tank density, the level of organic N per hectare and the proportion of land used for the growing of cereals are negatively associated with water quality, while the degree of forest cover is positively associated with water quality. Of particular significance is the finding that including environmental variables into the model and other spatially varying characteristics did
not reduce the statistical significance of the impact that septic tank density and agricultural activities have on water quality. This refutes the idea that previous models of this kind produced the significant results only due to the failure to control for spatial and environmental variability.

The results of the analysis in chapter III confirm the need to mitigate the pollution (and N pollution in particular) from agricultural activities.

Chapter IV introduces a microsimulation model that helps to assess the impacts of N mitigation measures on farm income in Ireland. The model is developed on the basis of NFS data and the results are validated by comparing the outcomes with two mitigation measures that have previously been studied by other researchers using different modelling frameworks. These mitigation measures are: 1) a stocking rate reduction to achieve a maximum level of organic nitrogen of 170 kg/ha; 2) a 20 percent stoking rate reduction. The model thus contributes to existing research by addressing the current uncertainty regarding the economic costs to farms in Ireland of the mitigation measures needed to comply with the Nitrates Directive and WFD. The results of the model are consistent with those of Fezzi et al. (2008) and Hennessy et al. (2005) and confirm that the measures would lead to a reduction in farm gross margins if introduced.

Within the modelling framework developed in chapter IV, it is also possible to calculate farm and aggregate level CPUAs to conduct a cost-efficiency assessment of the mitigation measures. These measures include: 1) inorganic fertiliser reduction by 20 percent; 2) LU reduction to achieve N 170kg per hectare; 3) 20 percent LU reduction; 4) change in feed mix; 5) fencing off adjacent streams; 6) higher yield dairy cows; 7) efficient slurry application. All these measures have the potential to reduce N losses (keeping other factors constant) from agricultural land. The analysis also revealed that in some instances a double-dividend may be achieved when reduction of N positively affects a farm’s output (e.g., more efficient slurry application and utilising higher yield dairy cow breeds). Thus, pollution is mitigated while efficiency is increased. The measures considered in chapter IV are the most straightforward methods of N reduction. However, the list of measure assessed is not exhaustive.
Conducting a relative cost-efficiency assessment by calculating CPUAs for individual measures at an aggregate and farm level has a number of applications. At an aggregate level such an analysis offers additional information to policy-makers for efficient environmental policy design. At an individual farm level this could further be developed into a decision-making tool for farmers enabling them to compare the costs of mitigation measures on their farm. Results from this study indicate that on average farm GM per hectare declines under all strategies except in the higher yield dairy cows and slurry scenarios, which allow efficiency gains to be achieved. The CPUAs for these scenarios have negative signs which means that these mitigation measures on average would not only produce a decrease in total N introduced into the environment but would also reduce costs at the farm level. However, the impact varies across individual farms. The most expensive measures (with the highest CPUAs) are LU reduction by 20 percent and fencing off watercourses, which indicates that large abatement costs are to be expected if these measures are introduced.

Further analysis of the individual farm CPUA ranking reveals that the ranking of measures from cheapest to most expensive is not the same for all farms in the sample. This result indicates that no measure is strictly dominating (cheapest or otherwise) across all farms in the sample. Individual farms have their own CPUA ranking of the measures. In other words, the measure that is the most cost-efficient (cheapest) at the aggregate level may not be the most cost-efficient at the micro (farm) level. This finding has an important policy implication - an efficient policy should reflect the heterogeneity of the possible impacts and allow flexibility and innovation at a farm level. A flexible approach would yield superior economic results and would also reduce opposition to the implementation of the measures. However, there may be increased transaction costs to establish the most efficient strategies at a farm level.

A sensitivity analysis of the CPUAs to the price changes on the market is also conducted in chapter IV. CPUA analysis is now widely used to assess the merits or otherwise of various policy options for reducing harmful emissions from agriculture both for human health and the environment (Perez et al., 2009; Moran et al., 2009; Blok et al., 2011). However, such analyses are rarely extended to include sensitivity analyses of the results to changes in input and output prices. The sensitivity analysis
In chapter IV shows that the CPUA ranking is sensitive to input and output prices and can change in response to changes in market conditions. Thus, policy-makers when using CPUA results to inform policy decisions should be cognisant that efficient results may only be possible if such policies are flexible over time and allow decision-makers to respond to changes in market conditions.

In conclusion, in chapter IV I argue that policy should not only be targeted at the individual farm level to account for farm heterogeneity, but should also be designed in a flexible manner in recognition of the impacts of changes in agricultural input/output prices. Thus, policymakers who must choose a particular policy response based on an CPUA analysis should be cognisant of the extent to which changes in commodity prices can alter the conclusions of their analysis and hence the optimal policy instrument. This requires the policy instruments to be flexible, as rigid regulations may lead to an inefficient allocation of resources and a loss of income for the farming community. A more flexible set of policy instruments may increase efficiency while maximising the achievable pollution reduction. Given the dynamic nature of the agricultural sector, flexibility will be crucial when it comes to formulating farm management actions to reduce N pollution.

In chapter V of this thesis, the model developed in chapter IV is extended to include a spatial macrosimulation model to conduct a case-study of the cost to farmers of fresh water pearl mussel (FWPM) protection in Ireland. This case-study is important in a number of ways: 1) as in most countries within the EU, the FWPM is either almost extinct or exists only in small senescent populations which may become extinct before future generations arrive if a major hydro-ecological recovery in the catchments where they are present is not achieved (Araujo and Ramos, 2001; Geist, 2006; DEHLG, 2010); 2) the spatial heterogeneity of economic and environmental conditions means that the impacts of policy measures will differ greatly across space, thus, highlighting the importance of assessing policy implications at a relevant spatial dimension. There is an increasing demand to reduce ecological pressures on pearl mussel habitats in order to ensure the survival and recruitment of these critically endangered species. In response to this demand and in line with the requirements of the WFD, Sub-Basin Management Plans (SBMPs) for 27 river catchments in Ireland where populations of FWPM are present were developed. In these management plans
a list of possible measures to reduce environmental pressures on the FWPM at their source and along the pathways are cited. Measures that aim to reduce the pressures at their source would require (among others) a reduction in stocking densities, in livestock numbers or a reduction in fertiliser usage. In chapter V the likely impact of such measures is explored using microsimulation techniques (as described in chapter IV) applied to the NFS and SMILE data.

The results of this analysis suggest that reducing the pressures on the FWPM at their source (as specified in the sub-basin management plans) will come at a considerable cost to the agricultural sector in the catchments where they are present. Moreover, the analysis revealed that some mitigation measures may potentially lead to increasing the ecological pressure in another dimension if the farmers were to switch to different systems of production in response to a call to discontinue ploughing. This leads to the conclusion that the current plan to mitigate the pressure on the FWPM may lead to disproportionally high costs to the farming communities with the benefits of such measures being somewhat ambiguous. The overall conclusion that can be drawn from the results of analysis conducted in chapter V is that FWPM protective measures as currently envisioned may lead to significant costs to the agriculture sector, while alternative measures which focus on increasing farm efficiencies possibly via the introduction of new technologies may be superior means to protect the FWPM. These impacts will vary across different catchments and spatially referenced analysis is invaluable in designing policy responses to regulation requirements. Policy approaches that take into consideration spatial heterogeneity and encourage efficiency gains at a farm level would ensure economic efficiency and would also be consistent with the objectives of Food Harvest 2020, which envisions an increase of agricultural output to meet food demand of a growing population, while the current measures are inconsistent with this objective.

Chapter V contributes to existing research in a number of dimensions. It has both policy and methodological implications. The policy debates on environmental subjects are always limited due to data limitations. Thus, the results of the model provide additional spatially relevant data to those involved into policy analysis and decision-making on the costs to the farming communities of protecting the FWPM highlighting that not only the magnitude of the economic impacts but also their
differences across space. Methodologically, this chapter extends the existing methodologies by merging two microsimulation models within a spatial modelling framework in one model allowing the impacts of the mitigation measures to be assessed at different spatial resolutions in the areas where they are more likely to happen. This model also highlights the importance of the compatibility of different models and shows how building new models on existing research can enhance the models’ capabilities.

VI.3 Caveats and Future Work.

The results of the analysis conducted in this thesis represent important contributions to existing knowledge regarding the impacts that water protection as outlined in current legislation may have on farms in Ireland and the associated policy implications. Notwithstanding these contributions, some caveats and limitations are present and there is scope for future work to address these.

Caveats

First, the spatial model in chapter III is static and of an exploratory nature and as such it cannot be used to develop policy prescriptions by assessing the impacts of a particular policy. Due to the morphological structure of rivers (water flowing downstream and changing direction) a more refined spatial matching of the data may be needed as well as a dynamic analysis to account for the dynamic nature of environmental impacts. However, it is hard to judge whether this would yield better predictive power of the model as previous attempts to develop sophisticated agro-hydrological models at a national level have failed due to the complexity of the underlying assumptions needed to develop such a model.

Second, the microsimulation model described in chapter IV could be extended to include all feasible mitigation measures and, most importantly, the combination of measures. It would be desirable to include a separate model that would estimate the environmental spatially referenced outcomes of each strategy that could be linked to the model that is described in chapter IV. A major limitation of this model is that it does not currently allow the simulation of a combination of the mitigation measures that may be needed if specific N reduction targets are introduced. As a static model, it does not account for dynamic changes in farmers’ behaviour, thus, at the moment the
model does not allow one to simulate “what if” scenarios over an extended period of time. Thus, further extensions to the model will improve the model’s predictive capacity.

Third, the results reported in chapter V should be interpreted with caution. The results of this case-study are subject to the underlying assumptions and are mostly informative as presented in this thesis. In order to fully inform policy decisions, the whole range of the measures outlined in SBMPs should be assessed. This requires a number of different modelling approaches to be used. Then, the value of the benefits arising from these measures needs to be known. Thus, more analysis is needed which will depend mostly on data availability.

**Future Work**

It is evident at this point in time that after fifteen years since the introduction of the WFD there are still more questions than answers regarding the water quality improvement and the costs associated with it. It is also evident that more time may be required to reach the objectives set out in the WFD. The analysis conducted in this thesis contributes to existing research with insights into the possible costs of the WFD implementation as well as provides a platform for future research.

Future work stemming from the research conducted in this thesis will include the extension of the models described in chapters III and VI. Future work in spatial modelling as described in chapter III will involve refining the model with better spatial matching that takes into consideration the direction of the water flow in the river systems. Including other human activities into the analysis may also reveal more information about the sources of pollution in Irish rivers; however, the possibility of conducting such a study will depend on the availability of data at a very high spatial resolution. This would enable the assessment of the level of mitigation required by sectors such as agriculture and construction.

All policy-relevant measures need to be included into the model described in chapter VI. Extensions of the model will allow the combination of mitigation measures to be assessed, while dynamic simulations would include a behavioural model of farmer’s responses allowing one to simulate outcomes resulting from these measures into the future. A hydrological model could be linked to the economic model described in
chapter IV, which would allow the actual amount of N mitigated as a result of each measure at an individual farm to be estimated.

While the analysis conducted in chapter V revealed that the costs of protecting FWPM will be high, such information (despite being useful in financial planning) is not sufficient for efficient resource allocation. Including an analysis of relevant benefits would reveal whether it would be economically efficient to forego the costs of mitigation. Thus, future economic analysis of the WFD implementation will focus on valuing the environmental benefits that would result from the introduction of the mitigation measures. This will allow a cost-benefit analysis of the policy options that are potentially available to decision-makers and to the policy-makers in particular to be conducted.

The main conclusion that could be drawn from the existing literature and the contributions that this thesis makes to existing research is that the WFD implementation will come at a high cost and the policy-makers will be faced with a difficult task of designing policies to ensure environmental compliance while increasing food production to meet growing demand for agricultural produce. The success of future policies will depend on the availability of relevant information necessary to conduct research at a micro level and the advances in modelling methods helpful in overcoming data limitations.
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