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An Economic Valuation of Biodiversity and Livestock Production in Ireland

By

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Abstract

This thesis applies the use of economic valuation techniques for valuing biodiversity focusing on production function approach. Biodiversity provides economic benefits, protects human health and offers food, food safety, recreational and aesthetic enjoyment. Biodiversity contributes to the economy through the provision of many ecosystem goods and services. However, many of the services provided by biodiversity are not traded in the marketplace so do not have an actual price or commercial value. Economics provides valuation techniques to help the decision-making process make better informed choices over trade-offs between biodiversity protection and livestock production and contribute to the management of biodiversity.

This research aims to identify impacts, trade-offs and influencing factors that affect biodiversity, livestock productivity and environmental efficiency relating to biodiversity. Various econometric estimation techniques are used to find appropriate management solutions for biodiversity and livestock production.

In what follows a brief outline of each chapter is provided. Chapter 2 determines factors affecting land abandonment. It examines the effects of different livestock grazing management on land abandonment amongst farms that manage commona-ge in the west of Ireland. It has been suggested that off-farm employment has played an important role in maintaining farmers in the Republic of Ireland (RoI). However, this results in farmers having less time to devote to farming activities and environmental stewardship; traditional practices such as mixed grazing, haymaking and commonage that frequently yield important public good benefits such as the provision biodiversity and landscape amenity may be abandoned.

An ordered probit model is used to explain the probability of land abandonment. The results show that on-farm labour, livestock income and agri-environment

scheme payments are found to reduce land abandonment whereas off-farm income, livestock costs and farmer age increases land abandonment. The risk of land abandonment is more likely to occur in suckler beef enterprises and least likely with mixed grazing. Findings indicate that mixed livestock systems may play an important role in preventing abandonment of commonage lands or by restoring damaged commonage lands in the Irish uplands.

Chapter 3 investigates the impact of habitat fragmentation on biodiversity and livestock productivity in the west of Ireland. This study uses a two-stage regression estimation procedure to estimate the relationship between biodiversity and livestock productivity. In the first stage, a Cob Douglas production function is used to investigate the impact of habitat fragmentation on biodiversity. In the second stage, a livestock production function is used to analyse the effects of habitat fragmentation on livestock productivity. The results show that habitat fragmentation is negatively correlated with biodiversity and livestock productivity.

A specialisation in livestock grazing management has accelerated habitat loss and reduced biodiversity. On commonage lands, subsidies have positively influenced biodiversity in sheep farming whereas subsidies have no impact on biodiversity in mixed farming. In private lands, the results indicate slightly different findings. Subsidies are significantly and positively correlated with biodiversity in suckler beef and mixed farming. Empirical evidence suggests that a mixed grazing method was found to be the best management practice to improve biodiversity and livestock productivity. Mixed grazing management is found to be less vulnerable to habitat fragmentation and provides the highest total returns to livestock production compared to specialized livestock grazing management. It is, therefore, recommended that mixed livestock grazing should be better integrated into agri-environmental schemes such as the Rural Environment protection scheme (REPs) to conserve biodiversity and maintain productivity particularly for commonage.

Chapter 4 considers to what degree efforts to enhance farm profitability compromises biodiversity conservation goals amongst livestock farmers in the west of Ireland. This paper aims to deliver empirical evidence on the links between environmental efficiency, biodiversity, and livestock management by analysing commonage farms in Ireland. The relationship between profit efficiency and Environmental Efficiency (EE) is examined comparing specialised versus mixed grazing farms; private versus commonage farmers; and full time versus part time farmers. A three-stage estimation procedure is employed. First, a stochastic biodiversity frontier function is used to estimate environmental efficiency. Second, a fixed effect stochastic profit frontier model is used to estimate profit efficiency. Finally, a truncated model is applied to estimate factors affecting the variation in environmental efficiency. The findings indicate that there is an inverse relationship between EE and profit.

On commonage, there is a positive relationship between EE and stocking rate. Profitability and family labour have a negative impact on EE on private farms. On mixed farms, purchased feed decreases EE. However, stocking rate and the number of plots (land fragmentation) plays a positive role on EE. On specialised farms, environmental efficiency is negatively correlated with livestock profit, stocking rate, family labour, and subsidies whereas in mixed livestock farms environmental efficiency is negatively correlated with only purchased feed. However, there is no significant difference in EE between full-time and part-time farming.

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Abbreviations

AES	Agri-Environmental Scheme
ALS	Aigner, Lovell and Schmidt
CAP	Common Agricultural Policy
CFPs	Commonage Framework Plans
CNDF	Cumulative Normal Density Function
CSO	Central Statistical Office
DEA	Data Envelopment Analysis
DAFM	Department of Agriculture, Food and the Marine
EE	Environmental Efficiency
ES	Ecosystem Services
FEM	Fixed Effect Model
HNV	High Nature Value
IVs	Instrumental Variable
LIPS	Land Parcel Identification System
LQ	Linear Quadratic Model
MEA	Millennium Ecosystem Assessment
NUIG	National University of Ireland, Galway
NPWS	National Parks and Wildlife Service
PE	Profit Efficiency
PA	Premier to Area ratio
RoI	Republic of Ireland
REPS	Rural Environmental Protection Scheme
SFPS	Single Farm Payment Scheme
SPFM	Stochastic Profit Frontier Model
SWI	Shannon Weaver Index
TLU	Total Livestock Unit
2SLS	Two Stage Least Square
UNEP	United Nations Environment Program
UNCED	United Nation Conference on Environment and Development

Dedication

This thesis is dedicated to my wife, Sis. *Nigestie Embiale Admasu*, and my parents Mr. *Mulugeta Woldesenbet* and Mrs. *Worknesh Berhe*.

Chapter1. Introduction

1.1 Background

Despite the low number of species, many of Ireland's habitats are internationally important due to their scarcity elsewhere in Europe and the unique species communities found within them. Habitats of particular significance include limestone pavement (e.g. the Burren in Co. Clare), turloughs, active peatlands, intact sand dunes and machair systems, and some species-rich grasslands (Ireland CBD Report, 2007). The Irish government protects biodiversity under global instruments including the United Nations Conference on Environment and Development (UNCED) and the Convention on Biological Diversity (CBD).

Biodiversity is also maintained under the Common Agricultural Policy (CAP). One of the goals of CAP is to maintain agricultural productivity. A further goal (typically associated with agri-environment schemes) is to safeguard biodiversity. These major policy goals may be described as livestock productivity and enterprise competitiveness (profit efficiency) versus biodiversity related environmental efficiency. There exists the trade-off between productivity and biodiversity conservation. These can represent conflicting goals. Given the existence of competing objectives such as improving livestock productivity, biodiversity environmental efficiency, cultural, historical and scientific values requires some common measure of comparison or numeraire in the absence of a market value for biodiversity. In such cases, trade-off analysis is a useful approach. Maximizing provisioning of services from agro-ecosystems can result in a trade-off with biodiversity and other Ecosystem Services (ES), but thoughtful management can substantially reduce or even eliminate these trade-offs. Thus, agricultural management practices are key to realizing the benefits of ecosystem services and reducing disservices from agricultural activities. This thesis applies a trade-off analysis between the major policy objectives and goals in Ireland.

This research mainly uses production function methods to estimate the indirect value of biodiversity focusing on livestock production, particularly of meat products. Biodiversity valuation typically focuses on the economic values of ecosystem goods (livestock) and services generated by biodiversity resources (productivity). Many of the challenges for this study are to develop new methods of analysis in the valuation of biodiversity and livestock production.

The principal challenges in managing ES such as biodiversity and livestock productivity are that they are not independent of each other and the relationships between them may be highly non-linear (Heal et al., 2001; Farber et al., 2002; Rodríguez, et al., 2006). Attempts to optimize a single service often lead to reductions or losses of other services—in other words; they are "traded-off" (Holling & Meffe, 1996). As societies continue to transform ecosystems to obtain greater provision of specific services, they will undoubtedly diminish some to increase others (Foley et al., 2005).

Trade-off analysis is important to evaluate impacts of influencing factors that affect biodiversity and livestock productivity in order to make informed decisions about possibilities of alternative development options. Moreover, proper inclusion of tradeoff analysis and impact assessment in a decision-support system is essential for achieving wise use of ecosystem resources (De Groot et al., 2006). Understanding the relative impacts and trade-offs of different livestock management decisions and designing appropriate livestock production systems may help efficient provision of ecosystem services. Effective decision making which allows policy makers to include a comprehensive view of biodiversity trade-offs should address the cumulative and synergistic effects of their decisions. Successful strategies will recognize the inherent complexities of ecosystem management and will work to develop policies that minimize the effects of ES trade-offs (Rodríguez, et al., 2006).

Trade-offs occur when the provision of biodiversity is reduced as a consequence of increased use of ES such as livestock productivity. In some cases, a trade-off may be an explicit choice; but in others, trade-offs arise without premeditation or even awareness that they are taking place. Trade-offs between ES, in our case, between biodiversity and livestock productivity arise from management choices made by farmers who can change the type, magnitude, and relative mix of services provided by ecosystems. These unintentional trade-offs happen when we are ignorant of the relationships among ES (Tilman et al., 2002), when our knowledge of how they work is incorrect or incomplete (Walker et al., 2002), or when the ES involved have no explicit markets.

1.2 Biodiversity

The United Nations Convention on Biological Diversity (UNEP, 1992) defines biodiversity as follows: '*Biological diversity means the variability among living organisms from all sources, including inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; these include diversity within species, between species, and of ecosystems'*. Strictly speaking the word biodiversity refers to the quality, range or extent of differences between the biological entities in a given set. In total it would thus be the diversity of all life and its characteristics.

Habitat diversity refers to the diversity between habitats, while ecosystem diversity refers to diversity at a supra-species level, namely: at the community level. This covers the variety of communities of organisms within particular habitats as well as the physical conditions under which they live (Nunes et al., 2001).

This thesis uses the terminology developed by the Millennium Ecosystem Assessment (MA, 2003) concerning ecosystems and ecosystem services. The MA defines ecosystem services as follows. "An ecosystem service is the benefit that people receive from ecosystems" (Millennium Ecosystem Assessment 2003). Valuation is defined by the MA (2003) as the process of expressing a value for a particular good or service in terms of something that can be counted, often money, but also through methods and measures from other disciplines (sociology, ecology and so on).

1.2.1 Valuation of Biodiversity

Economic valuation of biodiversity is important because the multi-functional use of an ecosystem is usually economically more beneficial both to local communities and to society as a whole (Balmford et al., 2002). Biodiversity valuation techniques are needed to account for the full range of values generated by changes in the stock of biodiversity. Economic valuation tools are increasingly used to advise policy makers to take decisions about the use and management of biodiversity

Identification of factors that can affect biodiversity loss is an important step towards a complete feasibility study of the value of biodiversity and the livestock production process. Biodiversity valuation may help resource managers to deal with the effects of market failure by measuring the cost to society in terms of the lost benefits of a decision. To ensure balanced decision-making (i.e., multiple uses), it is crucial that the total value of biodiversity should be recognized – both market as well as non-market values. Non-market values are often missing in decision making. Such information has often not fully been taken into account when decisions are being made about economic development and hence conservation decisions may be undervalued and degradation of many ecosystems still continues (Barbier et al., 1997; Finlayson et al., 2005).

In particular, Finlayson et al., (2005) indicated that one of the major continuing drivers of loss and degradation of grasslands was that decision-makers either do not have full information on the total value (market and non-market) of ecosystem services or choose to ignore these, when faced with development versus conservation decisions. Thus, better communication of the benefits of biodiversity to decision-makers and the public in general is crucial.

A large part of this thesis is concerned with analyzing livestock productivity and efficiency, which is an indirect value of biodiversity. The productive value of biodiversity is related to the economic activities that directly affect the livelihood and income of the society. Grassland biological diversity may have an important value in supporting and protecting ecosystem functions for economic activity although the value of these functions is essentially non-market in nature (Barbier, 2000). To make consistent decisions between biodiversity preservation and livestock production options and a decision to halt biodiversity losses requires evaluation of the alternative options.

1.2.2 Total Economic Value (TEV)

People derive many essential goods and services from natural ecosystems including food, fodder, fuel, timber, and pharmaceutical products. These ecosystem goods and services represent important parts of biodiversity. What has been less appreciated until recently is that biodiversity performs fundamental life-support services without which human civilizations would cease to thrive (Daily et al., 1997). These include the purification of air and water, carbon cycle, food production, detoxification and decomposition of wastes, regulation of climate, and regeneration of soil fertility from which the essential ingredients of our agricultural, pharmaceutical, and industrial enterprises are derived.

Biodiversity provides many important functions for humankind which can be grouped in to direct, indirect and non-use values using the total economic value (TEV) framework (Barbier, 1994). Use values of ecosystem services include recreation and tourism such as the viewing value of wildlife and landscape and these are often non-market values.

Direct use value is known as extractive, consumptive or structural use value and mainly derives from goods which can be extracted, consumed or enjoyed directly (Dixon & Pagiola 1998). Direct use values include food production such as livestock and crop products. For example, direct use values compare livestock and non-livestock products with the social value of alternative benefits forgone (Pearce & Warford, 1993).

Indirect use value is also known as a non-extractive use value or functional value, and mainly derives from the services the environment provides. Most of the functions and services provided by biodiversity are indirect values. Indirect use value or functional value may be derived from a service that the environment provides where biodiversity takes the major share. Other indirect values associated with biodiversity include carbon sequestration, nutrient cycling, fishery nursery grounds and pollination. Indirect values can also be estimated using the production function approach.

Some of the indirect market valuation techniques used as a vehicle to value biodiversity include: opportunity cost, avoided cost (flood control), replacement cost (fertilizer), and mitigation cost. For example, reduction in productivity mainly caused due to loss of biodiversity needs restoration cost. Some services could be replaced with man-made systems; a good example is fertilizer use which can be (partly) replaced with costly artificial treatment systems. Mitigation or restoration costs - the cost of moderating effects of lost functions or of their restoration can be seen as an expression of the economic importance of the original service. A good example may be the cost of preventive expenditures in the absence of wetland service (e.g., flood barriers).

Non-use values include existence and bequest values. In many cases, the most important benefit of biodiversity is its existence value - the value that people derive from the knowledge that something exists, even if they never plan to use it. Existence value arises from ensuring the survival of biological resources (Pearce & Turner, 1990).

Existence value is the willingness to pay for the knowledge that a natural environment is protected by wilderness designation even though no recreation use is contemplated (Walsh et al, 1984). Walsh (1984) identifies several possibilities of willingness to pay for the preservation of wilderness resources in addition to consumer surplus from actual recreation use. Bequest value of biodiversity is the value derived from the desire to pass a good or service on to future generations, that is, to our children and grandchildren. This value may be estimated from past government expenditure incurred to protect endangered species.

The value of biodiversity also includes option value[1]. Option value is the value attached to maintaining the option to take advantage of something's use value at a later date (Spaninks & Beukering 1997). The general public may be also be willing to pay to maintain and enhance biodiversity in grasslands simply for the knowledge and satisfaction that such biodiversity exists and will be passed on to future generations. To estimate option value, probabilistic models have been used to explore the benefits of biodiversity for pharmaceutical products in grasslands. Some authors also distinguish quasi-option value from the option value which derives from the possibility that even though something appears unimportant now, information received later might lead us to re-evaluate it (Walsh et al, 1984). Option values and quasi-option values are of importance to resource users (farmers) as well as non-users (researchers) in grasslands.

[1] Option value is the difference between expected consumer surplus of recreation use and option price, defined as the maximum amount consumers, under conditions of supply and/or demand uncertainty, are willing to pay for an option to have a resource available for recreation use in time and each year for which payment is made (Bishop, 1982).

1.2.3Market Values

Many of the services provided by biodiversity are not traded in the market place and thus do not have obvious price or commercial value (O'Neill, 1997). If these non-priced values are not included in the decision-making process, the final decision may favour outcomes which do have actual prices in marketplace. Market failure occurs when markets do not reflect the full social costs or benefits of a good or service. Thus lack of markets is one of the main reasons for the cause of biodiversity loss in many ecosystems and concerns are held over the inadequate provision of ecosystem services (Pearce & Moran, 1994). Most of the ecosystem services generated by biodiversity are typically provided free of charge and often have the characteristics of public goods. Like other public goods, biodiversity services may not be provided optimally by aggregating the decisions of individuals motivated by self–interest (Polasky, 2008). The sum of individual actions may result in the disruption of the flow of valuable biodiversity, thereby making all individuals collectively worse-off (Polasky, 2008). When there are no explicit markets for services, it is necessary to estimate the value using non-market valuation means.

1.2.4 Externalities

Provision of biodiversity may also be affected by externalities — where the actions of a farmer has impacts on others for which he does not pay, or for which they are not compensated — thus markets will not function well. Economic analysis of biodiversity that fails to internalize biodiversity losses created by externalities may cause inefficiencies in production and lower productivity. Recent research has shown that when there is imperfect information these externalities are pervasive. It has been indicated by many economists that biodiversity will not be supplied in adequate quantities by the market on account of incomplete information. The presence of information problems suggests that ecosystem services are often not provided efficiently.

The primary threats of losses in biodiversity are land-use changes. Today escalating impacts of human activities on biodiversity and other natural ecosystem services risk the provision of such goods and services. The supply of biodiversity is often influenced by a different set of individuals than those who benefit from the provision of these services. For example, the farmer who maintains boglands/wetlands and limits fertilizer application provides benefits of cleaner water to individuals who live downstream. There is a mismatch between those who influence the supply of services and those who benefit from services and this gives rise to an externality problem.

1.2.5 Valuation Methods

Numerous methods exist to estimate the social value of conserving and enhancing biodiversity some of which were mentioned in section 1.2.2 (Garrod & Wills, 1997). Direct market valuation methods can be used to estimate for direct value of biodiversity. Direct market valuation is based on real market prices or the exchange values of ecosystem services. These are mainly applicable to demand functions, but also to some information functions (e.g., recreation) and regulation functions (nutrient cycle).

Revealed preference methods use actual market choices made by consumers in which the non-market good is implicitly traded to estimate the value of the non-market good (McFadden & Train, 1997 and McFadden, 1999). Stated Preference (SP) methods have been developed to solve the problem of valuing those non-market goods that have no related or surrogate markets (Adamowicz, Louviere, and Williams, 1994). In these approaches, consumer preferences are elicited based on hypothetical, rather than actual, scenarios. The choice experiment method, in particular, can be used to measure a change in the quantity or quality of multiple attributes of a public good (Adamowicz, Louviere, and Williams, 1994). The goods and services chosen are an entity of different attributes, and the marginal utility measured is an aggregate of marginal utilities from different attributes of the good or service. SP methods state that consumers derive utility from the attributes of a good, and not good itself (Sy et al., 1994; Tano et al., 2003). At a constant utility level, the negative of the ratio of two attribute coefficients will measure the Marginal Rate of Substitution (MRS), and the MRS turns out to be WTP if the cost of the product is included.

Hedonic pricing (HP) focuses on service demand that may be reflected in the prices people will pay for associated goods; for example housing prices at beaches usually exceed prices of identical inland homes near less attractive scenery. In this case, an implicit price of a product in a competitive market is a function of the product attributes (Lancaster, 1966).

The production function approach represents an important means of quantifying indirect values associated with a particular habitat or enterprise. The production function approach can be used to take account of how change in habitat quality affects production (Barbier, 2000; Foley et al, 2008). A number of studies have been conducted from many ecosystems to determine indirect values of biodiversity using the production function approach (Ellis & fisher, 1987; Barbier, 1994:2000; Barbier & Strand, 1998; Daily, 1997, and Foley et al, 2008). Barbier, (2002) used a dynamic production function approach to analyse the influence of habitat changes in marine shell fisheries in Thailand. The main focus of this study is on livestock productivity, biodiversity relationships and environmental efficiency analysis in order to identify impacts and possible trade-offs.

1.3 Overview of Research Objectives

The overall objective of this study is to investigate the impacts and trade-offs that may influence biodiversity preservation, livestock productivity and environmental efficiency of extensive livestock production systems in the west of Ireland. This multi-paper thesis explores various topics at the intersection of environmental economics and agricultural economics, with an emphasis on the causes and productivity effects of biodiversity.

The specific objectives of this thesis include: an impact assessment of government subsidies on the performance of biodiversity improvement, the impacts of enterprise competitiveness measured in terms of profit efficiency on performance of biodiversity as well as influences of part-time and off-farm employment opportunities. This thesis aims to provide detailed information on economic valuation and links between biodiversity and livestock productivity. It provides on an overview of the concepts of biodiversity related to environmental efficiency. The policy and empirical objectives of this thesis are outlined below.

1.3.1 Policy objectives

Policy makers need tools to determine the appropriate trade-offs between biodiversity protection and the human activities that create value for society. Knowledge and awareness of the relationships between ES are necessary for making sound decisions regarding how to manage natural systems appropriately (Grasso, 1998; Rodríguez et al., 2006). Government regulations routinely ignore trade-off decisions because, in many instances, the potential decline in human well-being is deferred until cumulative loss of biodiversity passes some critical threshold. Management decisions often focus on the immediate provision of an ES, at the expense of this same ES or other services in the future. In addition, policies need to acknowledge that short-term demands on ES will often affect the longer-term, larger-scale provision of these or other ES (Rodríguez et al., 2006).

Biodiversity valuation may help illuminate which policy or management options generate the greatest social welfare. For example: Does a management option that increases livestock productivity also result in a loss of biodiversity? The answer to this question depends on how one can view the trade–offs between various services. In this thesis, policy and management actions chosen to accomplish certain objectives, such as increasing productivity of live animals and profit efficiency of the business often have effects on biodiversity and the services they provide. Understanding the full consequences of policy or management decisions and comparing the benefits and costs of alternative choices can result in better policy and management for use of land and natural resources. Thus the economic valuation of biodiversity has the following policy objectives:

i) To determine factors that affect the probability of land abandonment on private land and commonage;
ii) To evaluate the impacts of habitat fragmentation on biodiversity and livestock productivity under different livestock enterprises management regimes and land property rights;
iii) To explore the impact of livestock competitiveness measured in terms of profit efficiency on biodiversity-oriented environment efficiency; and

1.3.2 Empirical objectives

Polasky, (2008) argues that information on the potential trade-offs and impacts among different ecosystem goods and services are important for specific policy objectives and empirical investigations. The economic valuation of biodiversity in this study has the following empirical objectives:

i) Investigate the impact of off-farm income and subsidies on the probability of land abandonment on private land and commonage;
ii) Assess the impacts of agri-environmental subsidies on biodiversity and livestock productivity;
iii) Determine the factors affecting biodiversity loss and livestock productivity under different livestock management and land property rights;
iv) Determine factors affecting livestock profit based on stochastic profit frontier model;
v) Analyse the impacts of off-farm income on biodiversity-oriented environmental efficiency and test whether there are differences in environmental efficiency between part-time and full-time farms.
vi) Evaluate the source of variation in biodiversity-oriented environment efficiency in mixed livestock and specialised farms.

1.4 Thesis structure

The main focus of this thesis is on indirect values using the production function approach. Chapter 2 determines factors that affect the probability of land abandonment. Chapter 3 investigates the impacts of habitat fragmentation on biodiversity and livestock productivity under different livestock enterprises management regimes and land property rights. In chapter 4 biodiversity related environmental efficiency is estimated. Finally, chapter 5 concludes and makes recommendations. Each chapter is outlined in more detail in the following paragraphs.

The identification of factors affecting the abandonment of agronomic practices and the implications for biodiversity loss has received little attention in the literature. Labour allocation decisions are important for biodiversity provision and **chapter 2** deals with this issue. An important problem raised in this chapter is that the numbers of farmers are declining while farmers working off-farm have increased due mainly to economic reasons. Farm behavioural models are analysed to identify factors that reduce land abandonment and increase farm participation in order to sustain farming and biodiversity provision. The issue of part-time farming and its association with land abandonment has been dealt with extensively in the literature and an account of this is provided in chapter 2.

Chapter 3 looks at the trade-off between productivity and biodiversity provision. The main focus of this chapter is analysing indirect values. The technique used here for analysing the relationship between biodiversity and livestock productivity is the production function approach which is based on valuing biodiversity as an input into the production process (Ellis & Fisher, 1987; Freeman, 1991; Barbier, 2000).

One important way of addressing the problems of biodiversity loss in the face of market failure is through direct government intervention. Government and EU subsidies are crucial to maintaining the livelihoods of most Irish farmers.

Government payments could also influence farm survival through capital market mechanisms. Government transfers effectively raise a farm's net worth (Key & Roberts, 2006). Economic impacts and subsides and their influence on biodiversity and livestock productivity is, therefore, an important component of this work. Iraizoz et al., (2005) analysed technical efficiency and profitability in the Spanish beef sector and found that subsidies have a positive impact on efficiency. Tzouvelekas et al., (2001) argued that the use of subsidies might lead to increased technical inefficiency, especially if subsidies attract more efficient farmers who are more interested in improving efficient farming practices than additional support.

There is a gap in the literature in terms of an analysis of the impacts of agri-environmental subsidies on biodiversity and environmental efficiency. Chapter 4 is concerned with analyzing livestock productivity and efficiency, which is an indirect value of biodiversity. The productive value of biodiversity is related to the economic activities that directly affect the livelihood and income of society. Grassland biological diversity may provide important supporting and protecting ecosystem functions, however the value of these functions are generally non-market in nature (Barbier, 2000). To make consistent decisions and tradeoffs between biodiversity preservation and livestock production evaluation of the alternative options is required.

It is also important to measure indirect values of biodiversity and environmental efficiency linkages in the context of pastoral livestock systems. Efficiency with respect to biodiversity resources is a necessary (but not sufficient) step towards environmentally sound livestock production. A growing body of literature has explored the relationship between environmental and technical efficiency. There is a well-established literature on technical efficiency (Kumbhakar et al., 1989; Bollman, 1991; O'Neill & Matthews, 2001; Goodwin & Mishra, 2004; Carroll et al., 2007; Solis et al., 2009). However, there are a limited number of works that explore the effects of positive externalities (such as biodiversity) on environmental efficiency. Chapter four investigates the effects of such positive externalities on environmental efficiency. Finally, **chapter 5** concludes and makes policy recommendations and suggestions for future research.

Chapter 2. Pastoral management and off-farm employment: An ordered probit model of land abandonment and mixed grazing in Ireland

2.1 Introduction

Off-farm income has become an increasingly important part of rural household income in many European member states including Ireland, the United States and Australia (Gasson, R., 1986; Pfeffer, 1989; Weis, 1999, Glauben, 2007; O'Brien & Hennessy, 2007). Part time work has become a way of life for many farm families and it has been argued that off-farm income has prevented farmers from exiting farming. It has provided the economic means of holding on to the family farm and farming as a way of life (Gasson, 1986; Jensen et al., 1995; Kimhi, 2000; Gillespie, 2003; Dickey & Thedossiou, 2006; Commins, 2008).

Although off-farm employment opportunities have economic advantages, farmers have less time to devote to farming and agronomic practices are either abandoned or farmers leave farming altogether. Many of the regions where land abandonment is known to occur coincide with economically marginal areas that are known for their biodiversity and amenity values (Walther, 1986). The cessation of agronomic practices such as mixed grazing, haymaking, and stone walling to maintain high nature value farming systems is a threat to the provision of public goods such as landscape amenity and biodiversity. The issue of part-time farming and its association with land abandonment has been dealt with extensively in the literature review below. However, the identification of factors affecting the abandonment of agronomic practices and the implications for biodiversity loss has received little attention in the literature.

Baldock et al., (1996) report that extreme remoteness and physical disadvantage reduces competitiveness and increases abandonment. Changes in off-farm labour markets, rural depopulation, low relative prices for agricultural products and infrastructure development have been identified as important drivers of land abandonment (Baldock et al., 1996; MacDonald et al., 2000). In Western Europe, abandonment appears to be mainly driven by industrialization, urbanization and market-orientation (MacDonald et al., 2000). The supply of off-farm labour has been shown to be positively related to urban proximity (Lass et al., 1991). Bokdam & Gleichman, (2000) point out that increased labour costs have rendered traditional herding systems economically unworkable. Caskie et al., (1991) suggest that a consequence of low income for Irish farm households is that government transfers and farm subsidies constitute a large proportion of farmers' income in these regions. This makes farmers vulnerable to changes in subsidy payments. In Ireland, a tendency toward part-time farming has occurred in upland[2] regions where livestock farmers have declined in number due to a lack of successors (Visser et al., 2007). Weiss, (1999) find that the effect of age on the probability of survival is increasing for young farmers and then deceasing for higher age groups.

The move toward part-time farming and the abandonment of certain agronomic practices may have important implications for biodiversity loss and environmental degradation. Abandonment of low-intensity traditional farming systems is now seen as a threat to High Nature Value (HNV) farming because agronomic practice that are thought to support biodiversity are being abandoned (Baldock et al., 1996; Bignal et al., 1996, Bignal et al., 1998; Webb, 1998; Bokdam & Gleichman, 2000). These practices include abandonment of upland hay meadows (Smith & Rushton, 1994; Jefferson, 2005) and hedgerow (Keena, 1998), stonewall, field margin maintenance and a tendency to replace mixed grazing with specialized systems. It is the link between abandonment and mixed grazing as a traditional practice that is the central focus of this study.

A review of the literature suggests that scant attention has been specifically devoted to an investigation of the empirical relationship between land abandonment and part-time farming as well as mixed grazing management in the

[2] In Ireland, the uplands are defined as all land above 300 m. s. l. (IUF, 1995)

context of private and commonage lands. Bakker et al., (1998) have suggested that more has to be done to make clear the effects of mixed livestock grazing at different stocking rates in different temporal sequences. Vickery et al., (2001) reported that relatively little is known about the impact of livestock enterprises on biodiversity for abandoned land. Bokdam & Gleichman, (2000) indicated that the design of effective enterprise management offers an opportunity for model validation. Research on mixed grazing and stocking rates constitute key drivers of pastoral systems and their biodiversity and thus is worthy of consideration at this time (Grant et al., 1996).

The effects of mixed livestock management on land abandonment are not well documented in heathland habitat (Grant et al., 1978). Consequently, this paper aims to fill this research gap in the literature. The main objective of this paper is to determine factors that affect the probability of land abandonment on private land and commonages. Specifically the study also investigates the impact of off-farm income and subsidies on the probability of land abandonment on private land and commonage.

The structure of the paper proceeds as follows; first a detailed background on part-time farming, land abandonment, mixed grazing and commonage is given; second a description of the survey and methodological approach is provided; third empirical results are discussed and finally conclusions and recommendations are drawn.

2.2 Background

2.2.1 Part-time farming and drivers to land abandonment

Kimhi, (2000) asks a question: Does the part-time farming phenomenon represent a stable situation whereby the land holder remains in agriculture over the longer term or, alternatively, is it just a step on the way out of agriculture altogether. The literature appears to be divided as to whether part-time farming represents a ‘stable phenomenon' rather than the first step towards land owners exiting out of farming (Walther, 1986).

In Germany, Pfeffer, (1989) reported that those who farmed part-time had lower expectations of continuing to farm in the future. Part-time farming is often seen as a stepping stone on the way out of farming and as means of facilitating structural reform of the German farming sector (Pfeffer, 1989). Higher proportions of part-time farms were related to high farm abandonment rates (Gellrich et al., 2007b). Kimhi (2000) suggests that gradual land abandonment occurs partly due to a transformation from full-time to part-time farming and part-time farming is one of the channels in which labour is moving out of agriculture.

Farming as part-time job can be considered as a gateway to land abandonment and abandonment of traditional livestock farming practices. Land abandonment is thought to be more prevalent in places where part-time farming is common (Gellrich et al., 2007a). Part-time farmers often have less time to work on their farms.

Research also suggests that land may be abandoned as a result of a reduction in farm income (Baldock et al., 1996; MacDonald et al., 2000). The local demand and supply of labour and land have affected land abandonment decisions (Gellrich et al., 2007a). Weiss, (1999) indicates that there is a negative relationship between part-time farming and farm expansion. For example, off-farm work increased the exit probability of Austrian farmers (Weiss, 1999). Patterns of farm entry and exit are thought differ by farmers' age group (Gale, 1993).

On the other hand, Weiss, (1999) indicates that policy makers seek to promote part-time farming as a stable element for gradual structural change of the farming sector. Ahearn et al., (2006) have remarked that off-farm work can no longer be viewed as a transition position but rather as a lifestyle choice with farming being perceived by the farmer as a second job. Goetz & Derbertin, (2001) suggest that off-farm employment stabilises household income and lowers the transaction costs of closing down the farm. Bollman & Kapiatani, (1981), Kimhi and Bollman, (1999) and Kimhi, (2000) found that the probability of exit from farming is decreased by working off the farm. Off-farm income has a stabilizing influence in terms of the disappearance of small holdings (Gassson, 1986). Shively, (2006) shows that the existence of off-farm employment opportunities can reduce deforestation and erosion damage of using improper farm practices. He concluded that in order to reduce the level of negative externalities produced by farmers, government may provide opportunities for off-farm income.

A number of factors may influence part-time farming and land abandonment. These include demographics, human capital, risk and uncertainty, intensity of production and intergenerational transfer.

2.2.1.1 Demographics

The characteristics of land abandonment vary considerably according to demographic factors. Weiss, (1999) finds that age, sex, education, farm size and off-farm employment significantly influences farm growth and survival. The impact of a farmer's age on the probability of farm survival is nonlinear. Weiss, (1999) also finds that the effect of age on the probability of survival is positive for young farmers and becomes negative as the age exceeds fifty years. Walther, (1986) also indicated that mountain people may be less adaptable to farming innovations due to older age and skill constraints. Increasing famers' education can be expected to increase their off-farm wage rate and farm output (Huffman, 1980). Dickey & Thedossiou, (2006) also argue that education is an important factor that shapes the characteristics of secondary job market. Gellrich et al., (2007a) show that whether cultivation continues depends on whether neighboring land is abandoned or not. Factors such as age and education of farmers, farm income, farm size and the degree of mechanization are important factors affecting land use changes (Baldock et al., 1996; Macdonald, et al., 2000).

One explanation given is that immigration is related to land abandonment. Gellrich et al., (2007a) report that a positive relationship between immigration and land abandonment. Mundlak, (1978a) shows that occupational migration is often motivated by income differentials. Income differences between on-farm and off-farm jobs have been identified as important drivers of changing land use intensity and land abandonment (Gellrich et al., 2007a). There is also a cost advantage of family labour relative to hired labour. Off-farm job opportunities lead to a higher opportunity cost of farm labour, which is one of the main determinants of land abandonment.

In Ireland, farm size, farmer's age, education, marital status, geographical location, farm investment and seasons of farming are identified as factors affecting the decision to adopt off-farm employment (O'Brien & Behan, 2007). It is thought that a large number of farm households would not have survived without off-farm employment (O'Brien & Hennessy, 2007). Hennessy & O'Brien, (2007) suggest that there may be a positive relationship between off-farm income and farm investment. However, Carroll et al (2007) show that there is no significant difference in technical efficiency between full-time and part-time farming.

In Western Europe, Breustedt & Glauben, (2007) report that exits from farming are strongly influenced by household characteristics and policy conditions: exit rates are higher among smaller farms and are closely related to particular enterprise and production structures. Exit rates are lower in regions with more part-time farming, high subsidy payments, and high relative price increases for agricultural outputs (Breustedt & Glauben, 2007).

2.2.1.2 Human capital

Human capital has an ambiguous effect on land abandonment. An increase in human capital can be expected to improve the effectiveness of a farm operator in allocating resources and adopting new technologies (Zepeda, 1990) which should translate to higher growth and survival rates. Human capital is thought to contribute positively to farm growth thereby reducing the likelihood of farm exits (Weiss, 1999), which may reduce land abandonment. Human capital may increase off-farm earning capacity which has the opposite effect by reducing the probability of farm survival (Goddard et al., 1993). On the contrary, a farmer's opportunity for employment outside the farm also increases with human capital and raises the probability of switching to part-time farming or exiting from the farm sector (Weiss, 1999). If the operator does not exit from farming, switching to part-time farming may lead to an increase in land abandonment.

2.2.1.3 Risk and uncertainty

Barlett, (1991) found that the primary reason farmers worked off-farm was to reduce variability, risk and uncertainty associated with fluctuations in farm income. Huffman, (1980) showed that variance of income was significantly correlated with off-farm labour supply. Sander, (1986) found that total income was significantly less variable when a farmer and spouse were involved with off-farm labour. Farm income variability as represented by the coefficient of variation on farm income has a significant positive effect on the off-farm labour supply of farmers (Mishra & Goodwin, 1997). Higher farm income variability increases off-farm income. Schultz, (1990) pointed out that off-farm employment is an important means by which farmers and their spouses attempt to reduce the variance of total income. It has been shown that the variance of farm income is perceived to be greater than off-farm income due to livestock price uncertainty thus farmers supply less labour to the farm (Mishra & Goodwin, 1997).

2.2.1.4 Intensity of production

There are a number of other characteristics associated with land abandonment. Whether farms are extensive or intensive seems to matter. Intensive production systems such as dairying and poultry tend to be associated with full-time farming whereas extensive systems such as suckler beef or sheep are linked to part-time farming. Dairy farmers often require high wage rates to switch to off-farm employment. Bokdam & Gleichman, (2000) indicate that part-time farming lead to the abandonment of traditional agriculture practices. There are other decisions such as being an inactive shareholder in commonage farms and abandoning haymaking may lead to more land to be abandoned. Abandonment of traditional agricultural practices (Bignal et al., 1996) such as mixed farming and haymaking is the most common during part-time farming.

2.2.1.5 Inter-generational transfers

The demand for agricultural land might rather be affected by a declining agricultural population (Gellrich et al., 2007a). There is a concern in the farming community concerning declining numbers of young people entering the farming sector. There is also a trend that the farming workforce is aging, the probability of having a successor increases with age and at extreme age declines again (Glauben et al., 2004). Farmers may abandon the land if they cannot find a successor. The inability of the natural resource base to provide adequate income for the farming population and relatively scarce employment opportunities have resulted in the depopulation of a younger generation and better educated people from farming. Macdonald, et al., (2000) suggest land abandonment is related to rural depopulation to which isolated and poorer regions are more vulnerable.

Gale, (1993) and Gasson & Errington, (1993) concluded that the transfer of the farm to the next generation is often the main objective of full-time farming. Kimhi & Nachlieli, (2001) found that the age of the farmer, education and age of the elder child were significant factors affecting intra-family succession.

A farm successor may attach different marginal utilities to time devoted to on-farm work relative to off-farm as compared to previous ancestors (Wallace & Jack, 2011). Kimhi & Lopez, (1999) suggest that a large proportion of farmers abstain from transferring the farm while they are still alive. Laband & Lentz, (1983) state that farmers are nearly five times more likely to have followed in their fathers' footsteps than any other self-employed proprietors. Pesquin et al., (1999) indicated that the family farm sector relies heavily on intergenerational succession and also mention additional advantages of intra-family farm succession such as a safe transition, reduction in transfer cost, and lower transfer taxes. Additionally, a farm or small business can reduce its tax burden by declaring a successor (heir) (Mishra & El-Osta, 2008).

Furthermore, Mishra & El-Osta, (2008) point out that intra-family farm succession allows new younger entrants to overcome borrowing constraints at least in commercial farms. It provides an implicit contractual insurance arrangement since the generations overlap and share income (Glauben et al., 2004). Mishra & El-Osta, (2008) found that farm capital stock, livestock, farm machinery and equipment, and farm buildings, at the beginning of the year had a positive and significant effect on the succession decision of farm operators. Larger farms tended to have higher capital stocks, resulting in higher earned incomes for the operators. A further consequence is that farm work becomes more attractive for the successor of the farm business relative to other occupations or relative to working off the farm.

2.2.2 Part-time farming and the off-farm labour market

In what follows we provide some background on the characteristics of farmers that engage in the off-farm labour market. Dickey & Thedossiou, (2006) indicate that multiple job-holding is carried out primarily by farmers of low socio-economic status. Off-farm employment is most likely to be adopted when the farmer is relatively young and willing to improve their economic status. The additional income has a stabilizing influence preventing small farmers exiting farming (Weiss, 1999).

Huffman, (1980) indicates that there is a positive relationship between expected wages and off-farm employment participation. An increase in the off-farm wage leads to a decrease in on-farm hours (Lass & Gempesaw, 1992). If one employment opportunity has greater expected marginal returns then more labour will be devoted to that activity (Dickey & Thedossiou, 2006). Weiss, (1999) suggests that farm size is an important determinant of off-farm labour market behaviour. Farm size is negatively correlated with farmers' off-farm labour supply (Mishra & Goodwin, 1997). Kimhi & Bollman (1999) suggest that farm size or farm value would positively contribute to farm survival because larger farms are more likely to provide the farm operator and his family with a reasonable and sustainable income.

Lass et al., (1991) found that the supply of off-farm labour has been positively associated with education and farm experience. Mishra & Goodwin, (1997) indicate that years of farm experience is a significant determinant of off-farm labour supply of farmers and their spouses. They show that farm experience corresponds to less work off the farm. Wallace & Jack, (2011) find that farmers who engaged in off-farm employment are more likely to have higher levels of education. Welch, (1970) reported that education contributes to production efficiency through a 'labour allocation effect' and 'worker effect'. The worker effect is the effect of education on efficiency where a more educated farmer has a higher level of efficiency. The decision of farmers to continue education is a function of their labour allocation on the farm. Women are less likely to participate in off-farm work if they have children (Furtan et al., 1985) whereas the probability of off-farm work for farm men is increased by having children (Goodwin & Mishra, 2004).

In Ireland, the trend toward part time farming is likely to continue in the future. This has resulted in a fall in livestock production and income from livestock and increased off-farm income (IHC, 1999; Binfield & Hennessy, 2001). O'Brien & Hennessy, (2007) report that about 54% of income was derived from farming in 1994 but by 2006 this had fallen to 34%. In marginal and upland areas of Ireland, the number of farmers who had worked off-farm increased from 37% in 1995 to 58% in 2007 (O'Brien & Hennessy, 2007).

2.2.3 Government payments and the link to land abandonment

Economists have recognised that subsidies influence the growth and survival of farming communities (Key & Roberts, 2006) and government payments can influence farm survival and land abandonment. Government payments may protect farms from exiting agriculture through land price and capital market mechanisms (Key & Roberts 2006). Farms receiving high government payments per hectare are usually better able to bid for high land prices causing low payment farms to exit thus reducing land abandonment (Key & Roberts, 2006). Government payments tend to decrease the likelihood of a farmer working off the farm or decrease the amount of off-farm working hours (El-Osta & Mishra, 2008). Government payments may also make agriculture more profitable relative to off-farm employment, reducing the incentive to exit farming (Key & Roberts, 2006). Payments increase returns and may also ease liquidity constraints (Mishra & El-Osta, 2008).

Government payments could also influence farm survival through capital market mechanisms. Government transfers effectively raise a farm's net worth (Key & Roberts, 2006). Different capital market structures and risk considerations may affect farmers' investment and disinvestment decisions and may also alter the change in the number of farmers (Serra et al., 2004).

Key & Roberts, (2006) find that government payments are positively correlated with farm survival for both small and large farms. The effect of government payments on farm survival continues to be an important issue in ongoing international negotiations where agricultural support schemes are a major source of contention (Key & Roberts, 2006).

In Ireland, agri-environmental protection schemes such as REPS have been in operation for more than two decades to ensure the continuation of farming, support rural population levels and conserve the countryside. Agri-environment schemes are designed partly to reduce land abandonment (Primdahl et al., 2003). Ahearn et al., (2006) report that both decoupled and coupled payments help to decrease off-farm work hours in USA. Farm investment is likely to be greater after decoupling than in the absence of such payment (Andersson, 2004).

Barkley, (1990) suggests that government payments do not necessarily influence changes in the structure of agricultural employment. Government payment may have indirectly slowed the rate of depopulation from agriculture through higher land prices. In Ireland, O'Brien & Hennessy, (2007) report that the introduction of decoupled payments has a significant positive effect on off-farm labour allocation.

The effects of government payments on land abandonment may continue to be an important issue in ongoing decisions regarding biodiversity and environmental protection during international trade discussions over agricultural products. According to Wallis De Vries et al., (2007), economic analysis has indicated that agri-environment support for farmers is essential to reconcile sustainable livestock grazing systems with high biodiversity.

2.2.4 Commonage

An important feature distinguishing Ireland from other EU countries is the existence of a high amount of commonage land mostly in upland areas (van Rensburg et al., 2009). Commonage is land held in common ownership on which two or more farmers have grazing rights (Lyall, 2000). The boundaries of contemporary commonage were created by the Irish Land Commission who formally granted grazing rights to Irish tenants during the period of land reform from the end of the 19^{th} Century until the 1980s as a form of land distribution (Lafferty et al., 1999). The term 'commonage' refers to jointly owned lands, usually agriculturally marginal areas such as moorland and coastal dunes, on which the joint owners (shareholders) hold grazing rights. Other shared rights may include grazing on commonages, peat cutting on moorland, and dune grass harvesting and seaweed collection in coastal areas. Traditionally, the total number of grazing units (in effect the carrying capacity) on a commonage was calculated, and these units were then allotted to each landholder in proportion to the size of his infield holding.

Farm households continue to be key stakeholders of Irish commonage, although other interest groups, particularly recreation and conservation bodies have become more involved in its stewardship in recent years (Phillips & Tubridy 1994; Hynes et al., 2007). Access to any commonage is restricted to a group of shareholders who have the legal right to exclude non-shareholders thus creating the potential to prevent the tragedy of the commons. Commonage can be considered a Common Property Regime rather than open access. Commonage as a system of land tenure is significant for Irish agriculture, upland conservation, and for sustaining rural livelihoods. Active commonage shareholders continue to graze their livestock, they participate in measures to prevent overgrazing and are involved in the maintenance of commonage infrastructure (fencing, stonewalls, drainage).

Commonages are recognised as being of high recreation and conservation value and include special areas of conservation, special protection areas and natural heritage areas (van Rensburg et al., 2009). Policy makers have become concerned with threats to commonage and their public good values. The European Council Habitat Directive (EEC, 1992) lists commonage semi-natural grassland habitats as being of European importance for their biodiversity value. It has been estimated that this habitats contain 65 pasture types that are under threat from overgrazing and 26 that are under threat from abandonment (Rook et al., 2004). Baldock et al., (1996) have introduced the term High Nature Value (HNV) farming for farming systems associated with commonage habitats.

However, commonage grasslands are considered to be vulnerable to the risk of land abandonment. Changing demographic patterns, amalgamation of farm holdings, modernization of farming methods and emigration have led to fewer active commonage shareholders and in many instances a decline in the quality of management (Di Falco & van Rensburg, 2008). Property rights and the non-excludable nature of commonage are known to affect land management (Ostrom, 2000). Consequently, information on the number of active shareholders, abandonment of commonage and farmer attitudes to commonage was sought in this analysis.

2.2.5 Haymaking

Haymaking is a traditional agricultural practice whereby grass or legumes are cut, dried, and stored for use as animal fodder. Upland hay meadows have significant aesthetic and recreational value. However, many of the farm practices associated with haymaking are being abandoned and upland hay meadows are becoming increasingly scarce as a result of abandonment of haymaking (UK Biodiversity Group, 1998; Jefferson, 2005; Critchley et al., 2007). Smith et al., (2000) showed that as livestock production moves away from haymaking, there is a greater reduction in plant species diversity. Farmland abandonment has been associated with a cessation of management which leads to undesirable changes in biodiversity and ecosystem services. Abandonment of haymaking is harmful for HNV farming (Bignal & McCracken, 2000). Isselstein et al., (2005) also raise the issue of compensation payments for farmers to support forage production of high-biodiversity value species because farmers can supply these environmental goods more cost efficiently.

Traditional extensive livestock farming and haymaking are often considered to be important practices for the survival of many threatened plant and animal species in Western Europe (Bignal & McCracken, 2000). Extensive livestock systems are acknowledged as a means of maintaining and restoring open managed landscapes (van Braeckel & Bokdam, 2002). Attributes of this farming activity include low livestock density, low chemical inputs, and the adoption of traditional livestock breeds (Baldock et al., 1994).

The abandonment of grasslands is thought give rise to a decline in plant species richness (Persson, 1984; Bakker, 1989, Hansson & Fogelfors, 2000; Pykala, 2003).

It may lead to farm biodiversity and habitat diversity loss (Bakker, 1998) and profoundly affects the ecosystem (Rey Benayas et al., 2007). Peco et al., (2006) found that abandonment may result in the loss of more than 60% of plant species. Small sized plants are particularly, sensitive to land abandonment (Pykala, 2004). Rare grassland species are less sensitive to abandonment than abundant species (Pykala, 2005). In abandoned grasslands, the persistence of plant species also depends on local abiotic conditions (Vandvik & Birks, 2002). Occasionally, species may be replaced by other species, in such case the total species richness may remain unchanged (Bakker, 1998).

Modern management of hay differs from traditional management in the time of cutting of grass and reduction in the length of forage harvesting (Smith et al., 2000). Some studies have quantified the effects of fertilizer in reducing species richness and increasing the yield of fodder (e.g. Tallowin et al., 1994) and the relationship between species richness and timing of grazing and cutting (Bakker, 1989; Smith & Rushton, 1994). The time of cutting has resulted in a further decline of species richness of the grasslands.

Traditional hay making are reported to be key to the conservation of upland fauna and flora (Baldock et al., 1994; Bignal et al., 1998; Visser et al., 2007). Such hay meadows are critically important for the endangered Corncrake (*Crex Crex*) (Stowe et al., 1993), a visiting bird to Ireland from Eastern Africa that is afforded high priority for conservation actions. Upland hay meadows have been identified as a priority for conservation in response to the 1992 Rio Convention on Biodiversity (Anon, 1998; Critchley et al., 2003) and supported by Biodiversity Action Plan Priority Species (Colenutt et al., 2003). The upland hay meadow provides important nesting and feeding habitat for various waders (Jefferson, 2005).

Substituting silage for hay has become more prevalent in the Irish uplands. The production of silage has increased from 0.3 million tonnes in 1960 to over 20 million tonnes in 1990 (GoI, 1997). Recently, the restoration of species-rich hay meadow communities has become an important issue in Ireland. The restoration of species lost from grassland as a consequence of land abandonment is a much more difficult task. In commonage grasslands, appropriate grazing management of the livestock sector has a critical component in the effort to restore and improve biodiversity. Long-term fertilizer use is a problem for the restoration of upland hay meadows and species-rich grass-lands (Janssens et al., 1998). Reseeding, drainage, and slurry may have the greatest damage on hay meadow biodiversity (Critchley et al., 2007).

2.2.6 Mixed livestock grazing management

The following provides a background on the effects of mixed grazing on biodiversity. Mixed livestock systems involve the use of two or more animal species to graze an area simultaneously. Historically farmers developed mixed grazing systems to utilize forage resources more effectively, for example, exploiting differences in grazing habits by cattle and sheep that can lead to a degree of complementarity in the use of forage resources (van Rensburg & Mill, 2010).

Biodiversity coincides with certain production goals such as stability of production under unpredictable environmental conditions. Farmers later recognised that a diverse system helps reduce variation in productivity from year to year thus using mixed livestock practices may promote biodiversity. Farmers may have evolved mixed grazing systems specifically to be able to exploit resources in species rich environments. Therefore, certain conservation practices succeed because are perceived to coincide with the interest of land managers. These so called HNV farming systems are also valued for their environmental benefits including biodiversity and the visual appearance of the landscape.

Mixed livestock has had a profound effect on landscape and biodiversity (Collins, 1989). Mixed grazing using sheep and cattle simultaneously is thought to be beneficial for biodiversity (Bignal et al., 1998; Dunford & Feehan, 2001; Vickery et al., 2001; Anger et al., 2002; Rook et al., 2004; Sanderson et al., 2009), and spatial diversity (Bignal & McCracken, 2000). Rook et al., (2004) have shown that the diversity of plants on pasture is highest when grazing heifers alone, followed by mixed grazing of heifers and sows. Co-grazing of sheep and cattle illustrates the importance of browsers in many grazing systems and shows how management practices can be employed to maintain or increase their prevalence and vegetation diversity.

Mixed grazing systems are widely recognised as being the best form of grazing management to promote biodiversity in the upland habitats. The biological importance of a mixed farming system relates to the spatial diversity, it introduces (Bingal & McCracken, 2000). The positive relationship between mixed livestock farming practices and biodiversity is the main reason these farming systems are considered of high conservation value (Anger et al, 2002). A mixed farming system with a high proportion of grassland habitats is likely to maintain a number of farmland bird populations in many European countries (Sanderson et al, 2009). Nature, (2001) stated that grazing by different animals has different effects on the mix of plants and the present of animal species. Mixed livestock systems lead to more diverse vegetation than single type livestock systems which in turn result in a greater range of plants and animals. Mixed systems, for example, are particularly important for breeding waders (Vickery et al, 2001). Abaye et al, (1994) shows that mixed grazing of sheep and cattle could be beneficial for quality of forage and performance of grazing animals.

Mixed livestock grazing systems may also be preferred to single systems because they improve yields and do not overexploit productive herbaceous species. It has been reported that sheep and cattle may affect the plant community in different ways. Bedell, (1971) has shown that sheep can reduce the abundance of clover in a sward. In contrast, a high proportion of cattle will increase the amount of clover relative to grass. Thus combined cattle and sheep grazing systems may be more productive than single species systems. Studies have shown that biodiversity tends to decline in areas where mixed grazing has been replaced by more specialised farming (Dunford & Feehan, 2001). Hamell, (2001) found that specialization in European agriculture increased pressure on the environment and led to marginalization and abandonment of farming practices.

Upland grassland areas in Ireland have traditionally been maintained through the use of mixed livestock farming systems that include hardy upland cattle and sheep breeds. However, farmers on Irish commonages have been switching to specialised sheep only at an alarming rate. This has led to increase sheep stocking densities relative to cattle. This practice can lead to a decrease in species diversity and habitat structure (Mitchell & Hartley, 2001). High sheep stocking rates are also thought to cause vegetation change and give rise to land degradation (Bleasdale, 1995; Evans et al., 2006). Fuller & Gough (1999) indicated that high stocking rate especially by sheep only negatively affect vegetation in many upland areas.

Mixed livestock farms together with a few specialized finishing farms appear to be the major controlling factors over vegetation change. Mixed livestock seems to have the potential to facilitate the restoration of diverse swards and to support reasonable individual performances of the grazing animals (Isselstein et al., 2005, Tallowin et al., 2005). In most rough grazing where land has been abandoned, to reduce land abandonment it requires mixed livestock grazing management and the restoration of traditional livestock practices.

In the German uplands, Anger et al., (2002) report a positive relationship between mixed livestock farming practices and biodiversity. Bignal et al., (1996) suggest that mixed livestock grazing is a fundamental mechanism through which low input extensive livestock farming influences biodiversity. McMahon (2008) find that upland bird numbers were higher in a mixed grazing system compared with a single-species grazing regime. A mixed farming system with a high proportion of grassland habitats is thought more likely to maintain a number of farmland bird populations than a more specialized system in a number of European countries (Vickery et al., 2001; Sanderson et al., 2009).

Mixed grazing may provide a more favorable foraging habitat for species by increasing vegetation structure heterogeneity and hence the availability of arthropods (Evans et al., 2006). Animut & Goetsch, (2008) show that mixed grazing can be used effectively in order to enhance plant diversity and animal performance but caution that high livestock densities can be harmful to habitat diversity. Bignal, (1998) indicate that the biological importance of mixed grazing systems relates both to the spatial and temporal diversity that it introduces. Reduced habitat heterogeneity in all farming landscapes is an important factor related to biodiversity loss when mixed grazing has declined.

Diverse multispecies herbivore systems, such as game ranches on the savannas of Africa, can include up to 20 different mammal herbivore species (Cumming, 1993). Short grass grazers benefit from the modification of sward structure brought about by long grass grazers, for example, sheep generally perform better when grazed in mixed systems than when grazed alone (Nolan & Connolly, 1977). This is usually only the case when large quantities of unpalatable poor quality fodder are available. McNaughton, (1984) reports that the bulk of grazers consume long grass. These are then followed by smaller ungulates that create ‘grazing lawns’. These lawns are sources of high quality forage and so herbivores are seen to influence the quality and productivity of the grazing resource.

Examples of rangeland management systems that attempt to encourage diversity in herbivore populations in order to enhance resilience include replacing monocultures of domestic livestock with multispecies game systems and combined cattle/game ranches such as the Campfire programme in Zimbabwe (Cumming, 1993). Scholes & Walker, (1993) have suggested that events such as fire and herbivory play an important role in maintaining the diversity and resilience of such systems. The reduction of such perturbations is thought to reduce landscape diversity and the ability of the system to survive similar shocks in the future.

Moreover, in African savannas (Walker et al., 1981) and in British grasslands (Hulme et al., 1999) grass species are important in maintaining the system's productivity. In other habitats, such as boreal and deciduous forests in North America and Europe, where insectivorous bird species are considered to be instrumental in controlling outbreaks of forest insect pests, overall species diversity is important for maintaining stability (Holling, 1988). There may also be indirect effects of diversity with some species influencing the survival of other species. Key plant species determine the course of successional processes through the provision of so-called 'nurse effects'. Several studies have observed a greater number of seedlings beneath mature trees compared to more open areas (Espelta et al., 1995). Similarly, shrub species may influence seedling establishment, acorn consumption and the extent of browsing by herbivores (Herrera, 1995).

2.2.7 Abandonment of traditional agriculture practices

One of the negative consequences of part-time farming is abandonment of traditional agriculture practices. Part-time farmers simply have less time to engage with the day to day business of farming. Bokdam & Gleichman, (2000) have suggested that abandonment is a major threat to traditional pastoral landscapes and their wildlife in Europe. They report that increased labour costs have undermined traditional herding systems, which are being replaced by free-ranging grazing systems leading to a decline in species rich open heathland. Many traditional extensive farming practices have been shown to maintain plant and animal diversity (González Bernáldez, 1991; Naveh, 1994). Where these activities cease, susceptibility to disturbances, especially fire, can be increased. Fire in turn can have a negative effect on biodiversity (Faraco et al., 1993).

Landscape homogenization can also result from the abandonment of agricultural/pastoral land (Fernandez et al., 1992). Without human management diverse plant communities in the Mediterranean basin, for example, become overgrown, and displaced by relatively few, shrubby unproductive species. When land is abandoned, there is a high probability of wild-fires (Romero-Calcerrada & Perry, 2004). Land abandonment is considered to have a high economic cost in connection with fire hazards. The management of Mediterranean woodland has become an important issue in many areas because of the abandonment of large areas that were previously exploited by grazing. In many cases impenetrable thickets have developed with continuous accumulation of fuel leading to catastrophic wildfires.

In managed landscapes good conservation practice often succeeds because it is perceived to coincide with the interests of land managers. Such conservation practices may also have been developed to avoid over-utilisation of the resource on which the human population depends. Consequently most biodiversity exists in human dominated ecosystems (Pimmental et al., 1992).

The development of species rich raised coastal dune and bog habitats in the North Western Europe, known as Machairs, is also thought to be strongly associated with agriculture and human activity, particularly fire and grazing (Mate, 1992; Edwards, et al., 2005). Machairs, which are priority habitats under the European Habitats Directive, are unique ecosystems confined, in the northern hemisphere, primarily to west and north-west coasts of Ireland and Scotland. Machairs are priority habitats because of the high plant species richness which contain elements of calcareous grassland and sand dune plant communities. In traditional land husbandry, maintenance of biodiversity and economic outputs are closely intertwined. For example, the relationship between habitat characteristics, weather and spatial variation in animal behaviour was investigated by De Miguel et al., (1997). They suggest that shrub areas provide shelter and represent an important browse resource during winter. This leads to the occurrence of a diversified landscape with different successional stages (from pastures to clear and dense woodlands) that occur in close proximity which in turn leads to high levels of flora and fauna.

Baldock et al., (1994, 1996) outline a general model of the abandonment process in which a series of changes take place that may involve modification of traditional agriculture practices and physical land abandonment. Spatial and temporal factors of abandonment play an important role in the succession process (MacDonald et al., 2000). At a regional level, landscape spatial abandonment may increase heterogeneity as productive and non-productive areas become more differentiated (Baudry, 1991). In the short-run, at landscape level, habitat diversity may increase as abundance of components increase but over the long-run abandonment would ultimately reduce diversity as certain elements dominate (MacDonald et al., 2000). In the early stages of abandonment, biodiversity is likely to decrease as aggressive or dominant species invade the grassland.

The dynamic pattern of effects of land abandonment on biological diversity is not yet fully understood (MacDonald et al., 2000). Land abandonment related to cessation of traditional practice rather than extensive physical land abandonment does not demonstrate negative landscape impact (MacDonald et al., 2000). In the medium term as scrub cover develops, the spatial degree of biodiversity may increase but then decline as the woodland canopy closes. The effects of abandonment that leads to increase woodland cover may not be desirable in terms of creating a variety of habitats. Abandonment also affects the remaining agriculture as one plot is abandoned this may make adjacent plots harder to manage through invasion of pests and weeds. However, the positive effects of these adjacent abandoned areas will serve as refuges for wildlife species which may contribute to pest control (CEC, 1980). There could be other positive effects of land abandonment. For example, land abandonment decreases soil erosion (Tasser et al., 2003) and improves water quality (Kramer et al., 1997).

2.3 An Ordered Probit Model

The use of ordered probit models is common when the dependent variable in a regression takes a qualitative and ordinal categorical form. In this case, dependent variable is land abandonment. In this model, a direct measure of land abandonment was not used because information on the actual amount of abandoned land from the total land area was not available. Instead, the number of abandoned traditional livestock farming practices is used as a dependent variable. Thus, we can specify an ordered probit model for unobserved land abandonment as follows:

1) $$Y^{*} = X\beta_{i} + \varepsilon_{i}$$

Where Y^* is a dependent variable representing land abandonment of the i^{th} farm, X is the vector of observed independent variables and ε is the error term. It is assumed that the disturbance term is a normally and independently distributed random variable. It is also assumed that there is heterogeneity in land abandonment across farmers' in terms of the use of different grazing management regimes (i.e., cattle, sheep and mixed systems).

Farmers are considered to be the major decision makers in choosing livestock production practices. Because it may not be possible to observe Y^*, we instead can observe abandonment of traditional agricultural practices such as part-time farming, inactive commonage shareholding and abandonment of haymaking.

In this paper, three indicators of land abandonment farm characteristics are identified as building blocks of land abandonment. These farm characteristic indicators are: part-time farming, inactive commonage shareholders, and abandonment of haymaking. The fact that information on the actual area of land abandoned is not available, it makes sense to add these indicator variables together rather than using separate probit regressions to examine the different dimensions.

First, part-time farmers are more likely to abandon land compared with full-time farmers (Baldock et al., 1996). Part-time farming participation can be used as their first preference indicator function for land abandonment. This indicator function takes the form: $I_1 = 1$ if the farmer is a part-time farmer or $I_1 = 0$ if the farmer is a full-time farmer.

The second preference indicator function is related to active participation in commonage farming. It is assumed that farmers are less likely to abandon their commonage land if they are an active commonage shareholder. Active commonage shareholders utilise the commonage more effectively for grazing and/or feed production and are less vulnerable to the risk of land abandonment. In this case, the preference indicator function takes the form: $I_2 = 1$ if the farmer is an inactive commonage shareholder. We can also add the third preference to indicate a function based on abandonment of haymaking. The third preference indicator function takes, $I_3 = 1$ if the farmer abandons haymaking that led to land abandonment. Combining these three observed preference indicator functions; $Y = I_1 + I_2 + I_3$ provide a better surrogate for land abandonment (Y^*). Land abandonment decisions on these different livestock practices are mutually dependent. The actual interpretation of the categorical values may be as follows:

Y= 0 if {if the farmer is a full-time farmer, active commonage shareholder and abandons haymaking};
= 1 if {if the farmer abandons one of the farming practices };
= 2 if {if the farmer abandons two of the farming practices};
= 3 if {if the farmer abandons three farming practices};

Thus abandonment of practices can be placed according to the number of categorical responses of land abandonment. Then the ordered probit technique can use the observations on Y, which are a form of censored data on Y^*, to fit the parameter vector β. The combined preferences can be interpreted based on the following regression specification:

$Y=0 \quad \text{if } Y^* \leq 0$

$Y=1 \quad \text{if } 0 < Y^* \leq \mu_1$

Y=2 if $\mu_1 < Y^* \leq \mu_2$

Y=3 if $\mu_2 < Y^* \leq \mu_3$

Where μ_1, μ_2 and μ_3 are unknown threshold parameters and cut points to be estimated along with β coefficient. It is assumed that the parameter β is fixed within a particular livestock grazing regime and property right but that this varies from one grazing regime to another and from one property right to another. Thus it is also assumed that land abandonment is heterogeneous across different livestock grazing management regimes because farmers face different market conditions and cost structures across different enterprises and products as a result of being exposed to different levels of abandonment. The inclusion of livestock management into the model is an important step to test the hypotheses as to whether the choice of farm management has influenced land abandonment or not.

This paper contributes to the literature by identifying the economic factors affecting the probability of land abandonment using an economic behavioural model, i.e. an Ordered Probit Model. It estimates the probability threshold values that are important in predicting future land abandonment in the region. These threshold values vary when one is predicting the future probability of abandonment at a household level. Mckelvey & Zavoina (1975) suggest that the interpretation of threshold values should be based on the Cumulative Normal Density Function (CNDF). The threshold parameters also help to estimate the future probability of abandonment.

2) $$\Pr ob[Y = j] = \Pr ob[\mu_{j-1} \leq Y^* \leq \mu_j]$$

3) $$\Pr(Y = j / X) = F(\mu_j - \beta' X) - F(\mu_{j-1} - \beta' X)$$

Where F(ε) is the CNDF function of ε. Calculation of these probabilities allows a better understanding of the process and factors that determine the future probability of land abandonment in western Ireland.

2.4 Data

The study is located in Connemara, County Galway and County Mayo. The landscape of southern Connemara is low-lying and composed of large expanses of western blanket bog. The soils of the upland grazing areas are generally of low productivity and are best suited to extensive cattle and sheep production. No arable farming occurs in the study areas.

Two data sources were used for this chapter. Data were drawn from the official list of Commonage farmers (CSO, 2002). The list includes 555 households registered as commonage shareholders. These farms are also in receipt of farm financial support. In the spring and summer of 2004, a total of 282 farms were identified as operating management regimes considered typical of commonage farmland. All of the farms were asked to participate in the survey and 282 agreed to take part in the analysis. The collected data set includes livestock cost, labour, farm participation, household characteristics, and sheep and cattle stocking on commonage as well as private land.

In addition, for the second dataset, personal interview with a sample of 100 farmers were undertaken by staff from the National University of Ireland, Galway (NUIG) at the owner's property. Each interview lasted approximately 45 minutes and followed a standard format. The questionnaire was piloted for one month during October 2010 and this aided the design of the survey. Each survey provided detailed data on revenue and cost summaries, farm premia, use of technology, labour and costs of farm operations, particularly grazing and livestock activities. The survey focused principally on market costs and benefits. Information on land abandonment and on part-time versus on farm labour activities were collected. Current and past land management practices including mixed grazing and haymaking were also documented. The range of enterprises on these farms included sheep, beef and suckler cow production.

All of the farmers surveyed had commonage land and additional private land. Exactly 25% of farms were in suckler beef enterprises. Livestock grazing management in the sheep enterprise consisted of ewes, rams, hogget and lambs. Most sheep farmers produce lambs for export for the Mediterranean market. About 28% of livestock farms in the region were involved with sheep enterprises. The remained were in mixed farming. About 60% of the farmers were from disadvantaged and less favoured areas. Approximately, 68% of the observations participated in REPS. The sample consists of 70% livestock farmers in Co. Galway and the rest in Co. Mayo.

2.5 Results

2.5.1 Descriptive statistics results

Table 1 reports the descriptive statistics of variables that may affect the probability of land abandonment in different livestock enterprises. Close inspection of Table 1 indicates that the majority of farmers (46%) are in a mixed grazing system. Only 30% of the farmers have specialised in sheep grazing and the remaining run a cattle grazing system. A number of variables are thought to be associated with land abandonment. These include proportion of part-time farming, inactive commonage shareholders and abandonment of haymaking in the west of Ireland.

Descriptive statistics results indicate that 45% of all livestock farmers were involved in part-time farming. The percentage of part-time farming involvement was significantly higher in suckler beef enterprises than sheep/mixed enterprises. More than 89% of farmers were active commonage shareholders and used their commonage for grazing and livestock production. The percentage of inactive commonage shareholders was much higher in suckler beef enterprises than in sheep/mixed enterprises. The survey results indicate that 47% of farmers produce hay for livestock feed. On average, farm income covers only 26% of total income. The results indicate that, off-farm income constitutes 36% of total income and subsidies contribute to 38% of income when other incomes are taken into account.

Income statistics reported here are given in relative terms as a percentage of total income. Stocking rate is relatively high for mixed grazing systems compared with specialised enterprises.

Table 1. Descriptive statistics by livestock enterprises

Variables	Livestock enterprise			
	All	**Suckler beef**	**Sheep**	**Mixed (Cattle and sheep)**
Percentage of farmers	100	24	30	46
Percentage of part-time farmers	45	58	45	39
Percentage of inactive commonage shareholders	11	24	12	4
Percentage of feed producers	47	53	26	57
Stocking rate on commonage land (TLU/ha)	0.76	0.20	0.70	1.0
Stocking rate On private land (TLU/ha)	1.05	0.70	0.97	1.27
Farm labour	38	29	37	43
Livestock income (%)	26.4	24.6	25.9	25.4
Off-farm income (%)	36.3	38.7	37.1	34.6
Subsidy (%)	38.3	36.7	37.0	40.0
Livestock cost (Euro per TLU per ha)	9.3	23.2	4.0	5.4
Average farmer age	57	57	56	57
Education	1.5	1.5	1.6	1.4

In mixed livestock systems, the survey results indicate that average livestock production occurs at a stocking rate of 1.00 total livestock units (TLU)/ha and 1.27 TLU/ha in commonage and private lands, respectively. Similarly, the average stocking rate for specialised sheep enterprises was 0.70 TLU/ ha and 0.97 TLU/ha in commonage and private lands, respectively. Stocking rates, particularly in suckler beef enterprises were relatively low at 0.20 TLU/ha for commonage lands. The results indicate that land abandonment is high for beef enterprises because of high livestock production costs prevailing in beef enterprises (i.e. which is five times greater than sheep and mixed enterprises).

2.5.2 Results of the ordered probit regression

The results of the ordered probit regressions are presented in Table 2. The values of the pseudo - R^2 and χ^2 critical values of goodness-of-fit measures indicate that the estimated ordered probit regression model exhibits a good fit. The regression produces consistent coefficients across livestock enterprises and the pooled regression tells a consistent story of land abandonment in Ireland. The results indicate that the probability of land abandonment is found to be positively correlated with off-farm income. Higher off-farm income has significantly increased the probability of land abandonment in all livestock enterprise regressions. The evidence suggests that land abandonment takes place primarily where there is a strong off-farm labour market. Thus farmers that abandon their land frequently allocate more time to off-farm activities which leaves less time and human resources for labour-intensive livestock farming such as haymaking and mowing with a consequent increase in land abandonment.

In the region studied farmers' income is highly dependent on external supports such as direct payments through the Single Farm Payment Scheme (SFPS) as well as REPS payments. In addition, payments have been allocated through the Disadvantaged Area Payment Scheme. The ordered probit results in Table 2 show that agri-environmental support payments significantly reduce the probability of land abandonment in all livestock grazing regimes. Subsidies are negatively and significantly correlated to the probability of land abandonment. A possible explanation is that government payments increase the viability of the farm and reduce the risk of land abandonment. In Ireland, Agri-Environmental Schemes (AES) are often considered as a solution to land abandonment and a means to keep farmers in farming. An expectation of continued support payments in the future is also predicted to be the most influential factor in reducing the future probability of land abandonment in the region.

Table 2. An Ordered Probit regression of land abandonment by livestock enterprises

Variables description	All sample		Suckler beef		Sheep		Mixed livestock	
	Coefficient	**Std. err**	**Coefficient**	**Std. err**	**Coefficient**	**Std. err**	**Coefficient**	**Std. err**
Location (Galway=1 Mayo=0)	0.2535	0.2213	-2.0207*	1.1528	0.8584**	0.3815	0.1140	0.3621
Disadvantaged area (Yes=1; No=0)	-0.1844	0.1825	-0.2314	0.4266	0.0658	0.3651	-1.0403***	0.3804
Proportion of active commonage shareholders	-0.3220*	0.2369	-0.7895*	0.5092	-0.4135	0.4403	0.0017	0.4570
Land size	0.0009	0.0011	0.0027	0.0076	0.0028*	0.0016	-0.0017	0.0025
Improved grazing land	-0.1798*	0.0993	-0.5187**	0.2129	-0.3211*	0.1922	0.2668	0.2146
Cattle stock	0.0142*	0.0090	0.0327**	0.0153	-	-	0.0047	0.0236
Sheep stock	-0.0088*	0.0050	-	-	-0.0070	0.0078	-0.0097	0.0098
Farm labour	-0.0398***	0.0062	-0.0500***	0.0165	-0.0199**	0.0100	-0.0856***	0.0167
Livestock income	-0.0121***	0.0040	-0.0057	0.0084	-0.0153*	0.0086	-0.0085	0.0077
Off-farm income	0.0121***	0.0033	0.0179**	0.0070	0.0163**	0.0065	0.0147**	0.0063
Subsidies	-0.00014***	4.34E-05	-0.00016**	8.06E-05	-0.00018**	8.9E-05	-1.9E-05**	9.98E-05
Livestock cost	-0.0017	0.0050	-0.0078	0.0074	0.0452	0.0248	0.0053	0.0208
Age	-0.0140*	0.0077	-0.0239*	0.0183	-0.0109	0.0136	-0.0203*	0.0139
Education	0.5214***	0.1607	0.5976*	0.3468	0.1006	0.3072	0.8056**	0.3362
Thresholds points								
Cut1	-2.8440	0.8336	-6.1593	2.1563	-2.2156	1.5214	-3.3972	1.6195
Cut2	-0.3701	0.8130	-3.3250	2.0166	0.3287	1.5036	-0.0829	1.5520
Number of obs.	274		66		81		127	
Pseudo R^2	0.3977		0.3746		0.3974		0.5685	
LR chi2	194.65		46.25		58.05		114.66	

Notes: Level of significance: ***=p<1%, **=p<5%, *=p<10%.

The role of farm labour with regard to the effectiveness of reducing land abandonment has not been well addressed in previous studies. A key finding in this study is that off-farm income leads to greater part-time farming. Off-farm employment plays a key role in increasing land abandonment in the uplands. The reduction in labour time devoted to traditional livestock farming practices is a threat to land abandonment.

Of all the variables considered, farm labour is the most significant determinant affecting the probability of land abandonment (Table 2). The results show that the probability of land abandonment is found to be negatively correlated with farm labour. Family labour significantly reduces the probability of land abandonment in all livestock grazing regimes except for mixed livestock.

The ordered probit regression results also show that increasing livestock income has significantly reduced the probability of land abandonment but only in sheep farming. Livestock income has no significant impact on the probability of abandonment in suckler beef and mixed livestock enterprises. Low cattle prices prevailing in the primary livestock markets[3] may be one of the reasons for a low impact of livestock income in reducing land abandonment in suckler beef and mixed farms. Increases in livestock income should reduce the supply of labour to the off-farm labour market and reduce land abandonment. Low livestock prices are a critical factor thought to influence land abandonment in the study region. Thus we speculate that low livestock incomes may provide some evidence for high land abandonment in suckler beef enterprises.

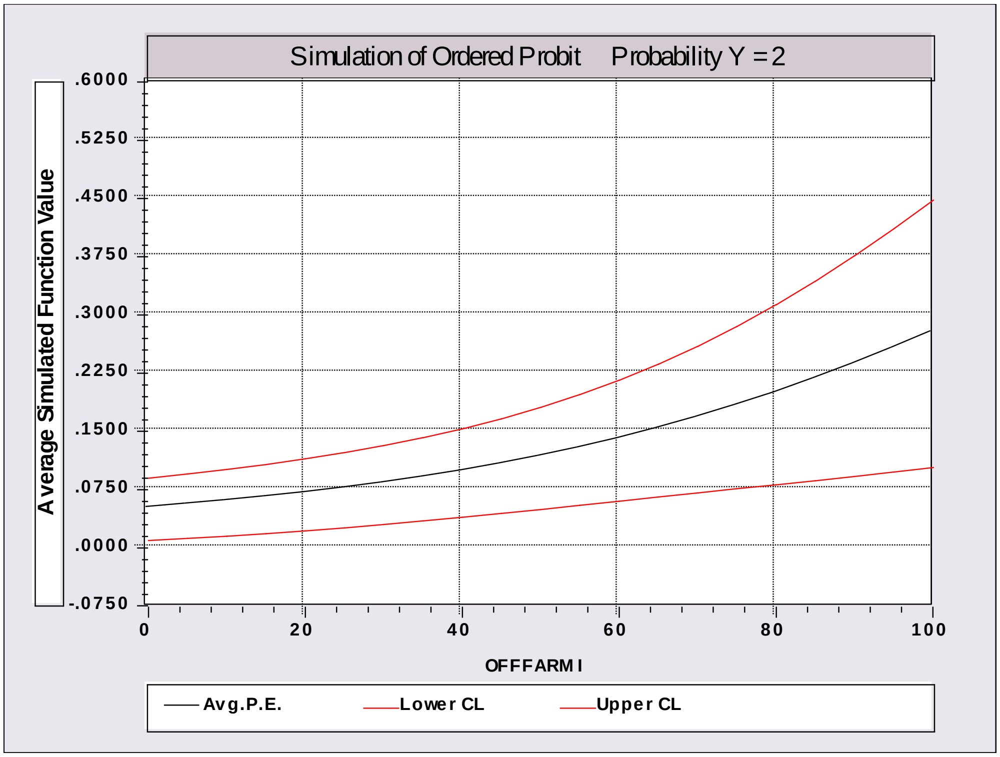
It is of interest to note the difference between the commonage resource and private farm land in terms of land abandonment and land-use behaviour of farmers. The pooled regression shows that the extent of land abandonment declines as the proportion of active commonage shareholders increases. There are many

[3] Primary markets are markets where producers strongly dominate to sell live animals to exporters and traders at market centres located in rural areas.

commonages in the region where substantial amounts of land were held by only a few users and in such cases the commonage resource is under-utilized. This may lead to under-grazing of commonage. On the other hand, on private land, the extent of land abandonment declines as the proportion of improved grassland increases.

A sensitivity analysis was conducted to examine how a change in the predicted probability of land abandonment alters as a result of increases in the percentage of off-farm income. Figure 1 shows a two-way graph of the predicted probability of land abandonment on the percentage of off-farm income. Off-farm activities usually compete for farming time and there is a trade-off between household time spent on off-farm and on-farm activities. The figure shows that the predicted probability of land abandonment increases sharply as the percentage of off-farm income increases. If the predicted probability of land abandonment is constant the shape of the graph would have been horizontal.

Figure 1. Simulation result of the probability of land abandonment and off-farm income

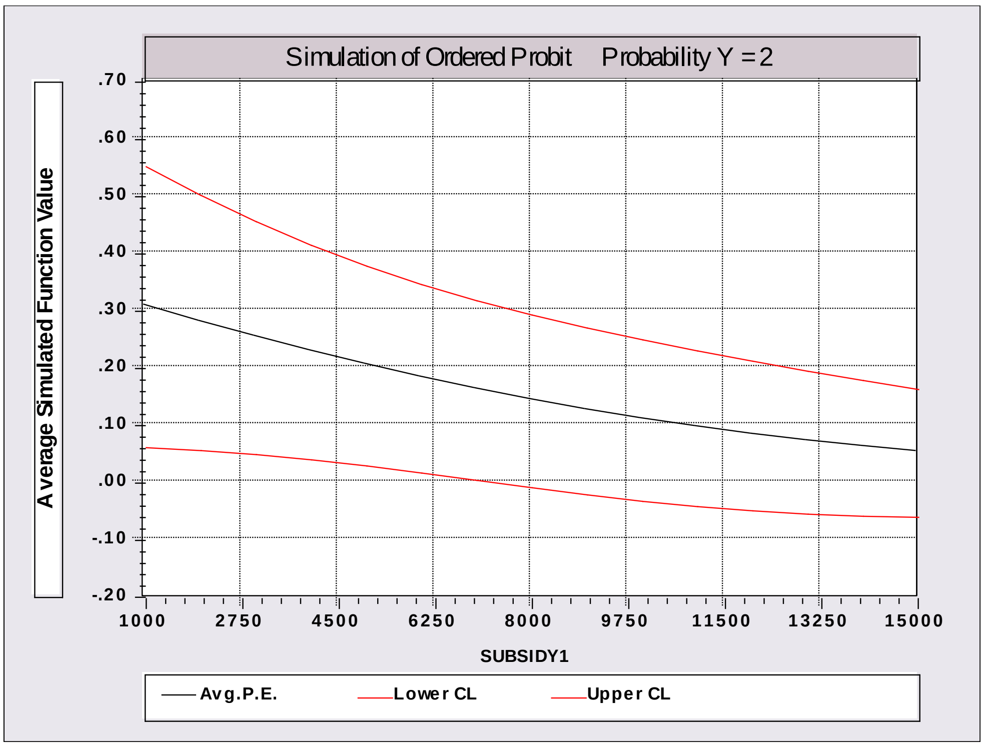


The availability of off-farm employment is related to higher opportunity costs of farm labour which may be another reason for the high cost of production and land abandonment in the regions studied. In areas where land abandonment is high, the opportunity cost of labour can increase in the next best alternative farming activity and as a consequence the predicted probability of land abandonment curve becomes concave. Favorable opportunities for labour outside the livestock sector do not only increase the cost of labour but also create a high cost of livestock production and further land abandonment.

Figure 2 also shows the simulation results of the predicted probability of land abandonment as the government payment increases. The graph indicates that the probability of land abandonment sharply declines with increase in farm subsidy.

The break-even point (the point where there is no land abandonment and the predicted probability of land abandonment is zero) will be when the government subsidy amounts to 8000 euros per hectare per year.

Figure 2. Simulation result of probability of land abandonment and subsidies



The main difference between the ordered probit regressions presented in Table 3 and the previous livestock enterprise regressions is mainly concerned with the livestock cost variable. Livestock costs have no significant impact on land abandonment in the livestock enterprise. On the other hand, the ordered probit regressions in Table 3 reveal that livestock costs were positively correlated with the probability of land abandonment. Livestock costs significantly increase the probability of land abandonment. High livestock costs of production reduce the profitability and competitiveness of the livestock enterprise. Feed costs appear to be the main component of livestock production costs in the region. The main source of feed is identified to be purchased feed from external sources. Increasing the cost of production, particularly feed costs have rendered traditional herding systems economically uncompetitive. Once haymaking is abandoned farmers have to substitute with purchased feed and this gives rise to a substantial increase in overall financial costs. Thus abandonment of haymaking and the high feed cost may be the main driving force for land abandonment in the region.

Table 3. An Ordered Probit regression of land abandonment by land property rights

Variables description	All lands		Commonage lands		Private lands	
	Coefficient	**Std. err**	**Coefficient**	**Std. err**	**Coefficient**	**Std. err**
Location (Galway=1 Mayo=0)	0.4649***	0.1529	0.6139***	0.2276	0.4566**	0.2189
Disadvantaged area (Yes=1; No=0)	-0.1328	0.1275	-0.2090	0.1901	-0.1093	0.1789
Proportion of active commonage shareholders	-0.2519	0.1640	-0.1702	0.2388	-0.3097	0.2340
Land size	2.76E-05	0.0001	0.0001	0.0002	0.0021	0.0025
Improved grazing land	-0.0699	0.0788	0.3154	0.2365	-0.1686*	0.0973
Cattle stock	0.0017	0.0089	-0.1216**	0.0493	0.0100	0.0112
Sheep stock	-0.0140***	0.0050	-0.0229***	0.0069	-0.0050	0.0078
Farm labour	-0.0471***	0.0044	-0.0465***	0.0064	-0.0484***	0.0062
Livestock income	-0.0141***	0.0027	-0.0148***	0.0040	-0.0134***	0.0039
Off-farm income	0.0089***	0.0021	0.0105***	0.0031	0.0092***	0.0030
Subsidies	-0.00013***	2.79E-05	-9.7E-05**	4.19E-05	-0.00014***	3.93E-05
Livestock cost	5.98E-05***	1.54E-05	7.48E-05***	2.06E-05	4.41E-05*	2.55E-05
Age	-0.0224***	0.0047	-0.0209***	0.0068	-0.0227***	0.0067
Education	0.0990	0.0835	0.0951	0.1264	0.1154	0.1169
Thresholds point						
Number of landscape (obs.)	548		274		274	
Pseudo R^2	0.3716		0.4009		0.3725	
LR chi2	363.82		196.25		182.33	

Notes: Level of significance: ***=p<1%, **=p<5%, *=p<10%

The results of the analysis show that farm size has increased the probability of land abandonment only for sheep enterprises. Larger land holdings with greater income potential have possibly increased the chances of a farmer remaining in farming compared to small farms. Thus controlling for land property rights may be an important factor in reducing the correlation between land size and land abandonment. Determinants of land abandonment vary from location to location. Many of the suckler beef cattle enterprises are predominantly located in remote areas. The impact of access and availability to off farm employment opportunities has been captured by the inclusion of location variables. Location has significantly influenced land abandonment for both the enterprises and the commonage regressions. The availability of infrastructure serves as a proxy for different market outlets that can influence land abandonment. The results indicate that the probability of land abandonment is higher in Co. Galway than Co. Mayo (Table 3). The results also show that the probability of land abandonment was lower in the suckler beef enterprises in Co. Galway whereas it was higher in sheep enterprises in Co. Mayo.

There are two important categories of grazing lands in the Irish uplands: rough grazing and improved grasslands. Improved grassland makes up the largest proportion of Irish private productive farmlands. Although rough grazing lands are dominant in Irish commonage, there is limited variation in the quality or condition of grazing land. The probability of land abandonment was evaluated for improved grazing lands as compared to rough grazing. The regression analysis results suggest that improved private grasslands have less probability of being abandoned than commonages. However, the condition of grazing land has no significant impact on land abandonment in commonages whereas it does have a significant impact on private land. Most of the Irish uplands are commonages. In terms of reducing land abandonment, the regression results indicate that improved grazing lands are more important for specialized livestock enterprises than for mixed enterprises.

Farmer’s age is negatively and significantly correlated with land abandonment in private land and commonage. Land abandonment decreases as farmers’ age increases. Young farmers are more likely to abandon commonage lands than older farmers. It appears that fewer young people are moving into farming partly due to low economic returns. However, farmers’ age was not found to be significantly associated with abandonment in sheep farming.

Education appears to have less explanatory power than age as a reason for land abandonment. Older farmers often have little education and few opportunities to work off-farm, which forces them to maintain livestock farming in marginal lands. Education was positively and significantly correlated with land abandonment almost in all livestock enterprises. Education increases the probability of land abandonment indicating more educated farmers abandon livestock farms and join off-farm labour activities. However, education was not a significant factor affecting land abandonment in private and commonage property resource management. In terms of reducing land abandonment, educational attainment may be of more practical importance in livestock enterprise management than commonage property resource management.

Table 4 shows the results of the ordered probit regressions based on the survey data collected from 100 farm households in the west of Ireland in 2009. New explanatory variables were added to the regression including: biodiversity and mixed grazing. The results of the ordered probit regressions indicate that mixed grazing systems have significantly reduced the probability of abandonment both in private and commonage lands. This shows that mixed livestock management was less vulnerable to the risk of land abandonment. It is also worth noting that the probability regression results indicate that mixed grazing reduces the risk of land abandonment more in disadvantaged areas than other areas where land abandonment predominates. This strengthens the case for mixed grazing management as a means to reduce land abandonment.

Table 4. An Ordered Probit regression of land abandonment and habitat diversity

Variables description	All lands		Commonage lands		Private lands	
	Coefficient	**Std. err**	**Coefficient**	**Std. err**	**Coefficient**	**Std. err**
Location (Galway=1 Mayo=0)	0.7407***	0.2159	0.9412***	0.3583	0.6494**	0.2782
Mixed grazing (Yes=1; No=0)	-0.5782***	0.2038	-0.6442**	0.3414	-0.5414**	0.2737
Biodiversity	-0.1099	0.3798	-0.0690	0.4633	-0.0996	1.9397
Stocking rate	0.1681**	0.0718	0.2017**	0.0901	0.2061*	0.1526
Farm labour	-0.0084**	0.0036	-0.0143**	0.0060	-0.0061*	0.0047
Land size	0.0051**	0.0023	0.0085***	0.0031	0.0062	0.0120
Off-farm income	0.0124***	0.0034	0.0118**	0.0053	0.0130***	0.0045
Subsidies	-7E-05***	1.93E-05	-9.3E-05***	3.28E-05	-6.6E-05**	2.62E-05
Livestock cost	0.0003***	0.0001	0.0005**	0.0002	0.0002*	0.0001
Distance to main town	0.0089*	0.0055	0.0117*	0.0084	0.0094	0.0077
Age	-0.0073	0.0081	-0.0068	0.0130	-0.0086	0.0111
Education	0.0888	0.1609	0.0983	0.2648	0.0891	0.2095
Number of obs.		175		77		98
Pseudo R^2		0.2133		0.2629		0.1928
LR chi2		90.25		50.42		45.34

Notes: Level of significance:*=p<1%, **=p<5%, *=p<10%**

Habitat diversity was measured based on a Shannon Weaver Index (SWI) and was used as an indicator of biodiversity. Biodiversity was found to be negatively correlated with the probability of land abandonment (Table 4). Seemingly, biodiversity reduces the probability of land abandonment; however, the correlation was not significant. On the other hand, stocking rate was found to be positively and significantly correlated with the probability of land abandonment. Adjusting grazing pressure seasonally and carefully mixing different livestock species with different livestock age groups can reduce the risk of land abandonment which is also an important controlling factor for biodiversity. It appears that the impact of stocking rate on land abandonment depends on livestock species in the Irish uplands (Table 3). Increasing sheep stocking rate has significantly reduced land abandonment in commonage farms. Not only grazing management but also property rights are crucial factors in determining land abandonment in the study regions. Moderate stocking densities create a diverse habitat which is suitable for many plant species as well as low levels of abandonment.

2.5.3 Land abandonment thresholds

Land abandonment does not create a negative impact on biodiversity until a certain minimum threshold of labour input is crossed. For instance, land abandonment abruptly changes as farmers move from full-time to part-time farming. Threshold parameters can be estimated using the maximum likelihood estimation technique and the ordered probit model. Critical threshold probabilities are cut off points that show the transition and a gradual shift from farming to land abandonment. Threshold values can be defined as boundary parameters that separate farming continuity from land abandonment.

The next step is to estimate the predictive probabilities of land abandonment within specific livestock enterprise. The predicted probability of land abandonment is presented in Table 5. The results show that the average predicted probability of land abandonment rate is 7% in the region. The highest land abandonment rate occurs in the suckler beef farming which is 12%. This probability of land abandonment rate may be high because of high production costs and unfavorable market prices prevailing for suckler beef enterprises. The lowest land abandonment rate occurs in the mixed livestock regime only 4%. The predicated probability of continuing farming was high in mixed grazing regime with a figure of 61% and low in the suckler beef livestock regime (30%).

Table 5. The probability of land abandonment

Grazing system/ Land tenure	The probability of continue fulltime farming: Pr(y=0)	The probability of part-time Farming/ inactive in the commonage Pr(y=1)	The probability of land abandonment: Pr(y>2)
All livestock enterprises	**50%**	**43%**	**7%**
Suckler beef	**30%**	**58%**	**12%**
Sheep	**51%**	**42%**	**7%**
Mixed livestock	**61%**	**35%**	**4%**

The next chapter addresses the issue of habitat fragmentation and biodiversity.

Chapter 3. The impact of habitat fragmentation on biodiversity and livestock productivity in Ireland.

3.1 Introduction

Habitat fragmentation is the process whereby habitat loss results in the division of large and continuous habitats into smaller and isolated habitat fragments (Ranta et al., 1998; Franklin, et al., 2002; Opdam & Wiens, 2002). A number of reviews have investigated habitat fragmentation related topics. Most of these have dealt with specific sub-components of the enormous body of literature on landscape modification and habitat fragmentation such as the amount of native vegetation cover in relation to birds and mammals (Andrén, 1994), the relative effects of habitat loss and habitat sub-division (Fahrig, 2003), the history of fragmentation research (Haila, 2002), metapopulation dynamics (Hanski 1998), and edge effects (Ries et al., 2004).

A decline in habitat is thought to be one of the most significant causes of the loss in terrestrial biodiversity and reduced genetic resources (Wilcox & Murphy, 1985; Woodley et al., 1993; Wiens, 1995; Davis & Margules, 1998; Wilson, 1997; Chapin, 2000; Dirzo & Raven, 2003; Fahrig, 2003). There is a substantial literature on the effects of habitat fragmentation on biodiversity (Turner, 1989: Ranta et al., 1998; Opdam & Wiens, 2002). Most of which deals with the loss of biological habitat such as an opening up of closed canopy forest (Burgess & Sharpe, 1981; Ranta et al, 1998; Diaz et al., 2000) which becomes fragmented and decreases in size and value from an ecological point of view.

However, much of the world's biodiversity is associated with human dominated ecosystems and is not confined to natural ecosystems where human influence is limited (Pimmental et al., 1992). Globally, it is estimated that 38% of land is used for agriculture (FAO, 2004). In Ireland, more than 90% of the grass land is used for grazing and livestock production. The conservation of biodiversity on agricultural land has therefore become an important issue for policy makers and a number of studies indicate that biodiversity conservation must focus on managed ecosystems (Miller, 1996, Reid, 1996; Daily et al., 2001; Rosenzweig, 2003; Polasky et al., 2005; Harrop, 2007).

Studies on the effects of habitat fragmentation in an agricultural context are limited and we are not aware of any studies that investigate the effects of habitat fragmentation on biodiversity and livestock productivity in the context of common and private property. Several studies suggest that research on the biodiversity – productivity relationship in the context of livestock systems represents a useful area of enquiry in order to protect biodiversity, maintain productivity and safeguard ecosystem health (Grant, 1985; Bengtsson, 1998; Collins et al.; 1998; Midmore et al., 1998; O'Neill et al., 1999a; Tilman et al., 1999; Soder et al., 2007). The present paper, therefore, attempts to analyse the impact of habitat fragmentation on biodiversity and livestock productivity through the process of habitat fragmentation across different livestock enterprises.

The main research goal addressed in this study is to evaluate the impacts of habitat fragmentation on biodiversity and livestock productivity under different livestock management regimes and land property rights. The specific objective is to assess the impacts of agri-environmental subsidies on biodiversity and livestock productivity.

The structure of the paper precedes as follows; first a detailed background on biodiversity and the livestock productivity relationship is provided, next the effects of habitat fragmentation on biodiversity and impacts of mixed grazing on biodiversity is discussed; next the methodological approach is conveyed; third empirical results are discussed and finally conclusions and recommendations are drawn.

3.2 Background

3.2.1 Biodiversity-productivity relationship

In what follows, a background on the interrelationship between biodiversity and livestock productivity, the impact of habitat fragmentation on biodiversity and other factors that affect biodiversity in prevailing property regimes is given. Biodiversity provides a wide range of direct and indirect benefits that are essential for human well-being. Improving productivity and efficiency of ecosystem services is an indirect function provided by biodiversity.

The literature indicates that biological diversity is important from an economic perspective because it increases ecosystem productivity (Di Falco & Perrings, 2003:2005; Di Falco & Chavas, 2006). There is also a significant body of literature on the relationship between biodiversity and primary productivity (Grime 1979; Rosenzweig & Abramsky, 1993; Naeem et al., 1994; Tilman et al., 1994; Tilman & Downing; 1994; Abrams, 1995; Tilman et al., 1997; Tilman et al, 1999; Hector et al., 1998; Sala et al., 2000; Loreau et al., 2001; Sanderson et al., 2009; Soder et al., 2007).

Much of the literature, Naeem et al., (1994); Tilman et al, (1997); Girme, (2001); Sanderson et al., (2004); Soder et al., (2007) predicts that biological diversity and productivity are positively correlated. Total community biomass may increase with biological diversity (Lehman & Tilman, 2000). Some recent studies show a positive linear relationship between biodiversity and productivity in grassland communities (Bai et al., 2007). Other studies argue that the biodiversity-productivity relationship could be negative (Rosenzweig, 1971; Proulx & Mazmunder, 1998; Kondoh, 2001). However, Huston, (2000) shows that there is no consistent effect of plant diversity on productivity.

Grace, (1999) suggests that productivity is one of several factors influencing habitat diversity in grasslands. Tilman et al., (1996) examined the relationship between plant diversity and primary productivity and reported that the productivity of more diverse plots declined less and recovered more quickly after a severe drought than the productivity of less diverse plots. In addition, they found that a significant positive effect of diversity on the resistance of total plant biomass to drought and conclude that the preservation of biodiversity is important for the maintenance of productivity.

In European grassland systems, there is experimental evidence to suggest that maintaining high levels of plant species diversity increases grassland productivity (e.g. Tilman et al., 1999). Fagan et al., (2008) have observed that for restored grasslands on a range of soil types across southern England, species richness appears to have a positive effect of on ecosystem productivity. Costanza et al., (2007) have investigated the inter-dependence of net primary productivity and biodiversity at very broad spatial scales for eco-regions in North America. They found that over half the spatial variation in net productivity could be explained by patterns of biodiversity. Positive diversity-productivity relationships have been observed in a number of terrestrial systems at local scales. Lawton et al. (1998) have also provided evidence to support the existence of a direct positive relationship between biodiversity and productivity.

Many ecological studies have focused on the relationship between biodiversity and productivity with biodiversity as a dependent variable (Grime, 1973; Lawton & Brown, 1993; Tilman & Pacala, 1993; Tilman & Downing, 1994; Abrams, 1995; Waide et al., 1999; Huston, 2000). The causality relationship may focus on productivity and its response to biological diversity where productivity is a dependent variable and biological diversity is an independent variable (Tilman, 1996; Hector, et al., 1999). Worm & Duffy, (2003) emphasize that these relationships are often bi-directional, such that changes in biodiversity can be both a cause and a consequence of changes in productivity and stability. Abrams, (1995) suggests that there exists an intermediate level of productivity corresponding to maximum diversity.

There are three possible functional forms for the relationship between biological diversity and productivity: linear; asymptotic and no-relationship (Waide et al., 1999). Grime, (1973) proposed a humpbacked-shape relationship between biodiversity and productivity. A humpbacked-shape relationship means that biodiversity increases with productivity at a lower level of productivity and decreases at a higher level. MacArthur & Pianka, (1966) suggested that the relationship between biodiversity and productivity could be linear. The rivet-redundancy hypothesis states that a non-linear relationship may occur between biodiversity and ecosystem function (Lawton & Brown, 1993) and productivity is one of the main ecosystem functions. Similarly, the relationship between livestock grazing and biodiversity is non-linear (Olff & Ritchie, 1998). Tilman & Downing (1994) found a logistic-like curve for the relationship between species diversity and total biomass.

Di Falco et al., (2006) shows that diversity can help maintain productivity and reduce yield variability and farmers' exposure to production risk. Di Falco & Chavas, (2006) also show that higher crop diversity supports system resilience and maintains productivity under challenging climatic conditions. They concluded that crop diversity can help reduce the risk exposure from crop failure in low rainfall areas of the Ethiopian highlands. In a latter study, Di Falco et al., (2010a) find that increasing the number of crop varieties increases production and this result is stronger when rainfall levels are low. These findings suggest that diversity may constitute an insurance mechanism for farmers facing production failure due to climate change. Benin et al., (2004) find that households with more labour, livestock and farm size grow a greater diversity of crops.

Biological diversity may stimulate resilience and improve the ability of an ecological system to absorb disturbances (Holling, 1973; Tilman et al., 1996, Folke et al., 1996). Resilience is a measure of the capacity of the system to absorb stress and shocks without losing its 'self-organisation' (Holling, 1973). The loss of biodiversity will diminish the capacity of an ecosystem to provide a sustainable supply of essential goods and services (Tilman et al., 1994). Biodiversity loss affects the resilience of the system and can have adverse effects on the functioning and productivity of ecosystem services (Tilman & Downing, 1994). In addition, the economic value of biodiversity has an insurance value that is related to its role in protecting and regulating the resilience and productivity of ecosystems over a range of environmental conditions (Perrings et al., 1995; Yachi & Loreau, 1999). In many cases, the reduction in the "insurance value" of biodiversity is not signalled in the incentive structure of the society including the price mechanism (Folke et al., 1996).

Naeem et al., (1995) provide the first evidence that ecosystem processes may be affected by loss of diversity. Naeem et al., (1994) indicates that declining biodiversity can alter the performance of an ecosystem. Loss of biodiversity reduces ecosystem resilience and hence affects the foundation for economic activity (Perrings et al., 1995). A low level of ecosystem resilience can cause a sudden decrease in biological productivity (Arrow et al., 1995). In a low resilience ecosystem, the internal cycling of nutrients are affected and reduced. As a result, primary production becomes more dependent on external inputs.

Various studies have analysed the contribution of crop diversity to the mean and variance of agricultural yield and farm income (Smale et al., 1998; Scälpfer et al., 2002; Di Falco & Perrings, 2003:2005). Biodiversity increases productivity through an insurance mechanism (Perrings et al., 1995; Scälpfer et al., 2002; Brock & Xepapadeas, 2003). Di Falco & Chavas, (2006) show how crop genetic diversity increases farm productivity and reduces risk exposure. Diversity of grass species can reduce production variability and insurance costs (Schläpfer et al., 2002). Di Falco & Perrings, (2005) also find that risk aversion is an important driving force for biodiversity conservation and risk averse farmers use crop diversity in order to hedge their production and income risk.

3.2.2 Impact of habitat fragmentation on biodiversity

This study focuses principally on habitat fragmentation not on land fragmentation issue. Habitat fragmentation differs from land fragmentation[4] (see the second section for a detailed discussion on land fragmentation). Andrén, (199) argue habitat fragmentation implies the division of a larger area into smaller pieces and for habitat it means a decrease in fragment size and an

[4] Land fragmentation can be defined as follows: a state of division of holdings into discrete parcels that are dispersed over an area (King & Burton, 1982; Blarel et al., 1992; van Dijk, 2003; Sabastes-Wheeler, 2002; Todorova and Lulcheva, 2005; Niroula & Thapa, 2007).

increasing degree of isolation between fragments. Habitat loss on the other hand is simply a decline in a certain habitat in the landscape. Boundary characteristics and land use types have shown strong variability in habitat diversity at a patch level (Dauber et al., 2003). The total length of parcel borders increases with fragmentation (van Dijk, 2003).

In this present study, habitat fragmentation is defined as the loss of original habitats, a reduction in average patch size, an increase in the number of plots, and an increase in the isolation of habitats. Habitat fragmentation occurs on livestock farms that experience a reduction in average patch size (or total habitat area), an increase in the number of patches and an increase in the perimeter to area ratio. We use information on habitat diversity, patch size, the number of patches and the perimeter to area ratio on the farms surveyed.

A decline in original habitat, reduction in average patch size, increase in the number of plots and increase in perimeter to area ratio are all considered to contribute to a decline in biological diversity (Wilcox & Murphy, 1985). Habitat fragmentation typically reduces total habitat area, increases the number of patches and the amount of habitat edge (Fahrig, 2003). The high production of patches may provide higher quality food for longer periods.

As a landscape becomes progressively fragmented, a greater number of fragments of varying shapes and sizes are created (Baskent & Jordan, 1995) and these are scattered through a matrix of modified habitat (Opdam & Wiens, 2002). In fact, fragmentation only occurs when habitat loss reaches a point at which habitat continuity is broken (Opdam & Wiens, 2002) and this is quite clearly a landscape-level attribute that describes the size and spatial arrangement of remaining habitat (Baskent & Jordan, 1995). Fragmentation is not just a patch-level phenomenon, although this is the scale at which many of its biological impacts are observed.

Breaking the landscape into many small patches affects movement and persistence of the organism by controlling the redistribution of matter and nutrient cycling (Turner, 1989). Habitat loss can directly reduce supply of food, fuel, medicinal and genetic resources (Chapin, 2000). More fragmented landscapes contain more edge for a given amount of habitat, favours different species from the interior habitat and this may increase the probability of individuals leaving the habitats (Fahrig, 2003). As corridors are lost and habitat becomes disconnected, disturbance can cause local extinctions (O'Neill et al., 1988b). As the landscapes are more fragmented, the risk of losing plant species increases (Woodley et al., 1993). The risk of extinction of habitat fragmentation may not be a linear function of the associated reduction in habitat and fragmented area (Wilcox & Murphy, 1985). Bascompte et al., (2002) predicts a negative effect of habitat loss on population growth rates.

Individual species are thought to have minimum patch size requirements (Diaz et al., 2000) and effects of patch size are the primary determinants of the number of species in a fragment. Burgess & Sharpe, (1981) indicate that patch size is positively correlated to habitat diversity. It is accepted that habitat patches of equal quality will usually have the same carrying capacity (Forman & Godron, 1986). As the distribution of patch sizes changes, the landscape becomes more hospitable to some species and less hospitable to others (Wiens & Milne, 1989). Habitat fragmentation affects biodiversity by reducing the amount of suitable habitat available (Fahrig, 2003). Reduction in the area of suitable habitat can result in population declines. The rarest species is the most likely to be the first species to disappear as the proportion of suitable habitat declines in the landscape (O'Neill et al., 1988b).

Many studies show that habitat fragmentation has a substantial negative impact on biodiversity (Saunders et al., 1991; Fahrig, 2003; Dirzo & Raven, 2003). Changes in spatial pattern of habitat fragmentation have been implicated in the decline of biological diversity and in the ability of the ecosystem to recover from disturbance (Folke et al., 1996). It is well documented habitat fragmentation of terrestrial habitats has negative effects on native biodiversity (Saunders et al., 1991; Andrén, H. 1997). Native species are lost from habitat fragments because of habitat changes, reduced immigration, and increase edge effects (Luoto et al., 2003 and Fischer & Lindenmayer, 2007).

Habitat fragmentation is currently ranked as the primary cause of species extinction (Pimm & Raven, 2000). Ecosystem resilience can also be degraded as a consequence of human activities through a diversity of drivers including overgrazing, habitat loss, species invasions and climate change (Folke et al., 1996; Baillie et al., 2004). Habitat fragmentation is widely viewed as an aspect of habitat degradation (Haila, 2002).

The most important and large-scale cause of changes in the degree of habitat fragmentation is the expansion of human land-use (Burgess & Sharpe, 1981) and anthropogenic habitat modification (Ewers & Didham, 2005). Human activities have contributed to the loss of biodiversity which threaten the stability and the continuity of an ecosystem as well as their provision of goods and services (Pimm et al., 1995, Nunes et al., 2001). Patchiness can also be associated with a change in composition of species in favour of woody plants, which may reduce economic productivity (Perrings & Walker, 1997).

Human impact through land-use change is projected to have the largest impact on biodiversity loss as a consequence of economic activity followed by climate change by the year 2100 (Chapin et al., 2000). Land-use and land management decisions have major impacts on ecosystem management and the goods and services they provide (Daily et al., 2009). Wilson et al., (2003) and Sala et al., (2000) find that there is a high correlation between human land-use intensity and loss of biodiversity. Since biodiversity of the terrestrial ecosystems are expected to be mainly affected by land-use change, researches have to focus on livestock production, land-use and sustainable land management for the preservation of biological diversity.

The Millennium Ecosystem Assessment (MEA), (2005) deals with economics of biodiversity loss and reports that livestock production as one of the principal drivers behind environmental degradation and depletion of ecosystem function and services. This argument suggests that there exists a trade-off relationship between increasing livestock productivity and biodiversity loss. Neglecting the environmental problems associated with livestock sector could threaten future level of livestock productivity and imposes serious environmental risk and biodiversity loss (Cassman & Wood, 2005). At the same time, researches argue that food production remains a primary function of agriculture. Livestock production is a viable source of future food and increased demand for livestock products including meat and milk. The future challenge will be how to maintain both biodiversity and livestock productivity in a sustainable way i.e., this needs balancing livestock production and biodiversity goals.

3.2.3 Land fragmentation

Land degradation is one part of habitat degradation. Many economic researches, Nguyen et al., (1996); Wan & Chang, (2001) Lerman, (2005); and Rahman & Rahman, (2009) find that land fragmentation has a significant detrimental effect on productivity and efficiency. Land fragmentation is often considered to be an obstacle for improving agricultural productivity and land abandonment (Theesfeld, 2005; Dirimanova, 2006; Di Falco et al., 2010a). van Dijk, (2003) suggests that most farmlands suffer from extreme fragmentation of land ownership. For crop biodiversity, Di Falco et al., (2009) find that land fragmentation reduces farm profitability and increases crop diversity. Jabarin & Epplin, (1994) indicated that land fragmentation induces inefficiency by increasing production cost. Land fragmentation can be seen to have economic cost in terms of lower agricultural productivity (Nguyen et al., 1996). Nguyen et al., (1996); Wan & Chang, (2001) and Lerman, (2005) find that land fragmentation has a negative impact on productivity. There exists a research gap in the impacts of habitat fragmentation on biological loss and livestock productivity as well as determining the best livestock management practices for biodiversity and livestock productivity.

Parikh and Shah (1994) report that land fragmentation gave rise to a fall in technical efficiency on farms. Hunsakar et al., (1990) show that patchiness may weaken productivity of ecosystem functioning. Fragmentation of land parcels produces a farm structure that prevents application of labour and fertilizer evenly to all plots of land and discourages use of land (Jacoby, 1970), thereby undermining efficient use of land (Coletta, 2000). However, Wu et al., (2005) indicate that land fragmentation does not have any significant impact on productivity.

There are a number of reasons why farmers may benefit from land fragmentation (Di Falco et al. 2010a). Land fragmentation is thought to promote crop and agricultural diversity (Bellon and Taylor, 1993; Hung et al., 2007). According to Lusho and Papa (1998), fragmented land parcels with diversity in biophysical conditions allow particularly, small farmers to grow a range of crops (Blarel et al., 1992). Land fragmentation provides a means of exploiting land parcels of differing quality. This facilitates crop diversification, spreads labour requirements, reduces production and price risks (Di Falco et al., 2010a) and better matches soil types with necessary food crops (Bentley, 1987; Blarel et al., 1992). By operating plots in different locations, farmers are able to reduce the variance of total output, because the scattering of plots reduces the risk of total loss from flood, drought, fire and other perils and allow the farmers to diversify their cropping mixtures across different growing conditions (Blarel et al, 1992). However, this benefit in terms of risk reduction should be compared with the cost in terms of possible loss of agricultural output, which may arise because of increased travelling time between fields and transport costs. Financial gain per unit of land is a function of cost and amount of production. The higher the cost of production, the lower is the profit.

Although there is some empirical evidence on how land parcel fragmentation reduces farm profitability and increases crop diversity (Di Falco et al. 2010a, 2010b), the impact of habitat fragmentation on livestock productivity and habitat biodiversity has not been analyzed before. In addition, the debate on land size and productivity has not yet been resolved (Johnston & Tomich, 1985; Niroula & Thapa, 2005:2007), the question arises as to whether more fragmented land can be considered as an indicator of production efficiency or not. A major problem that farmers in Ireland are confronted with is how to increase output per unit of land and per unit of input and preserve biodiversity. In this context, it is sensible to study the impact of habitat fragmentation on biodiversity and livestock productivity.

According to (Lusho and Papa, 1998), fragmented land parcels allow particularly small farmers to grow a range of crops or forage, increase diversification and ease seasonal labour bottlenecks (Blarel et al., 1992 and Di Falco et al., 2010a). This is true when few non-farming employment opportunities are available and farming operations are highly labour intensive. In many cases, the disadvantages of fragmented parcels far exceed their advantages. When land parcels are fragmented, the input use efficiency is reduced, resulting in increased cost of production, and reduced net return per unit of land and labour. The increased cost of production constrains agricultural development by weakening farmers' competitive capacity.

3.2.4 Impact of subsidies on biodiversity

One of the aims of this present study was to consider the impacts of agri-environmental subsidies on biodiversity and livestock productivity. A number of studies provide general arguments in support of using agri-environmental schemes to support biodiversity, rural amenities and stewardship (Hodge, 2000; Harvey, 2003; Wallis De Vries et al., 2007) Other work questions the efficiency of agricultural subsidy schemes for preserving biodiversity (Feehan et al., 2003). Murphy et al., (2011) investigated the effectiveness of REPS in supporting biodiversity focusing on management of biodiversity undertakings and farmers' behaviour. They find that maintaining water quality is most likely to be undertaken by suckler beef and dairy farmers, enhanced field margin options by full-time farmers whereas the creation of new habitats tends to be by married farmers. The authors also find that farmers with peatland are more likely to choose the enhanced field margin option.

Participation in AES depends on farmers' behavioural response and attitude (Wilson, 1997). Hynes et al., (2008) show that younger farmers are more likely to participate than older farmers. In environmentally sensitive areas, entry decisions have been found to be highly influenced by farm income (Hughes, 1994) and the probability of entry was increased where the scheme prescription fitted the farm situation and the cost of compliance were low (Wynn et al., 2001). Lynch & Lovell (2003); Wilson (1997) found that having a successor increases the probability of farmers decision to participate in AESs. Van

Rensburg et al., (2009) identify that sheep farmers are less likely to join REPS than suckler beef farmers and that being in receipt of other sources of government income acted as a deterrent to participation. Hynes & Garvey, (2009) found that low livestock productivity and poor soil fertility increases the probability of farmers' decision to participate in AES.

Hodge, (2000) argues that payments to farmers can represent the correction of market failure rather than distortion to trading relationships. In Ireland, REPS operates within the framework of 'management agreement model' (Hynes & Garvey, 2009). Harvey, (2003) studied the dependence of landscape services on subsidies to livestock production. Subsidies may be seen as payments for the production of countryside services that include recreation, amenity and environmental care for biodiversity (Harvey, 2003). Any change in the management of livestock production may impose direct and indirect cost on farmers in terms of reduced productivity and profit. The payments should reflect the cost of providing all services including the biodiversity. Thus the removal of production related subsidies clearly result in a reduction in biodiversity benefits and agricultural assets (Harvey, 2003).

In Ireland, agricultural production was intended to have a dual objective function: food production and biodiversity protection. At the same time, it focuses on improving competitiveness of agriculture and sustainability through appropriate land management. In this framework, recreation and tourism is also an important component of livelihood and income generation which contributes to quality of life for rural communities. The main effects of decoupling are to reduce stocking rates, change the mix of livestock activities and look for additional production activities that can be complimentary to biodiversity.

The viability of upland farms often depends on core subsidy support such as the SFPS and REPS payments. Decoupling of subsidy payments from specific livestock and grassland outputs is a necessary but not sufficient condition for achieving grasslands based environmental objective. In any case, removal of these payments would lead to negative farm income and land abandonment.

Livestock farming remains the dominant land-use in the in Irish uplands, even though it operates on the margins of profit and agricultural productivity.

Researchers indicate that it is the impact of livestock on biodiversity that led to the development of agri-environmental schemes. Agricultural activities also produce cultural landscapes and the associated wildlife that is valuable to the public. Environmental economists stress the aesthetic importance of pasture lands and the animals grazing on them. They also mention the social role of pasture management in maintaining cultural heritage and compensation for local animal breeds. Nilsson, (2009) reports that biodiversity restoration and conservation costs differ between geographical regions and financial support must be suited to local conditions.

Although it is difficult in terms of an economic point of view, people may find other means of paying the necessary costs of biodiversity provision in the future. By paying a premium for environmental friendly products, such as Connemara lamb, some people know that they are contributing to the preservation of the production method and the families that depend on these production systems (Harvey, 2003). Such contributions are an indication of public willingness to pay for the value of biodiversity.

3.2.5 Impact of stocking rate on biodiversity

The impact of livestock grazing on biodiversity depends on stocking rates. The literature would appear to indicate that both under-grazing and over-grazing have negative effects on species richness. Walker et al., (1981) found that grasslands with intensive grazing had lower levels of productivity than moderate opportunistic grazing practices. Persistent high levels of grazing is also thought to affect ecosystem function. Intensive grazing led to the decline of productive functional groups because herbivores showed a preference for the most palatable species, whilst under a moderate grazing regime these preferred species were able to persist in the sward and adapt to change and instabilities caused by grazing and drought thereby maintaining structural resilience (Walker et al., 1981).

Overgrazing may exacerbate the high inter-annual variation in productivity on grazing lands. Walker, (1988) has observed a much higher phenological diversity in semi-arid systems subject to moderate grazing compared to those that are intensively grazed. On lightly grazed areas he noted an even mix of early, mid and late season grasses which were able to respond to rainfall wherever it occurred in the season. High stocking rates appear to dampen or reverse the normally positive relationship between diversity and productivity (Worm & Duffy, 2003). In the west of Ireland, heavy grazing leads to an absence of highly palatable early season species which are replaced by later growing species and this has a direct effect on farm productivity and profitability (Silva, 1987; Milne & Osoro, 1997; Perrings & Walker, 1997; Bleasdale, 1998). The implication being that forage production was lower and more unstable on heavily grazed areas compared to lightly grazed land because the sward was not able to respond to early season rains. In the Serengeti, McNaughton, (1985) has also shown that forage production was more stable where the number of species contributing to biomass was high compared to swards where relatively few species contributed to forage production. Sternberg et al., (2000) conducted a 4 year study on the response of a Mediterranean herbaceous community to grazing management in north-eastern Israel. Contrasting different grazing treatments they found that low and high grazing regimes reduced herbaceous diversity but that moderately grazed areas increased diversity.

According to Moravic & Zemeckis, (2007), under-grazing could lead to a loss of biodiversity. Moderate stocking rates on the other hand may lead to maximum biodiversity levels (Milne & Osoro, 1997). Peco et al., (2006), stressed that moderate grazing increases fertility of poor soils and promotes species richness. Adequate vegetation cover contributes to protecting the soil from erosion. Adequate vegetation cover contributes to protecting the soil from erosion. The average stocking rate of 0.7 TLU/ha is considered to be optimal for commonage grassland management (Bakker, 1989).

3.2.6 Impact of other characteristics of livestock on biodiversity

Livestock grazing management is the central issue affecting the maintenance of plant diversity, productivity and management of biodiversity (Watkinson & Ormerod, 2001, Rook & Tallowin, 2003). Grazing offers a potentially important tool for livestock management because of its influence on habitat structure and biodiversity (Rook & Tallowin, 2003). Grazing animals affect ecosystem biodiversity through feed selection and by increasing competition (Duncan, 2005). The degree of feed selection depends also on resource composition and quality, when the resource is rich in diverse species of flora, animals tend to choose plants which meet best their nutritional requirements (Rook et al., 2004; Dumont et al., 2007).

Many studies suggest that spatial heterogeneity can modulate the strength of diversity-productivity relationship in livestock production (Cardinale et al., 2004). The fundamental difference between mown and grazed grassland is that in the latter the behaviour of the grazing animal leads to enhanced structural heterogeneity of the grassland resource (Rook et al., 2004). Factors such as environmental heterogeneity, habitat type, and ecosystem productivity may determine patterns in grazing and vegetation availability (Adler & Lauenroth, 2001). It is recognised that animal abundance and plant species diversity increases with greater habitat heterogeneity as a result of appropriate livestock management (Dennis et al., 1998).

Complementarity and substitutability properties between biodiversity and stocking rate and other physical inputs are particularly important in order to interpret the production model.The degree of substitution is a function of livestock productivity (Osoro et al., 1999). For example, overgrazing and increasing stocking rate beyond the carrying capacity may cause habitat degradation and a reduction in biological diversity. Essentially a reduction in biodiversity may affect the carrying capacity and the resilience of the grassland ecosystems and reduce livestock productivity. This shows a trade-off effect between livestock productivity and biodiversity.

The main issue here is to identify livestock management that can increase productivity and reduce the risk of biodiversity loss to an acceptable level by keeping the stocking rate at appropriate level. Livestock grazing regime and stocking rate often vary even within the same commonages and habitat type (Ni Bhriain et al., 2003; Moran, 2005; Visser et al., 2007). Not only is the level of grazing important but also the timing of grazing and the livestock species involved (Grant et al., 1996). The productivity of grassland depends on the balance between palatable and unpalatable grasses. Thus the losses of palatable grasses have direct effects on livestock productivity and profitability (Perrings & Walker, 1997). Diversity of vegetation may vary according to management and hence in quality and palatability (Milne & Osoro, 1997).

Animal gender and body size may also affect sward diversity. Olff & Ritchie, (1998) suggested that body size plays a role in pasture conservation as heavy animals prevent weed growth and churn up the soil with their hoofs. The age of marketed animal can also alter feeding preferences. Young animals and pregnant lactating females prefer highly nutritious forage and are highly selective during grazing (Rook et al., 2004). The demographic structure of a herd along with animal body size and age of marketed animals' on biodiversity appears to be of importance (Rook et al., 2004). The availability of vegetation, animal species and breed type significantly influences livestock productivity in disadvantaged areas (Osoro et al., 1999).

The age of marketed animals can also alter feeding preferences. Young animals and pregnant lactating females prefer forage with higher nutritive value and so are more selective when they are grazing (Rook et al., 2004). The effects of demographic structure of animals such as body size and age marketed animals' on biodiversity are of great importance (Rook et al., 2004). In grasslands, productivity depends on the balance between palatable and unpalatable grasses available and relationship between this may strongly influence the patchiness structure of the land (Perrings & Walker, 1997).

The impact of grazing differs between types of livestock species and grazing regime. The effects of cattle grazing on the upland areas are considerably different to those of sheep grazing. Sheep are able to select the more digestible parts of grass whereas cattle are relatively unselective (Rook, 2004). Cattle avoid grass species with a high proportion of stems and few leaves whereas stemminess does not reduce accessibility of these species to the same extent in sheep. Cattle often utilize grassland selectively by grazing some areas more intensively than sheep, resulting in local overgrazing (Coughenour, 1991). Cattle in mixed grazing can also increase diversity of species (Pykala, 2005) and structure and cause more disturbances from trampling than sheep, potentially creating bare patches and enhancing seedling germination. Grant et al, (1987) has conducted a comparative study and suggested that sheep has more role than cattle in vegetation management of blanket bog. The evidence from the same study suggested that cattle have advantage in the management of dwarf shrubs.

Traditional breeds may have an economic cost in terms of low profitability and/or production efficiency. In capital-intensive livestock production system, commercial breeds have been shown to outperform traditional breeds, producing more food at lower cost (Yarwood & Evans, 1999). Mostly farmers operating extensive livestock production systems and using traditional breeds to improve biodiversity should benefit from a policy changes. On the other hand, traditional breeds may be better suited to marginal lands and economically marginal conditions such as may arise when biodiversity is the major management goal and may also be able to command a market premium.

Traditional livestock breeds instead of commercial breeds are often recommended for grazing management to meet biodiversity conservation and production goals (Isselstein et al., 2005). Traditional breeds may be better suited to marginal lands and economically marginal conditions where biodiversity is the major management goal. And it has been suggested that more developed commercial breeds may threaten biodiversity (Rook et al., 2004). Policy makers have been given adequate attention to the advantage of indigenous livestock breed use and the impact of breed replacement on livestock biodiversity (Bullock & Oats, 1998).

There is strong evidence that breed loss leads to a significant reduction in genetic diversity. Genetic erosion of domestic animal diversity has placed 30% of breeds in the world at risk of extinction (Hammond, 1996). This is often due to government policy of promoting specialised improved livestock breeds and environmental economic valuation have an important role to play in supporting decisions regarding which breeds should be conserved (Drucker et al., 2001). In Irish uplands, biodiversity management and livestock productivity is of fundamental importance as a mechanism for buffering against output losses due to emerging pests and diseases and as a biological asset for future upland grasslands on which the supply of livestock products depends.

The main causes of livestock genetics resource erosion and factors that threaten indigenous breeds were crossbreeding which could be designed to improve livestock productivity and change in market demand (Drucker et al., 2001; Rege & Gibson, 2003). The introduction of exotic breeds and other social and economic pressures have exposed locally adapted indigenous breeds to the risk of extinction and could lead to a loss of potentially valuable genetic diversity (Rege & Gibson, 2003). Simianer et al., (2003) also indicate that the impacts of livestock breeds are unequal in terms of their contribution to animal genetic resource and importance to local farmers. Researches have proved that replacing hardy local livestock breeds by exotic breeds is unsustainable because they are not being able to reproduce themselves in harsh environments and apparent comparative advantage in productivity is not being realised (Ayalew et al., 2004).

However, there is less empirical evidence conserving effects of livestock characteristics on biodiversity (Rook et al., 2004). There is a need to strengthen understanding of the link between biodiversity and productivity under different livestock grazing management and land property rights. This study helps to improve decision making and policy through assessment of the effectiveness of grazing management in maintaining biodiversity and livestock productivity as well as test different management regimes to determine which performs best.

3.3 Empirical estimation strategy

3.3.1 Two Stage Least Square Method

The main technique used here for analysing the relationship between biodiversity and livestock productivity is the production function approach and it is based on valuing biodiversity as an input in to the production process (Ellis & Fisher, 1987; Freeman, 1991; Barbier, 2000). The appropriate basis for determining the value of an environmental resource is to view the resource as a factor input into the bio-economic production process (Freeman, 1991). The study applies a Two-Stage Least Squares (2SLS) regression procedure. The existence of potential correlation between biodiversity and the disturbance term in the productivity model justifies the use of a 2SLS strategy for the joint estimation of livestock productivity and biodiversity regressions. A 2SLS approach may address the potential endogeneity due to the inclusion of biodiversity covariates in livestock productivity regressions. Both biodiversity and livestock productivity variables are assumed to play the role of the dependent variable.

In the livestock productivity regression, biodiversity is most likely to be an endogenous variable because it is correlated with the error term. Without additional information, we cannot consistently estimate any of the parameters of the livestock productivity regression. Thus biodiversity as an input in to the livestock productivity will be affected by the issue of endogenous bias and needs instrumental variables (IVs) for its estimation. Habitat diversity and edge effects (plot level habitat fragmentation) are used as IVs for biodiversity. Furthermore, the impact of biodiversity is valued in terms of the corresponding change in body size of marketed live animals.

In what follows, we test for endogeneity bias. We first conduct a Wu-Hausman test comparing Ordinary Least Squares (OLS) and 2SLS estimates for the presence of endogeneity bias in the use of the biodiversity indicator and use of biodiversity as an input in the determination of livestock productivity. After including geographical location variables, we discovered that the 2SLS is superior over OLS estimation. This implies that we accept the endogeneity property of biodiversity with respect to livestock productivity. OLS estimation of livestock productivity is inconsistent not only in terms of endogeneity bias but also for the presence of enterprise heterogeneity. This means that livestock production varies in terms of final products and outputs i.e., suckler beef and lamb are completely different products. Such a heterogeneity problem can be solved through a separate regression for each output i.e. we can have independent regressions for beef, sheep and mixed enterprises.

3.3.2 Biodiversity function

In the Irish uplands, the biodiversity function specification is mainly influenced by land property rights. Commonage grasslands play an important role in livestock food production, particularly meat products. We use two indicators of biodiversity for private and commonage lands because biodiversity may be influenced by property rights and land size. Thus the specifications of the biodiversity functions can take two different functional forms for commonage and private landscapes. Thus these separate biodiversity regressions may be written as a linear projection of influencing factors as follows.

(4)

$$B_1 = \theta_1 LnX + \theta_2 LnS + \theta_3 Br + \theta_4 LnBsize + \theta_5 Lnsub + \theta_6 LnHQ + \\ + \theta_7 LnPA + \theta_8 nsh + \theta_9 Ln\bar{A} + \theta_{10} LnD + \delta_1$$

(5)

$$B_2 = \alpha_0 + \alpha_1 LnX + \alpha_2 LnS + \alpha_3 Br + \alpha_4 LnBsize + \alpha_5 Ln(Age) + \alpha_6 LnSub + \\ + \alpha_7 Ln(PA) + \alpha_8 Ln\bar{A} + \alpha_9 LnD + \delta_2$$

Where B_1 and B_2 are the biodiversity indicators for commonage and private lands, respectively. We do not use a measure of species richness or abundance in this study. Instead we measure habitat diversity. It refers to the term *biodiversity*. The terms θ_i and α_i are parameters to be estimated for commonage and private lands, separately. On commonage lands, biodiversity is influenced by conventional inputs (X), stocking rate (S), breed (Br), body size of marketed animals (Bsize), subsidies (Sub), and plot level variables such as habitat quality (HQ), perimeter to area ratio (PA), the number of plots and commonage shareholding (nsh), average land size (A), and distance to the main town (D). On private land, biodiversity depends on age of marketed animals (Age) and other factors listed above but not habitat quality and number of plots. Finally, δ_1 and δ_2 are error components.

In the biodiversity model, we may have more than two dependent explanatory variables that can serve as IVs for biodiversity. These include edge effects (perimeter to area ratio), habitat quality, number of shareholdings, and average plot size. 2SLS estimation requires the availability and validity of IVs. To examine the choice of instruments, we may need to test for their relevance by using an F-test of the joint significance of the excluded instruments. We also applied the over-identification restrictions using a Sargan Hansen J-statistics for over-identifying test for instruments (Di Falco et al., 2010a). In this case, we have more instruments to justify that we have an over-identified equation so the number of instruments should exceed the number of covariates.

3.3.3 Livestock productivity

A Linear-Quadratic livestock production function is used to represent the relationship between biodiversity and livestock productivity. From a livestock management perspective, the functional form plays the most important role in determining the relationship between biodiversity and livestock productivity. This type of production function is suitable for the study of the relationship between biodiversity and livestock productivity. In a sense that, the specification will take the trade-off effect into account and the value of biodiversity may be assessed in terms of its impact on livestock productivity.

The livestock productivity regression can be written in the following form:

6)

$$LnY_i = \alpha + \sum_{i=1}^{k} \beta_1 LnX_i + \beta_2 LnS_i + +\beta_3 Breed + \beta_4 LnSub + \beta_5 Ln\bar{A} + \gamma_1 B_i + \gamma_2 B_i * B_i + \varepsilon_i$$

Where α, β_i and γ_i are parameters to be estimated and the last term in EQ 6 represents the random error components. The dependent variable (Y_i) measures livestock productivity (livestock output per ha). Livestock output is measured in terms of total livestock units (TLU). Biodiversity (B_i) is the main environmental input in livestock production. The conventional inputs include stocking rate (S), number of breeds, farm labour, feed, fertilizer and average plot size (A). The explanatory variables (X_i) denote the rest of the conventional production inputs. Labour is measured in term of the number of hours spent in livestock production. Feed represents the sum of home produced and purchased feed for livestock production. Subsidies (Sub) are also included in the model.

3.3.4 Biodiversity indicators

The biodiversity indicator used in this study is habitat diversity. Habitat diversity refers to the diversity between habitats. A mosaic of habitats and landscapes within a region provides a means to develop a habitat diversity index as a biodiversity indicator and this is often referred to as α-diversity (Whittaker, 1972). The relationship between the proportion of habitats and patch-size in a landscape has been estimated using habitat diversity index with varying degree of habitat loss (Krummel, 1987; Turner et al., 1989). Biodiversity is an increasing function of the number of habitats and the size of habitat (Eichner & Pethig, 2006). Many ecosystem services are positively correlated with the number of habitats (Eichner & Pethig, 2006). Perrings et al., (1995) pointed out that if the main issue is to preserve resilience and ecosystem services then habitat diversity can be a good indicator for biodiversity.

Biodiversity indicators should reflect correlation between livestock management and certain aspects of biodiversity that may be used to analyse the relationship between biodiversity and livestock productivity. On commonage, changes in the landscape mosaic and spatial-pattern of habitats can be measured using Shannon Weaver Index. GIS provides spatial information and geographically referenced data and makes the calculation of habitat diversity possible (O'Neill et al., 1988a). Based on the National Parks and Wildlife Service (NPWS) data on commonage lands and as well as information theory, we can specify a Shannon Weaver diversity index B_1 as a measure of biodiversity (Turner, 1990; O'Neill et al., 1999b) for commonages in EQ7.

7) $$B_1 = -\sum_{i=1}^{m} p_i \ln(p_i)$$

Where p_i is the proportion of habitat types i in commonage land and m is the total number of habitat type in commonage land. The larger the value of B_1, the more diverse is the landscape (Turner, 1990).

The second mutually exclusive habitat diversity used as an indicator for biodiversity depends on private lands. Land use changes are an important factor affecting biodiversity (Sala et al., 1996). The effects of private land use change on biodiversity have focused mainly on the major land use types. The spatial pattern observed in landscapes has been influenced by human activities so that the resulting landscape mosaic is a mixture of natural and human managed patches that vary in size, shape and arrangement (Turner, 1989). Accordingly, we define the Shannon Weaver index of habitat diversity B_2, as a measure of biological diversity for private lands.

8) $$B_2 = -\sum_{i=1}^{k} q_i \ln(q_i)$$

Where q_i is the proportion of habitats type i in private land and k is the total number of habitat types in private land holdings.

3.3.5 Measuring habitat fragmentation (Edge effects)

The effects of habitat fragmentation on biodiversity can be measured through edge effects (O'Neill et al., 1999a). Edge effects are changes in physical and biological conditions at an ecosystem boundary or within adjacent ecosystems (Fischer & Lindenmayer, 2007). Biotic edge effects are changes in biological variables such as species composition of plants and animals. An edge effect index allows us to analyse changes in biodiversity which may be affected by land use changes.

Edge effect index allow us to analyze biodiversity losses which are relevant to human perspective impacts of ecosystem service and management at landscape level. Edge effect is a key component to understand how landscape structure influences habitat quality (Ries et al., 2004). The relationships between size and shape of the altered land cover can influence a number of important environmental phenomenons (Burgess & Sharpe, 1981). In terms of ecosystem function, in wetlands patch shape may change ecosystem's ability to provide certain functions (Geoghegan et al., 1997). The amount of edge between landscapes may be important for the movement of organisms or materials across boundaries (O'Neill et al., 1988b).

Edge effect is a key to understanding how landscape structure influences habitat quality (Ries et al., 2004). An increase in the ratio of edge to patch interior has an effect on the magnitude of biodiversity loss (Hunsakar et al., 1990). Recent research work by Harris, (1988); Quinn & Harrison, (1988); Robinson & Quinn, (1988) and Geoghegan et al., (1997) indicated that the amount of edge to interior space influences the abundance and diversity of organisms and other ecosystem processes such as wetland functioning.

Species richness may be negatively correlated with distance from the fragment edge into the fragment interior (Major et al., 2003). An increase in the ratio of edges to interior has an effect on the magnitude of biodiversity loss (Hunsakar et al., 1990). A lower edge to interior ratio might be expected to favour rare species as some species require a less disturbed interior habitat (Geoghegan et al., 1997). Habitat edges can alter the nature species interactions and thereby modify ecological processes and dynamics at a wider range of scales (Fagan et al., 1999). Habitat fragmentation indices can provide an idea of the risk of diversity loss and thereby provide a measure of ecosystem viability and productivity (Geoghegan et al., 1997).

This present study has adopted the Mundlak, (1978b) average edge effect approach to remove unobserved heterogeneity in the biodiversity-livestock productivity relationship. It is also recognised that geographical plot level variables have a strong impact on the biodiversity-productivity relationship. In what follows habitat fragmentation is measured as an average perimeter area ratio within private and commonage plots. This represents a good IV because it can correlate with biodiversity across habitats but may not necessarily be related to the error term in productivity. The mean of patch size and perimeter to area ratio then become potential indicators of biodiversity change (Geoghegan et al., 1997; O'Neill et al., 1999a) and this is estimated as follows.

$$9)\quad PA = \frac{1}{m}\sum_{i=1}^{m}\frac{P_i}{A_i}$$

Where P_i is the average perimeter, A_i is the area of plots and m is the total number of plots in commonage or private lands. PA is Perimeter to Area ratio.

3.4 Data

The study area includes Co. Galway and Co. Mayo in the west of Ireland. The soils of the upland grazing areas are generally of low productivity and are best suited to extensive cattle and sheep production. No arable farming occurs in the study areas. In the autumn and winter of 2010/11 a total of 100 farms were identified as operating management regimes considered typical of upland commonage farmland. Data were drawn from the official list of Commonage farmers (CSO, 2002). These farms are also in receipt of farm financial support. The list includes households registered as commonage shareholders actively managing commonage land. Of the farmers that were asked to participate in the survey 90% said they would take part. Personal interviews were undertaken by staff from NUI, Galway with the owner-operator at the owner's property. Each interview lasted approximately 45 minutes and followed a standard format. The questionnaire was piloted for one month during February 2010 and this aided the design of the survey. Each survey provided detailed data on revenue and cost summaries, farm premia, use of technology, labour and costs of farm operations, particularly grazing and livestock activities (livestock output, feed production, purchased feed) as well as information on the number of plots, their size, perimeter and land use. Data on livestock output, number of breeds, animal age, animal body size, feed production, purchased feed, and expenditure on major land improvements, fertilizer application and other livestock management were also sought.

The range of enterprises on these farms included sheep, beef and suckler cow production. The sample consists of 70% livestock farmers in Co. Galway and the rest in Co. Mayo. Cattle and sheep grazing is the dominant land use in the study area. Exactly 25% of farms are in suckler beef enterprises. Irish commonage lands are better suited to sheep production whereas private land is good for suckler beef. Livestock grazing management in the sheep enterprise consists of ewes, rams, hogget and lambs. Most sheep farmers produce lambs for export for the Mediterranean market. About 28% of livestock farms in the region are involved with sheep enterprises. The rest are in mixed farming. About 88 % of farmers participate in REPS.

A measure of species richness was not used in this study. Instead we measure habitat diversity as a measure of biodiversity. The spatial pattern observed across the Irish upland landscape has been influenced by human activities so that the resulting landscape mosaic constitutes a mixture of human managed patches that vary in size, shape and arrangement (Turner, 1989). Using information gathered in the farm survey on the size, shape perimeter and vegetation type associated with land parcels on each farm combined with information sourced from the National Park and Wildlife Service (NPWS) on habitat vegetation type, a Shannon Weaver diversity index of habitat diversity was specified as the biodiversity indicator used in the analysis.

Notably, this study uses a spatial data set - the Land Parcel Identification System (LIPS) to identify individual habitat types from the Department of Agriculture Fisheries and the Marine (DAFM). This enables the farm survey to be linked to a GIS habitat dataset. GIS and remote sensing data on habitat types and land-use types were sourced from the NPWS. One of the variables of interest is the quality of the habitat and its effects on biodiversity and productivity. Grazing is thought to affect habitat quality and high values for habitat quality are generally associated with low grazing damage. Thus, data on habitat quality, damage assessments and destocking was also derived from a NPWS and DAFM Commonage Framework Plans (CFP) study. The CFP provides digital information on land area, condition of the commonage, plant species, habitat types involved, presence of rare or endemic species and perimeter-area data (Bleasdale, 1995). It should be noted that perimeter-area data is only available for commonage land, not private land.

3.5 Results

In what follows we provide a short discussion of the habitat types involved with the study area and this is followed by the regression results. Commonage land consists of mainly blanket bog either on its own and/or in combination with a mosaic of other habitats including wet heath, dry heath and upland grasslands. Table 6 shows that upland commonages can be grouped in to five habitats: blanket bog, wet heath, dry heath, upland grassland, and abandoned/scrub habitats. Blanket bog lands cover about 41.3% of commonages, wet heath and dry heath combined cover 47% and other upland grasslands occupy 10.4 % of the rest of commonage lands.

Table 6. Area distribution of the commonage lands

Habitats	**Area (%)**
Blanket bog	**41.3**
Wet heath	**31.6**
Dry heath	**15.3**
Up land grasslands	**10.4**
Others	**1.4**
Total	**100**

Source: DAFM & NPWS

Table 7 shows that the dominant habitats on private land consists of blanket bog, species rich grasslands, wet heath, dry heath, lowland pasture, rough grazing and improved grazing. On private land, the dominant habitat is rough grazing. Blanket bog occupies 18.8% of private lands. Improved grazing lands cover about 18.6% of private land area. Species rich grasslands cover about 13.4% of private land area. Species rich grasslands are usually suitable for haymaking and no fertilizer is applied to these areas. These grasslands have diminished over time. Homesteads and forest cover only 4.1% of private land cover. Trees and buildings are usually not included in the stocking rate calculation.

Table 7. Area distribution of private lands

Habitats	**Area (%)**
Blanket bog	18.8
Species grass land	13.4
Pasture land	15.5
Rough grassland	33.7
Improved grassland	18.6
Total	**100**

Source: Survey data and GIS

Table 8 shows that the impacts of grazing were almost uniformly distributed across four of the dominant habitats. The results of DAFM and NPWS damage assessments indicate that grazing has had a negative impact on the dominant habitats. For example, if we take the severely damaged attribute, the percentage across the habitats was statistically insignificant. The same is true for undamaged as well as moderately damaged attributes. Furthermore, we can merge moderately and severely damaged attributes and calculate the relative ratio of damaged to undamaged areas, then the least undamaged habitat becomes blanket bog, and wet heath becomes the most damaged habitat. However, the percentage margin is very small between blanket bog and dry heath.

Table 8. Grazing damage and percentage distribution in commonages

Habitats/Damage	Blanket bog	Wet heath	Dry heath	Upland grassland	Total
Undamaged	60	41	52	67	54
Moderately damaged	21	30	29	24	26
Severely damaged	19	29	19	9	20
Total	100	100	100	100	100

Source: DAFM & NPWS

The estimated results of the biodiversity regressions and factors influencing biodiversity on commonage lands are reported in Table 9. The results indicate that stocking rate, body size of marketed animals, and subsidies have a significant positive impact on biodiversity in sheep enterprises. In mixed enterprises, habitat quality is positively correlated with biodiversity in commonage lands. Using a pooled regression, habitat fragmentation was negatively correlated with biodiversity for commonage land. Subsidies have positively influenced biodiversity in sheep enterprises whereas subsidies have no impact on biodiversity in mixed enterprises. On the other hand, the evidence suggests that habitat fragmentation is negatively correlated with biodiversity on commonage land in sheep enterprises. However, the biodiversity regression results suggest that edge effect has no significant impact on biodiversity in the mixed grazing system. The mixed livestock grazing regime is less susceptible to the risk of habitat loss.

Table 9. Biodiversity function in the commonage lands

Description of the variables	All commonage		Sheep		Mixed grazing	
	Coefficient	Std. error	Coefficient	Std. error	Coefficient	Std. error
Dependent: Biodiversity						
Log Stocking rate	0.0182	0.0123	0.0506*	0.0252	-0.0095	0.0160
Log Labour	0.0162	0.0283	-0.1538**	0.0570	0.0218	0.0339
Log Feed	0.0342	0.0233	-	-	0.0396	0.0286
No of Breeds	-0.0168	0.0132	-0.0294	0.0257	-0.0239	0.0145
Log Body Size of animals	-0.0168	0.0352	0.5082*	0.2590	0.3637	0.4584
Log Subsidy	0.0738**	0.0351	0.0222**	0.0105	-0.0679	0.0478
Log Habitat quality	0.0837***	0.0181	0.0313	0.0355	0.0906***	0.0196
Log perimeter area ratio- (Habitat fragmentation)	-0.1233***	0.0318	-0.2658***	0.0451	-0.0337	0.0439
No. of shareholding plots – (land fragmentation)	0.1035***	0.0061	0.1172***	0.0089	0.0632***	0.0141
Log average land size	-	-	-0.0118*	0.0103	-	-
Log distance to main town	0.0204	0.0188	0.0309	0.0292	0.0093	0.0231
Constant	-0.6319*	0.3319	-1.5336***	0.2550	-1.066	1.3223
N	65		24		41	
F	56.81		103.67		8.57	
R^2	0.91		0.99		0.74	

Notes: Level of significance: ***=p<1%, **=p<5%, *=p<10%

Habitat fragmentation has occurred as a result of three major components, all of which contribute to a decline in biological diversity. First, edge effects have a significant negative impact on biodiversity in commonage lands. Second, habitat fragmentation is correlated with biodiversity as a result of an increase in the number of plots in sheep enterprises. Third, habitat fragmentation has occurred as a result of a reduction in average patch size. Information such as average plot size, number of plots and plot level variables are useful instruments in characterizing a change in the level of biodiversity (Di Falco et al., 2010a; Di Falco et al., 2010b). The results indicate that there is a negative relationship between habitat fragmentation and biodiversity, particularly for specialized livestock grazing management. Habitat fragmentation has accelerated biodiversity loss on commonage lands, particularly in specialized livestock systems.

The results of a TransLog biodiversity regression on factors affecting biodiversity in private lands are presented in Table 10. In suckler beef enterprises, the results indicate that habitat fragmentation and average plot size is significantly and positively correlated with biodiversity in private lands. Subsidies are also significantly and positively correlated with biodiversity in both suckler beef and mixed enterprises. The evidence also suggests that body size of marketed animals has negatively influenced biodiversity in suckler beef enterprises on private lands. On the other hand, stocking rate is negatively and significantly correlated with biodiversity on private land. However, body size of marketed animals has no impact on biodiversity on private lands.

Table 10. Biodiversity function in private lands

Description of the variables	All private lands		Cattle		Mixed grazing	
	Coefficient	Std. error	Coefficient	Std. error	Coefficient	Std. error
Dependent: Biodiversity						
Log Stocking rate	-0.0224*	0.0153	-0.0249	0.0414	-0.0348*	0.0183
Log Labour	0.0184	0.0140	0.0064	0.0230	0.0343*	0.0182
Log purchased Feed	0.0130	0.0112	0.0381*	0.0203	-0.0117	0.0142
Log Fertilizer	-0.0002	0.0038	0.0070	0.0090	-0.0056	0.0045
No of Breeds	-0.0012	0.0060	-0.00422	0.0138	0.0090	0.0068
Log Body Size of animals	-0.1179**	0.0441	-0.2701***	0.0792	-0.1898	0.1301
Log Age of the animal	-0.0081	0.0534	-0.1100	0.1140	-0.0461	0.0311
Log subsidy	0.0416**	0.0205	0.0828*	0.0432	0.0485**	0.0240
No of private plots	0.0232***	0.0022	0.0291***	0.0077	0.0210***	0.0022
Log Average size of plot	0.1253***	0.0192	0.1225**	0.0501	0.0919***	0.0223
Log distance to main town	-0.0214***	0.0086	-0.0351**	0.0143	-0.0106	0.0108
Constant	-0.6223**	0.2813	-0.8750*	0.5786	-0.3738*	0.2127
N	71		24		47	
F	27.99		11.27		22.47	
R^2	0.84		0.91		0.88	

Notes: Level of significance: ***=p<1%, **=p<5%, *=p<10%

The effects of animal body size, age and breed on biodiversity also depends on livestock enterprise and property rights. Body size of marketed animals is negatively related to biodiversity in specialised enterprises whereas body size has an insignificant impact in the mixed grazing enterprise. The results indicate that animal body size has a negative impact on biodiversity in private lands. Animals with a large body size utilise resources over a large home range. Breed diversity and age of marketed animals has no significant impact on biodiversity in all enterprises.

On commonage land, subsidies are positively and significantly correlated with biodiversity in the sheep enterprise whereas subsidies have no impact on biodiversity in mixed grazing systems. Stocking rate has a positive impact on biodiversity on commonage lands. These results suggest that the present stocking rate does not indicate any harmful effects on biodiversity. However, stocking rate has a slightly negative effect on biodiversity for private lands. Labour has a more positive effect on biodiversity in mixed enterprises whereas purchased feed has the same effect on biodiversity in cattle enterprise.

Note that some coefficients associated with production inputs may not be directly interpreted as an elasticity due to the non-linear nature of the functional form. Table 11 depicts that the elasticity of biodiversity with respect to different factors can be calculated using the proper formulas of elasticity. The elasticity of biodiversity with respect to influencing factors is important in order to analyse the marginal economic impact on biodiversity. Stocking rate has a positive impact only for the sheep enterprise indicating that sheep stocking rate has a less negative impact than cattle.

Table 11. Elasticity of Biodiversity

Elasticity of Biodiversity w.r.t.	**Commonage land**		**Private land**	
	Sheep	**Mixed Grazing**	**Cattle**	**Mixed Grazing**
Stocking rate	0.3890	-0.1089	-0.2219	-0.25879
Labour	1.1827	0.2503	0.0569	0.2546
Purchased Feed	-	0.4541	0.3400	-0.0873
Fertilizer	-	-	0.0628	-0.0416
No of Breeds	-0.3231	-1.0410	-0.0807	0.2078
Body Size of animals	3.9081	4.1759	-2.4104	-1.4104
Age of animal			-0.9819	-0.3423
Subsidies	0.1711	-0.7796	0.7390	0.3601
Habitat quality	0.2408	1.0398	-	-
Perimeter to area ratio (habitat fragmentation)	-2.0439	-0.3870	-	-
No. of plots	1.5216	1.1986	1.1278	0.6757
Average land size	-0.0911	-	1.0927	0.6825
Distance to main town	0.2374	0.1066	-0.3132	-0.0786
Total returns	**2.8272**	**4.9089**	**-0.5888**	**-0.0384**

The elasticity of biodiversity with respect to habitat fragmentation and body size is highly elastic. Thus biodiversity is highly responsive to changes in these two variables (Table 11). On commonages, a 1% increase in body size of marketed animals (in TLU) may lead to a 3.9% increase in biodiversity in the sheep enterprise. In private lands, a 1% increase in body size of animals (in TLU) may lead to a 2.4% decline in biodiversity in suckler enterprises.

Mixed grazing has performed better in terms of biodiversity conservation compared with specialized livestock management. The total factor elasticity of biodiversity indicates that mixed grazing management has the highest total returns to biodiversity with respect to all influencing factors in commonage lands. The highest elasticity of biodiversity is observed with respect to body size of marketed animals' for animals involved in mixed grazing. Elasticity of biodiversity with respect to subsidies was also negative in mixed grazing management. The results also indicate that the biodiversity of private lands has a lower performance than for commonage lands.

The estimated regression results of livestock productivity on commonages are presented in Table 12. The impact of biodiversity on livestock productivity appears to be nonlinear in sheep enterprises. Livestock productivity increases with biodiversity for low values and then decreases for higher values of biodiversity. The sign of the coefficient of the quadratic term is negative while the linear term is positive. However, the relationship between livestock productivity and biodiversity is linear in mixed enterprises.

Table 12. Livestock productivity in commonage lands

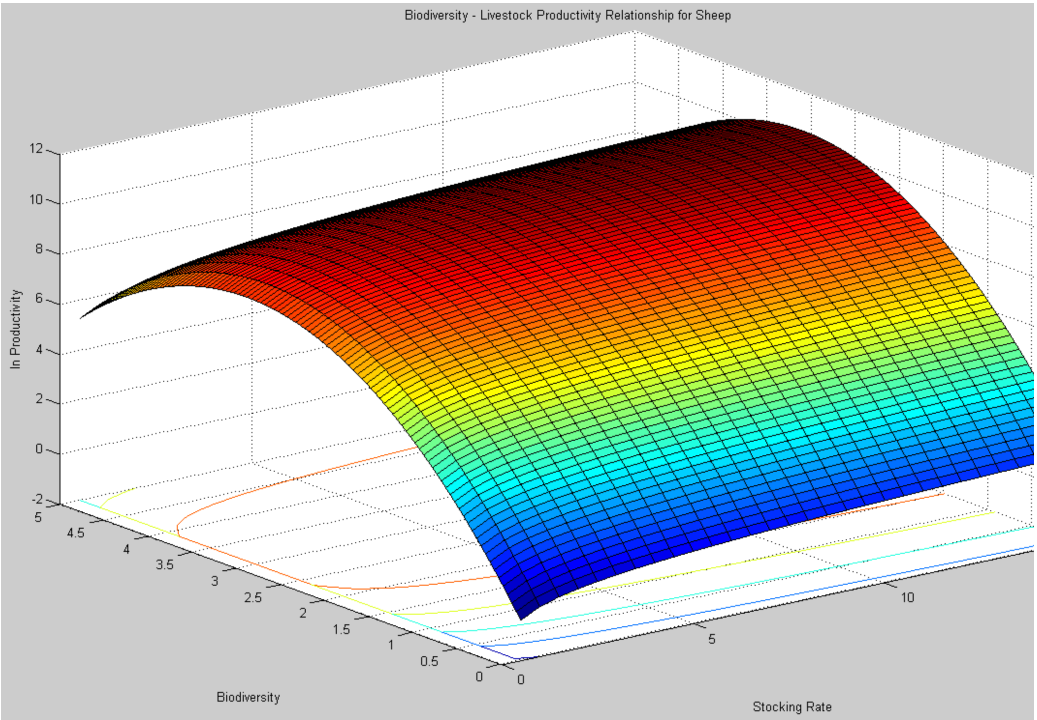
Description of the variables	All enterprises		Sheep enterprise		Mixed enterprise	
	Coefficient	**Std. error**	**Coefficient**	**Std. error**	**Coefficient**	**Std. error**
Dependent: Livestock productivity						
Biodiversity	3.0587**	1.4096	5.7295***	1.4441	1.7370**	1.0225
Biodiversity x Biodiversity	-1.2271**	0.5944	-0.9207**	0.4075	-	-
Log stocking rate	0.9689***	0.0782	0.5664***	0.0609	1.0600***	0.0817
Log Labour	0.1942	0.1565	0.6508***	0.1332	0.1375	0.1803
Log Purchased Feed	0.1708	0.1275	0.3511**	0.1206	-0.0030	0.1544
No. of Breeds	-0.1141*	0.0720	-0.4974***	0.0958	-	-
Log Body Size of animals	-0.0819	0.1580	-	-	-0.7608*	0.4926
Log subsidy	-0.4417**	0.1836	-1.3570***	0.2136	-0.0316	0.2540
Log habitat quality	-	-	0.2071**	0.0997	-	-
Log perimeter to area ratio (habitat fragmentation)	0.2692*	0.1623	1.2793***	0.2922	-0.0593	0.2357
No. of plots	-	-	-0.4480***	0.1035	0.0131	0.1086
Log average land size	-	-	-0.2037**	0.0904	-	-
Constant	3.2442	1.8725	14.2927***	2.3170	-1.3687	2.2959
N	65		24		41	
F (2SLS)	22.40		59.36		25.57	
R^2	0.78		0.98		0.86	
F- Test for endogeneity of biodiversity	39.54		103.53		57.79	
Sargan J-statistics chi (2)	3.34 (0.2145)		3.237 (0.1982)		3.578 (0.2235)	
Test for excluded instruments F	1.56 (0.2355)		1.84 (0.2033)		1.07 (0.2530)	

Notes: Level of significance: ***=p<1%, **=p<5%, *=p<10%

On commonage lands, habitat fragmentation is positively correlated with livestock productivity in the pooled regression at the 10% significance level and shows a highly significant positive relationship for sheep enterprises in particular. Subsidies have a negative impact on livestock productivity in sheep and mixed livestock enterprises. The impact of body size of marketed animals on livestock productivity is negative in mixed enterprises. Similarly, breed diversity has a negative impact on livestock productivity.

Figure 3 shows the relationship between livestock productivity and biodiversity in sheep enterprises on commonage lands. It is important to point out that the livestock productivity function is a non-linear, quadratic functional form with respect to biodiversity in specialized grazing. Such a relationship can be seen as a hump-shaped function. The hump-shaped implies a potential trade-off between biodiversity and livestock productivity.

Figure 3. The relationship between livestock productivity, stocking rate and biodiversity in sheep enterprises



The estimated regression results of livestock productivity in private lands are presented in Table 13. The impact of biodiversity on livestock productivity is found to be linear in mixed enterprises. Livestock productivity strictly increases with biodiversity. Similarly, there is also a positive correlation between stocking rate and livestock productivity in private lands. In private lands, subsidies have a negative impact on livestock productivity but no significant impact in suckler beef and mixed enterprises. Habitat fragmentation has a positive impact on livestock productivity only in specialized enterprises. The effects of body size of marketed animals is significantly and negatively related to livestock productivity in suckler beef and mixed enterprises. Similarly, factors such as labour and age of marketed animals are positively correlated with livestock productivity in mixed enterprises. On the other hand, fertilizer application has a negative impact on livestock productivity in suckler beef enterprises whereas it has a positive impact in mixed enterprises.

Table 13. Livestock productivity in private lands

Description of the variables	All private lands		Cattle		Mixed Enterprise	
	Coefficient	Std. error	Coefficient	Std. error	Coefficient	Std. error
Dependent: Livestock productivity						
Biodiversity	3.3828	3.0002	4.9661*	2.4805	1.1670**	0.5104
Biodiversity x Biodiversity	-3.4881	3.8510	-6.8770**	2.9586	-	-
Log Stocking rate	1.1046***	0.1481	1.3507***	0.2425	1.0660***	0.0889
Log Stocking rate x Log Stocking rate	0.1380**	0.0757	0.0904	0.0710	-	-
Log Labour	0.1304	0.0876	0.3127***	0.0883	-0.0743	0.0931
Log Purchased Feed	0.0922	0.0760	0.9690	1.1678	-0.0196	0.0851
No of Breeds	-0.0321	0.0392	0.0333	0.1213	0.1117**	0.0421
Log Body Size of the animals	0.1399**	0.0579	-0.2386*	0.1325	-0.8262***	0.1766
Log Age of animal	0.4205*	0.2917	-	-	0.3323***	0.0945
Log Subsidy	-0.2038	0.1391	-0.0210	0.1879	-0.16938	0.1413
Log Purchased Feed x Log Purchased Feed	-	-	-0.0778	0.0889	-	-
Log Fertilizer	0.0158	0.02324 1	-0.1020**	0.0374	0.0773***	0.0216
Log perimeter to area ratio (habitat fragmentation)	0.3160**	0.1348	-0.2716	0.2214	0.5379***	0.1512
Constant	-0.8206	1.4095	-4.6458	3.6305	2.0864*	1.0896
N	71		24		47	
F	17.82		16.55		32.65	
R^2	0.78		0.95		0.90	
F- Test for endogeneity of biodiversity	27.26		63.08		28.57	
Sargan J-statistics –chi	2.430 (0.4881)		3.479 (0.2914)		3.5310 (0.3168)	
Test for excluded instruments F-test	3.48 (0.0138)		3.84 (0.2245)		3.83 (0.2567)	

Notes: Level of significance: ***=p<1%, **=p<5%, *=p<10%

The results of the livestock productivity regressions indicate that edge effect has a positive impact on sheep productivity on the commonage land whereas it has no significant impact on beef productivity in private lands. In mixed grazing management, edge effect has a significant positive impact on livestock productivity in private lands. Edge effect has a negative but insignificant impact on productivity in mixed enterprises on commonage lands.

Table 14 presents the effects of elasticity of livestock productivity on biodiversity. Like any other input, the value of biodiversity can be determined by its marginal impact on livestock productivity. The results indicate that the value of biodiversity significantly varies across property right and livestock enterprise types. On commonage land, the magnitude of elasticity of livestock productivity with regards to biodiversity is found to be the highest in sheep enterprise.

Table 14. Elasticity of livestock productivity

Elasticity of Productivity w.r.t.	**Commonage land**		**Private land**	
	Sheep	**Mixed**	**Beef**	**Mixed**
Biodiversity	1.1282	0.3550	0.0325	0.2970
Stocking rate	0.5664	1.0600	1.2081	1.0660
Labour	0.6506	0.1375	0.3127	-0.0743
Feed	0.3511	-0.0030	-0.0390	-0.0196
Fertilizer	-	-	-0.1021	0.0773
No. of Breeds	-0.8084	-	0.0679	0.3947
Body Size of animal	-	-0.7608	-0.4645	-0.8262
Age of animal	-	-		0.3323
Subsidies	-1.3570	-0.0316	-0.0210	-0.1694
Habitat quality	0.2071	-	-	-
Perimeter to area ratio (habitat fragmentation)	1.2793	-0.0593	-0.2716	0.5379
No. of plots	-1.3439	0.0281	-	-
Average land size	-0.2037	-	-	-
Total return to scale (TRS)	**0.4699**	**0.7260**	**0.7558**	**1.6157**

Labour is an important factor in reducing land abandonment practices such as haymaking which is a labour-intensive activity. Livestock production largely relies on family labour not hired labour in the region. Labour productivity appears to be better in suckler beef production than sheep. Purchased feed has an insignificant effect on livestock productivity. This may be because livestock production mainly depends on grazing rather than supplementary feed in the study region. However, it can be seen that purchased feed has a greater impact in suckler beef than sheep production.

Finally, it is important to compare the influence of biological variables on livestock productivity among different livestock enterprises. Stocking rate is highly significant and positively related to livestock productivity. The impact of stocking rate on productivity is higher in private lands than commonages. Livestock productivity depends also on other variables such as body size in commonage and private lands. Animal body size has a significant and negative impact on livestock productivity in mixed grazing indicating that small ruminants (sheep) have a much higher productivity than cattle in the region. On commonages, breed diversity has a significant and negative impact on sheep productivity.

I now focus on the issue of environmental efficiency and biodiversity in chapter four.

Chapter 4. The impact of profit competitiveness on biodiversity related environmental efficiency: a fixed effect stochastic profit frontier model

4.1 Introduction

This paper considers to what extent efforts to enhance farm profitability compromises with biodiversity conservation goals such as Environmental Efficiency (EE) amongst livestock farmers in the west of Ireland. There is a well-established literature on technical efficiency (Kumbhakar et al., 1989; Bollman, 1991; O'Neill & Matthews, 2001; Goodwin & Mishra, 2004; Carroll et al., 2007; Solis et al., 2009) but studies that address the relationship between profit efficiency and environmental efficiency are rare. In addition, efforts to quantify environmental efficiency of livestock production as well as to identify the various sources of inefficiency are rare.

A growing body of literature has explored the relationship between environmental and technical efficiency. Reinhard et al., (1999) used a stochastic production frontier model to study the effects of nitrogen pollution on intensive dairy farms in the Netherlands. Reinhard et al., (2002) reported a positive relationship between technical and environmental efficiency. Much of this literature uses pollution not biodiversity as an indicator of EE whereby pollution such as nitrogen surplus is a detrimental input in the production model (see for example, Färe et al., 1989; Färe et al., 1993; Hetemäki, 1993; Färe et al., 1996; Hetemäki, 1996; Ball et al., 1994; Tyteca et al., 1996; Reinhard et al., 1999; Giannakas et al., 2000; Reinhard et al., 2002; Tyteca et al., 2002; Scheel, 2001; Hailu and Veeman, 2001; Kuosmanen, 2005; Omer et al., 2005; Coelli et al., 2007; Galdeano-Go'mez, 2008; Cuesta et al., 2009). These studies therefore investigate the effects of a negative external effect on technical efficiency where the externality is considered to be pollution to the environment.

This present paper sets out to explore whether biodiversity represents a positive externality and exerts a beneficial effect on the livestock production process.

The main motive for biodiversity related environmental efficiency is to provide information on source of environmental inefficiency in livestock production. There are, however, a limited number of papers that explore the effect of positive externalities due to biodiversity on environmental efficiency. Omer et al., (2005) used a Cobb Douglas frontier framework and considered biodiversity as a desirable input to cereal production. The problem with this approach is that during the modeling the estimation of EE may collapse to the same measure of technical efficiency. We, therefore, build on the work of Reinhard et al., (1999), however, we consider biodiversity as an external benefit not an external cost (such as pollution) and yet we avoid the restriction on the representation of technology which was limited to a Cobb Douglas production function as reported by Omer et al., (2005). This study represents a new departure as a means of assessing farm-specific environmental trade-off between biodiversity and a motive for profit and livestock competitiveness.

The source of driving force for environmental performance can be divided in to compliance push and demand pull categories (Galdeano-Go’mez, 2004). Compliance push force emanate from the expectation of stronger public environmental programme requirements. Demand pull forces are led by private market/consumer preference for specific environmental quality attributes. According to Galdeano-Go’mez, (2004), there are two explanations for the relationship between the application of environmental practices and farm competitiveness. The first win-win hypothesis argues that farms that increase investment on biodiversity improvements can obtain a competitive advantage and thus increase profit (Hart, 1995). The second win-loss perspective argues that environmental improvements introduce greater costs which will end up reducing profit (Walley & Whitehead, 1994). Incurring additional costs to improve environmental performance may increase some farm profit if the action causes an increase in the costs of other farmers over and above their own costs (Galdeano-Go’mez, 2004).

A further question of interest to policy makers is whether part-time farming influences biodiversity related environmental efficiency. Previous studies which have tested for links between off-farm income and efficiency have produced conflicting results. Kumbhakar et al., (1989) used a system approach to estimate technical inefficiencies for Utah dairy farmers and examined the impacts of off-farm income on farm level technical efficiency. The results show that off-farm income is negatively associated with technical efficiency. This indicates that whenever a farmer takes part-time job off the farm, he/she may earn more income but his/her farm may suffer from inefficiency (Kumbhakar et al., 1989). Although off-farm work provides an opportunity for farm households to stabilize household income and reduce uncertainty of future income, off-farm income may have adverse effects on profitability due to negligence (Kumbhakar et al., 1989).

In addition, Bollman, (1991) conducted a test for the differences in technical efficiency between part-time and full-time farms and found that full-time farms are more efficient indicating part-time farmers have higher costs for labour and human capital as a result of higher income and unique characteristics of part-time farming. Goodwin & Mishra, (2004) analysed the relationship between farm level efficiency and off-farm labour supply and they found that greater involvement in off-farm labour markets does indeed lower farm efficiency. The larger off-farm component of farmers' income the less time spent on the farm and the source of efficiency is negatively associated with part-time farming participation.

On the other hand, Solis et al., (2009) show that there is a positive relationship between efficiency and off-farm income. Mishra & Goodwin, (1997) suggest that profit efficient farmers may be expected to have higher relative returns to farm labour thus would be expected to supply less labour to off-farm activities. The cash from off-farm earnings can help stimulate farm investments and improve agricultural productivity (Hazell & Hojjati, 1995). Bagi, (1984) reported that there is no significant difference in the average technical efficiency between full-time and part-time arable farmers.

In Ireland, O'Neill & Matthews, (2001) showed that having an off-farm job was negatively associated with efficiency whereas Carroll et al., (2007) report that there was no significant difference in technical efficiency between full-time and part-time farming. However, there is no study that investigates for differences in environmental efficiency between part-time and full-time farms. Given the significance of biodiversity conservation there is an urgent need to design models that could link policy, farm profitability and biodiversity as a measure of environmental efficiency. This paper aims to deliver empirical evidence on the links between environmental efficiency, biodiversity, and livestock management by analysing commonage farms in Ireland. In addition, the paper determines factors affecting livestock profit and environmental efficiency based on stochastic profit frontier model.

This paper is structured as follows: the first section reviews the literature on environmental efficiency, profit efficiency, and the sources of environmental efficiency; second a description of the survey data and methodological approach is provided; third empirical results are discussed and finally conclusions and recommendations are drawn.

4.2 Literature Review

4.2.1 Profit Efficiency (PE)

The traditional definition of efficiency as defined by Farrell has three components: technical, allocative and economic. Technical efficiency is defined as the ability to achieve a higher level of output, given similar levels of inputs. Allocative efficiency deals with the extent to which farmers make efficient decisions by using inputs up to the level at which their marginal contribution to production value is equal to the factor cost. Technical and allocative efficiencies are components of economic efficiency. It is possible for a firm to exhibit either technical or allocative efficiency without having economic efficiency. Therefore, both technical and allocative efficiencies are necessary conditions for economic efficiency. Economic efficiency is equal to the product of technical and allocative efficiencies.

The profit function approach combines the concepts of technical and allocative efficiency in the profit relationship and any errors in the production decision are assumed to be translated into lower profits or revenue for the producer (Ali et al., 1994). Profit efficiency in microeconomics terms, therefore, is defined as the ability of a farm to achieve highest possible profit given the prices and levels of fixed factors of that farm. Profit inefficiency in this context is defined as the loss of profit for not operating on the frontier (Ali and Flinn, 1989). Battese and Coelli (1995) extended the stochastic production frontier model by suggesting that the inefficiency effects can be expressed as a linear function of explanatory variables, reflecting farm-specific characteristics. The advantage of this model is that it allows the estimation of farm specific efficiency scores and the factors explaining the efficiency differentials among farmers in a single stage estimation procedure.

PE can be defined as ratios of observed profit to maximum feasible profit. PE could also imply underutilization of inputs and if a farmer makes mistakes in allocating inputs, the resulting inefficiency is labeled as allocative inefficiency (Kumbhakar, 1994). Schmidt & Lovell, (1979) have shown that there could be a positive correlation between technical and allocation efficiencies. Technical and environmental efficiencies become the same measures of efficiency when livestock production yields constant returns to scale otherwise they are not equal (Färe & Lovell, 1978).

According to Schmidt & Lovell, (1979), a farm can be profit inefficient in many ways. Profit Efficiency (PE) is used to assess the economic performance of a farm and is a measure of farmers' competiveness in producing maximum profit from a given set of inputs. Chavas et al. (2005) point out that allocation efficiency holds when resource allocation decisions minimize cost, maximize revenue or maximize profit. PE implies both technical and allocation inefficiencies (Forsund et al., 1980).

Kumbhakar, (2001) shows that input demand, output supplies, demand and supply elasticities and return to scale are affected by change in profit efficiency. The overall profit efficiency is not necessarily the product of technical and allocative efficiencies, meaning that technical and allocative inefficiencies are not necessarily independent. Schmidt & Lovell, (1979) have shown that there could be positive correlation between technical and allocation efficiency. Allocative efficiencies were measured by deriving the cost function dual to the estimated production frontier.

Profit inefficiency implies underutilization of inputs and if the farmers make mistakes in allocating inputs, the resulting inefficiency is labelled as allocative inefficiency (Kumbhakar, 1994). If farm is profit inefficient, it implies that reallocation of resources could yield more profits. The presence of such profit inefficiency leads to a decrease in output supply. So the concept of profit inefficiency is used to characterize proper utilization of resources. Lee and Tyler, (1978) indicated that inefficiency could be due to poor managerial skill. In this case, it is assumed that farmers may differ in their ability to utilize the best management practice and input application for livestock production. Profit efficiency of livestock farm reflects the quality of feed utilized and its quality of forage management. It is impossible to measure profit efficiency independent of feed resource management. Profit efficiency of livestock farm reflects the quality of feed utilized and its quality management. For example, higher level of efficiency is important to achieve production of quality meat.

PE can be defined as ratios of observed profit to maximum feasible profit. Kumbhakar, (2001) suggested that the essential idea behind the frontier profit function is that, it is considered as a locus of maximum profit from a given input and output set where the profit of each farm is bounded above by a profit frontier. A farm is said to be profit inefficient for a given set of inputs if the farm profit lies inside the frontier line. Lee & Tyler, (1978) suggested that any downward deviation from the frontier is due to technical inefficiency for the firm as reflected in, e.g., work stoppages, material bottlenecks, and low employee effort. If these inefficiencies could be eliminated, the firm would produce on the frontier. Since most farms are not able to produce on the frontier profit an error is introduced to represent profit inefficiency, which is under the control of farmers. It is assumed that if such inefficiencies are eliminated the farmer would produce on the profit frontier.

In stochastic frontier PE analysis, there are two disturbance terms in the model. The first error term applies to livestock farm’s performance and may be affected by factors entirely under the farmers control such as livestock management. This error component represents profit efficiency. The second disturbance term may capture the effects of random exogenous shocks that may be related or outside the control of the farmer such as unfavourable harvest or market fluctuations. For example, market price fluctuations in primary livestock market are assumed to be exogenous and determined beyond the farmers' control. The essential issue behind the stochastic frontier model is to determine the probability distributions of the two disturbance terms. The first part of error component represents profit inefficiency and is assumed to follow a particular half-normal and one-sided normal probability distribution (Lee, 1983). The second part of the disturbance term represents statistical noise and it is assumed to be a normal distribution in most cases (Greene, 2002).

Orea & Kumbhakar, (2004) suggest that the estimated measures of profit inefficiency are sensitive to the choice of econometric model specification. A number of approaches to efficiency measurement and productivity analysis have been developed. The stochastic frontier estimation of technical efficiency first proposed by Aigner et al (1977) and Meeusen and van den Broeck (1977) used parametric econometric regression techniques. These studies also adopted a stochastic profit frontier approach with a fixed effect model to estimate biodiversity-oriented environmental efficiency and profit efficiency. A comprehensive literature review of stochastic frontier models are provided by (Kumbhakar & Lovell, 2000; Greene, 2001:2002). Essentially, a fixed or random effects model can be used as an extension to stochastic frontier models based on the degree of correlation between explanatory variables and fixed effect (Greene, 2002).

The idea of a fixed effect stochastic profit frontier model is that the profit fixed effect is assumed to be correlated with explanatory variables that are included to explain the profit function. For example, it may be reasonable to assume that farm specific fixed effects are highly correlated with biodiversity. Similarly fixed effects or inefficiency may also be correlated with the level of stocking rate, feed utilisation, and breed selection and of course with biodiversity. On the other hand, in a random effects model, it is assumed that there is no correlation between fixed effects and biodiversity. This assumption may not be reasonable. Furthermore, tests of the appropriateness of model specification can be made using specification tests proposed by Hausman & Taylor, (1981).

The virtue of a Fixed Effect Model (FEM) may be that parameter estimates and farm inefficiency levels could be obtained without assuming the distribution of fixed effect (Greene, 2002). In FE, enterprise specific FE efficiency scores may only be estimated relative to the ‘best’ farmer. Random effects model has an advantage over FEM in that it includes dummy variables in the model. Both models share one common shortcoming, unobserved farm specific heterogeneity is considered as profit inefficiency (Greene, 2002). Greene, (2002) indicated that if there are omitted environmental factors in the regression, inefficiency scores may include unobserved environmental factors and these factors may overestimate farm inefficiency. Heterogeneity can impact on the efficiency score, so it is important to account for it when it is present (Holloway & Tomberlin, 2007). Such characteristics are also a common feature of stochastic frontier models because models do not fully separate the source of heterogeneity from inefficiency.

As a solution, Kumbhakar, (1997) proposes that two stochastic terms for efficiency are considered for farm specific fixed effects and the normal error whereas Greene, (2002) used a true FE model. Heterogeneity may be captured by a management variable such as inclusion of Mundlak average (Greene, 2005b). These include average plot level variables. For instance in this present study, we used the number of plots, perimeter to area ratio and vegetation type to describe physical environmental characteristics. In addition, attributes such as the average body size and age of marketed animals may control for biological heterogeneity in livestock grazing management. Since we assume heterogeneity can be captured by management as well as the correlation between individual effects and explanatory variables, heterogeneity bias in this model is expected to be minimal (Greene, 2005b).

4.2.2 Environmental Efficiency

Environmental Efficiency (EE) is essentially one aspect of economic efficiency in that it focuses on inputs or output which have a negative or positive impact on the environment. A reduction in the level of polluting inputs or an increase in the level of beneficial inputs will have a positive or negative impact on environmental efficiency. EE can be used as a measurement of the trade-offs between livestock production and the environment. To provide an account of approaches taken in the literature to deal with EE, we first focus on studies that considered the effects of negative inputs such as pollution on productivity. Later we discuss positive inputs such as biodiversity.

A micro-economic model of the analysis of EE was first started by Pittman, (1981). Pittman, (1981) modelled pollution as an input in the production function because the relationship between an environmentally detrimental variable and output behaves like the relationship between a conventional input and output. Pittman, (1983) showed how to adjust productivity in the presence of undesirable outputs and estimated parametric environmental efficiency based on a production approach. Pittman, (1983) assigned the shadow price of a single undesirable output for a sample of US paper mills to develop an adjusted Törnqvist productivity index. This approach raises the need for shadow prices since undesirable outputs are not priced in the market. Furthermore, a revised productivity index was used to address the hypothesis that differences in productivity among different farms can be attributed to differences in pollution control behaviour. Beede et al., (1993) used such an index to assess the variation in the management of industrial waste to rank the US industrial sectors according to their pollutant intensities. Giannakas et al., (2000) analyse Greek olive farms and show the importance of efficiency on total productivity. Galdeano-Go’mez, (2008) investigated the impact of environmental performance on total factor productivity using a panel data of Spanish marketing cooperatives.

Nonparametric Data Envelopment Analysis (DEA) assumes a weak disposable technology with respect to detrimental outputs. A weak disposable production frontier is estimated and the relative performance of individual farms is measured with respect to the environmental efficiency (Färe et al., 1989). Such a measure commonly defines negative environmental effects as undesirable outputs (Färe et al., 1989). The disposition of such output involves a cost to the producer. The relative performance of individual farms is measured with respect to the environmental efficiency.

The approach taken by Färe et al., (1993) and others (Hetemäki, 1993) involves estimating a TransLog output distance function by revealing technical efficiency scores and shadow prices for the environmental 'bad'. Färe et al., (1996) achieved this by splitting productive efficiency into input efficiency and environmental efficiency. Other approaches determine an EE score by measuring the degree to which the pollution variable could be reduced. The model used by Ball et al., (1994) shows how to derive EE score by measuring the degree to which the pollution variable could be reduced. Hailu & Veeman, (2001) suggested that the reduction of undesirable outputs in the production process requires a sacrifice of desirable outputs. Scheel, (2001) suggests that efficiency measurement is usually based on the assumption that inputs have to be minimised and outputs have to be maximised. Scheel, (2001) also shows that using a monotonic decreasing transformation function to transform the undesirable output into an ordinary output which is then maximized by programming techniques. Valentin et al., (2004) analyse the relationship between Best Management Practice (BMP) and farm profitability. Results indicate that adoption of nutrient BMP has a significant positive effect on net farm income. Wadud & White, (2000) find that technical efficiency is significantly influenced by the factors measuring environmental degradation and irrigation infrastructure.

Reinhard & Thijssen, (2000) and Coelli et al., (2007) incorporate a material balance condition to estimate environmental efficiency. Coelli et al., (2007) used EE as a function of technical and allocative efficiency. An increase in technical efficiency contributes to an increase in simultaneous improvement of both cost and environmental efficiencies. Coelli et al., (2007) incorporated the material balance concept for nitrogen pollution in livestock production in to the efficiency estimation. The material balance condition implies that the nutrients in the desirable output and the discharge of nitrogen equal to the nutrients in original input (Reinhard & Thijssen, 2000). If the inputs are utilized efficiently, less input is necessary to produce identical output and nitrogen emission is reduced as a result. Coelli et al., (2007) indicate that a substantial potential exists for nutrient pollution reduction via efficiency analysis. Cuesta et al., (2009) suggested that a TransLog distance function specification that treats output systematically by allowing proportional desirable output expansion and undesirable output contraction to estimate environmental efficiency. More recently, a few empirical applications are based on hyperbolic distance functions and these include the work of Cuesta and Zofío (2005) and Cuesta et al., (2009).

A stochastic production frontier model to estimate environmental efficiency was first estimated by Reinhard et al., (1999). Reinhard et al., (1999) estimated both technical efficiency and environmental efficiency of Dutch dairy farms. Reinhard et al., (1999) formulate an analytical setting to calculate environmental efficiency as a single-factor measure of input-oriented technical efficiency based on a TransLog stochastic production frontier model to relate the environmental performance of individual farms to the best practice of environment friendly farming. Nitrogen surplus was assumed to be an environmentally detrimental input. First, a stochastic TransLog production frontier was specified to estimate technical efficiency. In the second stage, an environmental efficiency score was indirectly estimated using parameters of the production frontier model when the environmental effects are treated as freely disposable undesirable outputs, and the environmental determinant input is exogenous (Reinhard et al., 1999).

Reinhard & Thijssen, (2000) discuss the notion of environmental efficiency in dairy farms using optimal allocation of inputs determined by the nitrogen balance. They show that the degree to which input mixes deviate from those that minimize nutrient pollution. The general strategy of such studies has been to include environmental effects in the output vector of a stochastic distance function to obtain inclusive measures of technical efficiency and occasional measures of productivity change over time.

Reinhard et al., (2002) further investigate the variation of environmental efficiency with respect to different factors and a positive relationship between technical efficiency and environmental efficiency was also found (Reinhard et al., 2002). However, this approach for environmental efficiency exhibits severe shortcomings from a modeling perspective. It implies that EE is not estimated explicitly and involves the restriction of some parameter values to a certain functional form.

The literature with a focus on studies that use biodiversity to measure environmental efficiency is limited. There are only a few studies that are based on desirable inputs such as biodiversity to measure environmental efficiency. Omer et al., (2005) used a Cobb Douglas frontier framework and defined biodiversity as a productive and desirable input to model EE in crop production. The use of a Cobb-Douglas to represent technology cannot be used for the case of measuring environmental efficiency mainly because the environmental efficiency estimation may collapse to the same measure of technical efficiency. In the case of the translog representation the two measures can differ. However, as the required negative or zero value of the second own derivative with respect to the environmentally detrimental input is not guaranteed and hence has to be imposed over the whole range of the functional form, the latter is no longer globally flexible. Hence, from the perspective of a theoretically consistent econometric modeling approach also the translog specification is ruled out and consequently a globally flexible and consistent functional form other than the Translog has to be chosen. The Translog specification can be expected to show the best empirical performance of all second order flexible functional forms currently available as different applications have previously shown (Sauer, 2006; Sauer & Abdallah, 2006).

In the present study, environmental efficiency is defined as the ratio of the measured biodiversity using an indicator to the potential biodiversity frontier.

Habitat diversity is used as a measure for biodiversity. It measures habitat diversity across the upland farm landscape using the Shannon Diversity Index. A measure of EE adopted here can be categorized as an input-oriented measure of environmental efficiency. The approach taken here has a number of advantages compared with the work of Omer et al. (2005) and Reinhard et al., (1999). First, our approach is different from Reinhard et al., (1999) because we consider biodiversity as an endogenous positive input into livestock production (not a bad such as pollution). Second, the specification of EE is in contrast with Omer et al., (2005) because the representation of technology has not been restricted to a Cobb Douglas production function. Zellner et al., (1966) argue that biodiversity can be thought of as an exogenous input only if environmental inefficiency is completely beyond the control of the farmer and his/her day to day decision making. The distinction between endogenous and exogenous inputs is relevant in any production decision since the choice of endogenous inputs like biodiversity assumes farmers take decisions over the allocation of resources whereas exogenous inputs are not derived from such an analytical framework. This approach provides a means by which an environmental performance measure such as EE can be used to assess the interaction between livestock production and the environment.

A stochastic profit frontier measurement of biodiversity environmental efficiency is an important indicator of farmers' environmental performance, competitiveness and agro-ecosystem health. It incorporates biodiversity as an environmentally detrimental input and a production influencing factor. A measure of habitat diversity using the Shannon Weaver Index is employed as an indicator for biodiversity. Finally, a further issue is to consider the factors that give rise to the variation in environmental efficiency.

4.2.3 Determinants of inefficiency

Kumbhakar, (2000) indicates that technical inefficiency scores obtained from the production frontier approach have a very limited utility for policy and management purposes unless empirical studies investigate the sources of inefficiency. From a policy point of view, it is of interest to determine whether environmentally inefficient farmers share some common characteristics.

A number of studies suggest a positive relationship between environmental performance and profit efficiency. After controlling for variables traditionally thought to explain farm level economic performance, Konar & Cohen, (2001) find that poor environmental performance is negatively correlated with the intangible asset[5] value of firms and reductions in toxic chemical releases to be associated with greater firm market value. Davidova & Latruffe, (2007) analyse the relationship between financial structure and technical efficiency and show that the importance of financial variables as potential explanatory factors. A study on agri-food market shows that the influence of livestock enterprises and product differentiation on profitability (e.g. Galdeano-Go'mez, 2004).

Regardless of the tools used to measure environmental performance, several studies suggested that there actually exist highly significant differences among decision making units regarding environmental efficiency (Tyteca, 1996). Trip et al., (2002) show that there is a positive association between firm efficiency and the quality of management decision making. Cormier et al., (1993) results suggested that bad environmental performance negatively affects its market value. In commercial greenhouse growers, the impact of decision making on the firms' efficiencies is measured by Trip et al., (2002) using a stochastic frontier production function, and show that there is a positive association between firm efficiency and the quality of management decision making. In commercial greenhouse growers, Jaggi & Freedman (1992) found that markets are not rewarding for good environmental performance thus in short run, farms profitability is negatively affected by pollution abatement activities involving heavy expenditure. Stanwick & Stanwick (1998) report that a significant correlation exists between low pollution emission levels and high profitability of firms.

[5] Intangible asset is defined as factor of production and special resource that allows a firm to earn profit over and above the return on its tangible asset (Konar & Cohen, 2001).

In marked contrast, Giannakas et al., (2001) show a positive relationship between technical efficiency and use of chemical inputs (fertilizer and pesticide). Llewelyn & Williams, (1996) showed that inefficient farmers use excessive nitrogen fertilizer and become unprofitable. In Ireland, Buckley, (2010) indicates that there exists considerable inefficiency in the utilisation of nitrogen and phosphorous fertilisers. Zhang & Xue, (2005) used Translog stochastic production frontier function to estimate environmental efficiency along with technical efficiency in China's vegetable production. In their studies, fertilizer and pesticide use were considered as environmentally detrimental inputs. Diminishing return to scale was estimated and technical efficiency was impressively very high and suggested that a great deal of potential to reduce pesticide use. Whittaker et al., (1991) suggest that the existence of pest management practices could substantially reduce pesticide use without incurring economic losses.

In the south west of England, Hadri & Whittaker, (1999) used stochastic frontier approach to ascertain relationship between technical efficiency, farm size and use of agrochemical potentially contaminating the environment and they found that there is a positive relationship between technical efficiency and use of agrochemical contaminants. In addition, they find that a negative relationship between technical efficiency and farm size as well as use of fertilizer and pesticide. It was also observed that the more efficient farms were larger farms that used a higher volume of environmental contaminants.

The role of farm subsidies also has important effects. Empirical results from the Finnish dairy farms show that subsidy is positively related to efficiency (Kumbhakar, et al., 2009). Areal et al., (2012) find that environmental payments received by the farmer are linked to inefficiency and farm efficiency scores change when taking in to account the positive externality outputs. More efficient farms also drive a lower proportion of their gross margin from subsidies than do less efficient farms and essentially more diversified farms are more efficient for reason of flexibility because more specialized farms are less able to adapt to changing market and policy conditions (Hadley, 2006). Inefficiency also affects technology adoption decision (Kumbhakar, et al., 2009). Hadley, (2006) investigated dairy farms in the UK and found that the most efficient dairy farms have high subsidy to gross margin ratio, low debt to asset ratio and are also less specialised. Iraizoz et al., (2005) analysed technical efficiency and profitability in Spanish beef sector and found that subsidy has a positive impact on efficiency. Tzouvelekas et al., (2001) argued that the use of subsidies might lead to increased technical inefficiency, especially if subsidies attract more efficient farmers who are more interested in improving efficient farming practices than additional support. These findings make more sense. Since inefficiency is an important factor behind low productivity, one might ask whether less efficient farmers are indeed more likely to participate in environmental protection scheme. Thus, in the evaluation of environmental schemes, it is necessary to recognize that subsidy may be associated with inefficiency. Thus, subsidies should not be designed in a way that promotes inefficiency.

According to Abdulai & Huffman, (2000), the net effect of non-farm work on profit efficiency is ambiguous; participation in the nonfarm labour market may restrict production and decision-making activities, thereby increasing inefficiency. On the other hand, increased non-farm work reduces financial constraints, particularly for resource-poor farmers and thus enables them to purchase productivity enhancing inputs.

Lass & Gempesaw, (1992) used a random effects coefficient regression method to determine technical efficiency of Massachusetts dairy farms and the results indicate that hired labour, land and machinery were used in excess of the efficient level and livestock supplies were underutilised by farms. The results indicate that the substitution of family labour with off-farm labour has a detrimental effect on technical efficiency. Tzouvelekas et al., (2001) note that family operated farms are relatively more inefficient than farms using hired labour in olive farming in Greece, whereas Dhungana et al. (2004) conclude that use of family labour is positively related with efficiency in rice farming in Nepal. On the other hand, Battese et al., (1996) conclude that both hired and family labour are equally efficient in wheat production. In Ireland, O'Neill & Matthews, (2001) reported that large family size and higher level of borrowing are positively associated with technical inefficiency. In the Irish dairy sector, O'Brien et al., (2003) indicates that an opportunity exists to improve returns and farm efficiency on labour input.

For suckler beef producers, Leahy et al., (2004) reported that labour efficiency tend to increase with farm and herd size. In the UK beef and sheep farms, Dowle & Doyle, (2003) indicate that in order to study efficiency, it is important to assess the implications of altering fertilizer, stocking rate, feed and grazing management strategies. Hadley, (2006) shows that factors that consistently appear to have a significant effect on the difference in efficiency between the English and Wales beef and sheep farms. These differences can be explained by farm and herd size, debt ratio, farmers' age and level of specialisation. For Spanish dairy farms, Alvarez et al., (2006) explored the relationship between milk quota values and efficiency and they found that efficiency increases with stocking rates. In Ireland, Wallace & Moss, (2002) showed that there exists variation in terms of levels of technical efficiency, scale of operation and relative rates of return between dairying and beef/sheep production.

Traditional breeds may have an economic cost in terms of reduced economic output and/or production efficiency. Traditional breeds may be better suited to economically marginal land where biodiversity is the major management goal (Rook et al., 2004). In capital-intensive livestock system, commercial breeds have been shown to outperform traditional breeds, producing more meat at lower cost (Yarwood & Evans, 1999). In the New Zealand beef and sheep farms, Paul et al., (2000) indicates that it is the financial constraint that farms heavily in debt face and constrain their ability to adjust to changing market and policy circumstances and hence decrease technical efficiency.

Productivity was strongly influenced by the availability of manpower in the farm households. Gavian & Fafchamps, (1996) found that there exists an inverse relationship between farm size and efficiency. Fan & Chan-Kang (2005) show that a positive relationship exists between farm size and labour productivity. Helfand & Levine, (2004) show that the relationship between farm size and efficiency is non-linear, efficiency first falling and then rising with farm size. A stochastic profit function approach was adopted by Huang et al., (1986) to investigate the technical efficiency of small and large farms in India and technical efficiency of farms was found to be positively related to farm size**.** The authors conclude that small farms are less efficiency. Rahman & Hasan, (2008) show that land type, soil quality and delay in sowing have an important effect on efficiency and omission from the inefficiency estimation may lead to an upward bias.

Land fragmentation is considered to be an obstacle for improving agricultural productivity (Di Falco et al., 2009). Land fragmentation is thought to induce inefficiency in agriculture by increasing the cost of production (Tan et al., 2008). Tan et al., (2008) report that land fragmentation increases production costs and give rise to technical inefficiencies. Parikh & Shah, (1994) report that land fragmentation gives rise to a fall in technical efficiency. Niroula & Thapa, (2005) report that fragmented land parcels discourage farmers from adopting agricultural innovations and new technologies as a means of improving productivity. Tan et al., (2008) also suggest that farmers with more and smaller plots tend to adopt fewer modern technologies as compared to farmers with fewer and larger plots.

Consideration of farm fragmentation in efficiency appears important after the work by Nguyen et al., (1996) found that land fragmentation has negative impact on productivity and show that productivity gains were associated with economies in plot size rather than total farm size. Tan, (2005) conclude that increase in the number of plots has a positive impact on technical efficiency in rice production. Land fragmentation may foster crop diversification, but reduces overall profitability due to an inefficient allocation of land (Di Falco et al., 2010a). Penov, (2004) shows that land fragmentation has contributed to the abandonment and decline of Bulgaria’s irrigation systems.

Davidova et al. (2002) found a significant effect of farm location on efficiency in the Navarra region of Spain with the best performing farms located in the middle of the region and the worst in the northern counties due largely to their mountainous landscape. Environmental factors have been seen as the omitted variables from assessments of economies of size (Bhalla & Roy, 1988). For example, Benjamin, (1995) claims that unobserved agri-environmental factors are responsible for the observed inverse productivity relationship and Sen, (1975) suggested that such relationship could be the result of a negative correlation between farm size and unobserved land quality. Bhalla & Roy (1988) also found that differences in soil quality across households within the same district partially explain the inverse productivity relationship which they discovered.

The effect of the household head's age on inefficiency is nonlinear (Abdulai & Huffman, 2000), As a young household head ages, the efficiency of the household decreases until maximum inefficiency is reached when the household head is 33 years old. After that, the household becomes more efficient as the household head's age increases. Adesina & Djato, (1997) used a profit function to estimate relative efficiency of women in African agriculture and results showed that the relative degree of efficiency of women was similar to that of men and the study provides support for efforts to eliminate gender bias in agricultural production.

Kumbhakar et al., (1991); Mathijs and Vranken, (2000) studied human capital variables and indicate that there is a positive and significant relationship between education and technical efficiency in dairy farming. Kumbhakar et al., (1991) indicate that education was positively associated with greater productivity and enhances managerial ability. Rauf et al., (1991) estimated the relationship between education and technical efficiency in the entire irrigation areas of Pakistan and find that the effect of education on technical efficiency was substantial. Wang et al., (1996) developed a shadow price profit frontier model to estimate profit efficiency of Chinese farm households. Farmers' educational level, family size and per capita net income were found to be positively affecting profit efficiency. Ali & Flinn, (1989) estimated stochastic profit frontier of modified Translog type for rice farmers in Pakistan. Factors which were significantly contributing toward in describing the variability in profit losses were related to the level of education, off-farm employment and fertilized application. Kumbhakar, (2001) used augmented Translog profit function, incorporating both technical and allocation efficiency. Stefanou & Saxena, (1988) test for the effects of training of farm operators on efficiency and found that both education and experience have a significant positive effect on the level of efficiency and they are substitutes. In Ireland, Carroll et al., (2007) show that efficiency levels are positively correlated with extension use, soil quality, farm size and level of specialisation. Similarly O'Neill et al., (2002) found that extension services were positively associated with technical efficiency.

Kay & Edwards, (1994) and Wilson et al., (2001) find that having more years of farm experience were positively associated with higher levels of technical efficiency and concluded that differences in farm performance is due to variation in management skills. There are a significant opportunities to increase productivity through more efficient use of farmers resources and inputs with current technology by enhancing farmers skills (Ali & Chaudhry, 1990). Seckler & Young, (1978) argue that differences in management inputs are more important and farms with good managers may yield more profits to invest in land to increase their income and may purchase loss-making farms that have inferior management. Johnson et al., (1994) estimated production efficiency in Ukraine using farm level panel data and found decreasing technical efficiency in crop production. Factors such as managerial structure and policies on capital and other input allocation were contributing towards wide variability in technical efficiency of farms.

The results generated by profit frontier based efficiency models are particularly sensitive to outliers, since frequently it is the outliers that define the frontier. Hence it is perhaps surprising that the detection of outliers has not received more attention in the efficiency measurement literature. One notable exception is Wilson (1993), which generalises the outlier measure to the case of multiple outputs. Moreover, the detection of outliers can be complicated by the existence of multiple outputs. The stochastic frontier approach to efficiency analysis (Aigner, Lovell and Schmidt, 1977) does allow for some stochastic variation estimating the frontier, but requires the specification of particular distributions for stochastic deviations from the frontier, and the fit can similarly be affected by outliers. The only substantial difference in this case is that the efficient frontier can be affected by outliers that are well inside the frontier. Ruggiero, (1999) reported similar problems due to omitted variables when estimating SFA production frontiers. Because any loss of precision can be attributed to an increase in noise and the performance of all methods declines as noise increases.

Di Falco & Perrings, (2003) show that profit maximizing farmers choose greater crop diversity if diversity is positively related to productivity and reduce income variability. Increasing attention is being paid to the impact of biodiversity on potential yield variability and risk (Di Falco & Perrings, 2005). Ahmed et al., (2005) determined profitability and constraints in potato production. Net returns were not enough to cover variable cost due to low price of potato during that year. Battese & Coelli, (1995) allow simultaneous estimation of the parameters of the stochastic profit frontier and efficiency model. They applied a two-stage parametric procedure to investigate determinants of technical inefficiency among farms. In the second stage, the predicted inefficiencies obtained from a stochastic frontier were regressed upon a vector of farm-specific factors such as firm size, age, education of farmers and time. However, household age and education did not significantly affect inefficiency.

4.3 Econometric model

4.3.1 Biodiversity-Oriented Environmental Efficiency

This study provides a means of assessing farm-specific environmental trade-offs between biodiversity protection and profit efficiency. We modify a fixed effects stochastic profit frontier function approach (Schmidt & Sickles, 1984) to the biodiversity-oriented environmental efficiency estimation. The model described below follows a three-stage estimation strategy to estimate the relationship between biodiversity-oriented environmental efficiency and profit efficiency. In the first stage, biodiversity-oriented environmental efficiency is estimated using a fixed effects stochastic biodiversity frontier function and

Instrumental Variable (IV) technique. In the second stage, a stochastic profit frontier regression technique is used to estimate profit efficiency. It is assumed that biodiversity is treated as a desirable endogenous input. Finally, a truncated model is applied to estimate factors affecting the variation in environmental efficiency.

There are two key conditions to be considered for the first two-stage fixed effects estimation. Instrumental variables (IVs) should be uncorrelated with fixed effect livestock management and it should be significantly correlated with biodiversity. In the first stage of estimation, the stochastic biodiversity frontier can be explained by a linear projection on conventional inputs and additional information obtained from the NPWS GIS data which is used as IVs in the model. The two main plot level GIS variables used as IVs in our model are the number of plots and Perimeter to Area (PA) ratio. Number of plots in the landscape measures land fragmentation whereas perimeter to area ratio measures habitat fragmentation. In addition, information on animal characteristics such as average body size and age will be used to explain variation in biodiversity. The method of IVs provides a solution for the problem of endogeneity.

The properties of various estimators to be considered depend on the existence of the relationship between biodiversity and the fixed effect. Thus we introduce a fixed effects stochastic biodiversity frontier model with a Linear Quadratic (LQ) specification form. It may be written as follows:

10) $$B_{ij} = c_i + \beta_2' x_{ij} + \theta_2' z_{ij} + \upsilon^*_{ij} - u^*_{ij}$$

Where, j=1, 2 stands for repeated observation of cross sectional data for private and commonage lands, respectively. It is assumed that the coefficients of the variables vary between private land and commonage. B_{it} is an endogenously determined biodiversity input. c_i is farm specific fixed effects which are assumed to be random variables and distributed independently across farms. x_{ij} and z_{ij} are vectors in linear and quadratic specification form. x_{ik} is a vector of production inputs (livestock, breed, labour, feed, and land). z_{ij} represents farm specific heterogeneity not directly related to the production structure but which indirectly affects the production process. We used the animal body size variable to control for heterogeneity in animal production. x_{ij} represent explanatory variables and z_{ij} is IVs.

The biodiversity frontier model consists of the usual regression type but with an error term equal to the sum of two parts. The first part is typically assumed to be normally distributed and represents the usual statistical noise, such as luck, weather, market failure, and other events beyond the control of the firm. Thus the first error, $\upsilon*_{ij}$ is a random shock created by the biodiversity frontier, this could take a positive or negative value. The second error component, u^*_{ij} represents biodiversity-oriented environmental inefficiency.

The distributional assumptions of error terms are as suggested by Aigner, Lovell and Schmidt (1977), υ^*_{ij}~N$(0, \sigma^*{}_{\upsilon}{}^2)$, a normally distributed random variable. And the environmental inefficiency component, u^*_{ij}=|U^*_{ij}|~N(0, $\sigma^*{}_u{}^2$) has a half normal distribution (Lee, 1983). In a fixed effects biodiversity frontier model, the household level fixed effect is assumed to be correlated with explanatory variables, e.g. biodiversity is correlated with environmental inefficiency. It is also important to note that the environmental efficiency measure is an input-oriented estimate based on a biodiversity frontier function whereas in the second stage, profit efficiency is an output-oriented estimate based on a fixed effects profit frontier function.

Further biodiversity-oriented environmental efficiency can be estimated as the ratio of frontier biodiversity to observed biodiversity input. According to Schmidt & Sickles, (1984), the environmental efficiency (EE_i) score in fixed effect stochastic process can express using the following two steps:

11) $u^*_i = \max(c_i) - c_i$

where $c_i = \varepsilon^*_{ij} \lambda^* / \sigma^*$; $\sigma^* = [\sigma^*{}_{\upsilon}{}^2 + \sigma^*{}_u{}^2]^{0.5}$; $\lambda^* = \sigma^*{}_{\upsilon} / \sigma^*{}_u$

12) $EE_i = \exp(-\hat{u}_i^*)$ and $0 < EE_i \leq 1$

The environmental efficiency of farmers can be compared in relation to the most efficient farmer in the sample. The final value of EE is independent of the level of inputs. The parameters of the stochastic biodiversity frontier are estimated simultaneously given appropriate distributional assumptions associated with errors. If the $u_i > 0$ and ε_i is negatively skewed, then there is evidence of environmental inefficiency. The biodiversity frontier function can be obtained from the regression of biodiversity on observed levels of conventional inputs, characteristics of marketed live animals and plot level geographical variables (area to perimeter ratio, number of plots and farm size etc.). The change in biodiversity is understood as the modification of biodiversity components caused by livestock management and land use changes. Thus high environmental inefficiency may be related to livestock management practices that threaten biodiversity.

The main advantage of the biodiversity frontier model is the use of plot level geographical location variables and characteristics of marketed animals (z_{ij}) which are used as IVs in the fixed effects stochastic frontier model. Mundlak, (1978b) was the first to suggest that the average plot level variables be incorporated in the biodiversity frontier regression. This implies that environmental inefficiency estimates derived from this model will reduce unobserved farm specific heterogeneity that are correlated with production inputs. The plot level variables and farm specific characteristics of marketed animals (z_{ij}) are variables that help explain heterogeneity effects in the biodiversity frontier function. The inclusion of Mundlak's adjustment variables is expected to improve the estimation of profit efficiency as well as environmental efficiency estimation.

4.3.2 A Fixed Effect Stochastic Profit Frontier Model

A fixed effect stochastic profit frontier model in Linear Quadratic (LQ) specification form may be written as a function of biodiversity, conventional inputs and key environmental variables as follows in EQ. 13.

13)

$$\pi_{ik} = p_L Y_{ik} - C(w, Y_{ik}) + \upsilon_{ik} - u_{ik} = \alpha_i + \beta' X_{ik} + \theta' Z_{ik} + \gamma_1 B_{ik} + \gamma_2 B_{ik}^{\ 2} + \upsilon_{ik} - u_{ik}$$

Where, k=1,2,3 stands for repeated observations of cross sectional data for livestock enterprises: suckler beef, sheep and mixed enterprises, respectively. It is assumed that the coefficients of the variables vary across the livestock enterprises. π_{ik} is farm profit which is considered to be a stochastic variable. Y_{ik} is livestock production. p_L and w are livestock price and input prices, respectively. pY_{ik} is farm livestock revenue and $C(w,Y_{ik})$ is the cost of production. We assume that producers face output prices p_L and input prices w. X_{ik} and Z_{ik} are vectors in linear and quadratic specification form. X_{ik} is a vector of production inputs (livestock, breed, labour, feed, and land). Z_{ik} represents farm specific heterogeneity not directly related to the production structure but which indirectly affects the production process. Heterogeneity variables include subsidy, body size of marketed animals and spatial plot level geographical variables (e.g. number of plots and area to perimeter ratio). Animal's body size and plot level variables may capture heterogeneity in livestock grazing management in commonages whereas subsidy serves to control for heterogeneity in the livestock enterprises.

The model focuses on a parametric representation of an LQ functional form to estimate profit inefficiency. The main treatment input is biodiversity (B_{ik}) which is included in the model as a LQ form. Biodiversity is treated as the main input in livestock production. The model is interpreted by treating (α_i) as a farm specific fixed term. The farm specific fixed effects are assumed to be random variables and distributed independently across farmers. Such fixed effects can capture the marginal effect of grazing management which is under the influence of livestock farmers. A fixed effects stochastic frontier model assumes that individual effects are correlated with explanatory variables included in the profit function, i.e. COR (α_i,X_{ik}) ≠0. In the presence of such correlation, OLS (Ordinary Least Square) and GLS (Generalized Least Square) estimation yields biased and inconsistent estimates of technology parameters

($\alpha_i, \beta, \theta, \gamma_1, \gamma_2\, \sigma_\upsilon^{\,2}, \sigma_u^{\,2}$). The traditional technique to overcome this problem was to eliminate the individual effects in the sample by using a fixed effects stochastic profit frontier function as suggested by Schmidt and Sickles, (1984).

A fixed effects stochastic profit frontier is used as a parametric representation of the production system and assumed to be stochastic in order to capture internal and external exogenous shocks. Kumbhakar & Tsionas, (2007) indicate that the efficiency of a farm is inherently a stochastic concept. This model involves the specification of two error terms: one with statistical noise while the other represents technical inefficiency in the production process. Two sided shocks may explain the different levels of inefficiency across farms. The stochastic error part of the frontier υ_{ik} captures events beyond the control of farmers. It could be either positive or negative. For example, statistical noise can be profit shocks related to market fluctuations. The second error component (u_{ik}) represents biodiversity-oriented profit efficiency and must be positive (Greene, 2002). ALS suggested that the distributional assumptions for υ_{ik}~N $(0, \sigma_\upsilon^{\,2})$ is a normally distributed variable and the profit efficiency variable $|U_{ik}|$~N $(0, \sigma_u^{\,2})$, is half normal distribution.

There are three important practical advantages of this model. First, biodiversity is considered as an environmentally determinant input. Second, the relevant environmental factors and geographical location characteristics are usually omitted in many previous models but in our case they are indeed included in the model. Third, it may also be possible to include plot level average variables such as number of plots and average area to perimeter ratio as suggested by Mundlak (1978b). These variables allow parameters to be consistently estimated using a fixed effects stochastic profit frontier regression. Without including these variables, the FE model would tend to overestimate profit inefficiency.

The estimation of profit efficiency in this model consists of a number of steps. First, we save the residuals of the stochastic profit frontier regression. Our main interest centres on disentangling measures of farm profit efficiency. With the parameter estimates in hand, it is possible to estimate the composed deviation by "plugging in", the observed data and the estimated parameters as shown in EQ. 14.

14) $$\varepsilon_{ik} = \upsilon_{ik} - u_{ik} = \pi_{ik} - \beta' X_{ik} - \theta' Z_{ik}$$

Next, observation-specific estimates of inefficiency can be obtained by using the distribution of the inefficiency term conditional on the estimate of the entire composed error term, E(u_{it} $|\varepsilon_{it}$). Jondrow et al., (1982) (JLMS); Greene, (2005a) have devised a formula for disentangling these effects as shown in EQ.15.

15) $$\hat{u}_{ik} = E(u_{ik} / \varepsilon_{ik}) = \frac{\sigma\lambda}{1+\lambda^2}\left[\frac{\phi(\alpha_{ik})}{1-\Phi(\alpha_{ik})} - \alpha_{ik}\right] \quad ;$$

Where, ϕ and Φ are PDF and CDF of normal probability distribution. σ^2 is the variance of ε_{ik} and $\sigma = [\sigma_\upsilon^{\ 2} + \sigma_u^{\ 2}]^{0.5}$. Maximum likelihood estimation could provide rho $\lambda = \sigma_\upsilon / \sigma_u$ and the fixed effects, α_i. Greene, (2004) suggested that the benchmarking analysis should be used as an instrument to disentangle the errors. Thus we can address the issue of inefficiency measurement given the observed data.

16) $$\hat{u}_i = \max(\alpha_i) - \alpha_i \quad ; \quad \text{where} \quad \alpha_i = \varepsilon_{ik}\lambda / \sigma$$

Profit efficiency (PE_i) score of the ith farmer can be estimated as the distance between the maximum fixed effect and farm specific fixed effect as proposed by Schmidt and Sickles, (1984). Fixed effect (α_i) provides an estimate of proportional inefficiency. In other words, farm level inefficiencies can be estimated by shifting the profit function upward so that each fixed effect is measured as a deviation from the benchmark level (Greene, 2003). PE is a relative value and estimated with respect to the best farm in the sample. Such

Profit Efficiency (PE) measures are not dependent on the level of factor inputs for a given farm (Battese & Coelli, 1995). Thus individual specific inefficiency is typically estimated from the exponential disturbance term (EQ. 17).

17) $PE_i = \exp(-\hat{u}_i)$; $0 < PE_i \leq 1$

A basic distinction is made between profit efficiency and environmental efficiency. There are some basic differences between Output Oriented (OO) technical efficiency and Input Oriented (IO) efficiency (Kumbhakar & Tsionas, 2007). In this case, profit inefficiency is a combination of both OO and IO inefficiencies (Kumbhakar & Tsionas, 2007). The notion of profit efficiency encompasses the inefficiency of all factors employed in production including biodiversity. Biodiversity-oriented environmental efficiency represents Input Oriented (IO) efficiency. The focus of the paper now turns to the issue of data collection.

4.4 Data

This study is located in County Galway and County Mayo in the Republic of Ireland. The landscape is comprised of large expanses of western blanket bog. The soils of the upland grazing areas are generally of low productivity and are best suited to extensive cattle and sheep production. Very little arable farming occurs in the study areas. In the autumn and winter of 2010/11 a total of 100 farms were identified as operating management regimes considered typical of upland commonage farmland. Data were drawn from the official list of Commonage farmers (CSO, 2002). These farms are also in receipt of farm financial support. The list includes households registered as commonage shareholders actively managing commonage land. Of the farmers that were asked to participate in the survey 90% said they would take part. Personal interviews were undertaken by staff from NUI, Galway with the owner-operator at the owner's property. Each interview lasted approximately 45 minutes and followed a standard format. The questionnaire was piloted for one month during February 2010 and this aided the design of the survey. Each survey provided detailed data on revenue and cost summaries, farm premia, use of technology, labour and costs of farm operations, particularly grazing and livestock activities as well as information on the number of plots, their size, perimeter and land use. Data on livestock output, number of breeds, animal age, animal body size, feed production, purchased feed, and expenditure on major land improvements, fertilizer application and other livestock management were also sought. The survey focused principally on market costs and benefits. The range of enterprises on these farms included sheep, beef and suckler cow production. Exactly 25% of farms are in suckler beef enterprises. Livestock grazing management in the sheep enterprise consists of ewes, rams, hogget and lambs. Most sheep farmers produce lambs for export for the Mediterranean market. About 28% of livestock farms in the region are involved with sheep enterprises. The rest are in mixed farming. The sample consists of 70% livestock farmers in Co. Galway and the rest in Co. Mayo.

Regulatory measures, supported by the Common Agricultural Policy (CAP) are known to play an important role in supporting farm incomes and in influencing farm management. Data were gathered on the single farm payment as well as agri-environment measures such as REPS (Emerson and Gilmour 1999).These two instruments are very different. The single farm payment is given to the farmer according to land area. REPS, instead, is a different scheme that aims to link financial support to environmental goals. All respondents were asked a series of questions on sources of household income and socioeconomic characteristics (i.e. age of the decision-maker, availability of off -farm income). Property rights and the non-excludable nature of commonage are known to affect land management (Ostrom, 2000). Consequently, information on the number of active shareholders, size of the commonage and size of private land was also sought.

4.5 Results

4.5.1 Results of Stochastic biodiversity frontier model

OLS regression does not provide estimates of environmental efficiency but coefficients may be used to compare results with a fixed effect model. Sometimes OLS residuals do exhibit a negatively skewed distribution. In this case, OLS parameter estimates are inconsistent and the use of a stochastic frontier model may be appropriate. Parameters estimates of OLS regression and a fixed effects stochastic biodiversity frontier are presented in Table 15. It appears that the results of these biodiversity regressions are quite similar except in a few significant cases.

The results presented in Table 15 indicate that the impact of stocking rate on biodiversity is nonlinear. Effects of stocking rate on habitat biodiversity have two components: substitution and complementary effects. The substitution effect of stocking rate on biodiversity is negative (as it is captured in the linear coefficient) whereas the complementary effect is positive and is reflected in the quadratic coefficient (as a significant variable).

In a fixed effects stochastic biodiversity model, land fragmentation (measured by the number of plots) has a positive effect on biodiversity. In contrast, habitat fragmentation (measured in terms of the log to perimeter to area ratio) has a significant and negative effect on biodiversity. Biodiversity declines as habitat fragmentation increases. Thus, a negative and significant value of habitat fragmentation is associated with habitat loss.

Notably, the average body size of marketed animals is negatively correlated with biodiversity. However, subsidy has no significant impact on biodiversity. Similarly, the average age of marketed animals have no influence on biodiversity. The impact of land size is minimal because the biodiversity indicator was measured separately for commonage and private lands. Thus controlling for common property rights may be an important factor in reducing the correlation between land size and biodiversity.

Table 15. Parameter estimates of biodiversity frontier model

Explanatory variables	OLS		Fixed effect	
	Coefficient	Std error	Coefficient	Std error
Dependent: Biodiversity				
Inputs				
Ln (Stocking rate)	-0.0138	0.0239	-0.0072	0.0236
Number of Breeds	0.0101	0.0112	0.0097	0.0110
Ln (Labour)	0.0070	0.0239	0.0009	0.0236
Ln(Purchased Feed)	0.0190	0.0199	0.0223	0.0196
Ln(Land)	0.0104	0.0216	0.0021	0.0215
Ln (Stocking rate) x Ln (Stocking rate)	0.0109*	0.0070	0.0154**	0.0071
Other factors				
Average body size of animals	-0.0790**	0.0414	-0.0725*	0.0407
Average age of animals	0.0012	0.0011	0.0010	0.0011
Ln(Subsidy)	0.0147	0.0304	0.0170	0.0298
Number of plots	0.0425***	0.0042	0.0437***	0.0042
Ln(PA) (Perimeter to Area ratio)	-0.0953***	0.0185	-0.1661***	0.0328
Constant	-0.6785***	0.2493	-0.9828***	0.2717
σ_u			0.1193	
σ			0.1920	
Rho (λ)			0.2785	
H_0: Random effect H_1: Fixed effect			$F(1,159)=6.72$	
N	175		175	
F	16.1***		17.1***	
R^2	0.52		-	

Notes: Level of significance, ***=p<1%, **=p<5%, *=p<10%

The marginal effect of stocking rate as well as body size on biodiversity is negative in all livestock enterprises as shown in Table 16. The highest negative marginal effect is reported for suckler beef enterprises. The marginal effects of subsidy on biodiversity appear to be very small compared with the habitat fragmentation indicator.

Table 16. Estimated marginal effect of biodiversity frontier function

Livestock enterprise	**All N=175**	**Beef N=36**	**Sheep N=51**	**Mixed N=88**
Inputs				
Stocking rate	-0.5422	-1.0440	-0.4616	-0.3836
Number of Breeds	0.0097	0.0097	0.0097	0.0097
Labour	2.94E-05	4.64E-05	3.06E-05	2.18E-05
Purchased Feed	4.13E-05	5.37E-05	5.3E-05	2.95E-05
Land	1.93E-04	2.06E-04	1.64E-04	2.04E-04
Other factors				
Average body size of animals	-0.0725	-0.0725	-0.0725	-0.0725
Average age of animals	0.0010	0.0010	0.0010	0.0010
Subsidy	2.01E-06	2.75E-06	2.10E-06	1.66E-06
Number of plots	0.0437	0.0437	0.0437	0.0437
Perimeter to Area ratio (PA)	-18.8740	-15.5975	-20.1866	-19.4537

In Table 17 the results indicate that the elasticity of biodiversity with respect to stocking rate is inelastic. A 1% increase in stocking rate will decrease biodiversity by 0.54%. It appears that elasticity of biodiversity with respect to labour is very small indicating that biodiversity is not sensitive to changes in labour input.

A 1% increase in subsidy levels will increase biodiversity by only 0.16%. On the other hand, the estimated value of elasticity of biodiversity with respect to body size of marketed animals is quite high. Notably, the body size of marketed animals has a major influence on biodiversity. A 1% increase in perimeter to area ratio will reduce biodiversity by 1.6%. The perimeter area ratio and the number of plots exert an important influence on biodiversity. It is possible to change biodiversity without changing stocking rate by altering animal body size, plot number and plot perimeter. The results indicate that livestock production is constant returns to scale in the west of Ireland.

Table 17. Estimated elasticity of biodiversity using a fixed effect frontier function

Livestock enterprise	All N=175		Beef N=36		Sheep N=51		Mixed N=88	
	Elasticity	Std error	Elasticity	Std error	Elasticity	Std error	Elasticity	Std error
Elasticity of biodiversity								
Stocking rate	-0.3068	0.7977	-0.5925	1.3906	-0.2743	0.4059	-0.2088	0.5967
Number of Breeds	0.2346	0.3441	0.2029	0.2164	0.1163	0.0839	0.3161	0.4457
Labour	0.0086	0.0103	0.0112	0.0160	0.0072	0.0042	0.0084	0.0098
Purchased Feed	0.2134	0.2561	0.2765	0.3968	0.1786	0.1049	0.2078	0.2431
Land	0.0201	0.0241	0.0260	0.0374	0.0168	0.0099	0.0196	0.0229
Other factors								
Average body size of animals	-1.2015	1.7088	-1.6070	2.1760	-0.2521	0.1883	-1.5858	1.7801
Average age of animals	0.5901	0.8487	0.6241	0.7928	0.2030	0.1963	0.8005	1.0161
Subsidy	0.1627	0.1952	0.2108	0.3025	0.1362	0.0799	0.1584	0.1853
Number of plots	1.1949	1.1036	1.5280	1.3242	1.0602	0.8332	1.1367	1.1275
Perimeter to Area ratio	-1.5897	1.9071	-2.0600	2.9554	-1.3305	0.7810	-1.5477	1.8103
Total Return	-0.6737		-1.3796		-0.1387		-0.6949	

4.5.2 Results of stochastic profit frontier model

Parameters estimates of the OLS and fixed effects stochastic profit frontier regressions are presented in Table 18. The results of OLS and fixed effects regressions produce quite different results. The impact of biodiversity on livestock profitability appears to be nonlinear. Livestock profitability increases with biodiversity for low values and as profit reaches a maximum it then decreases for higher values of biodiversity. In the fixed effect regression, the negative coefficient of the quadratic term captures the substitution effect (trade-offs) between livestock profitability and biodiversity. It also indicates diminishing marginal returns to biodiversity. On the other hand, subsidy has an insignificant and positive impact on livestock profitability. Body size of marketed animals is positively related to livestock profitability. Sex ratio[6] of marketed animals has a negative impact on livestock profitability but is insignificant.

[6] Sex ratio is defined as the number of female to male of marketed animals.

Table 18. Estimated parameters of stochastic profit frontier model

Explanatory variables	OLS		Fixed effect	
	Coefficient	Std. error	Coefficient	Std. error
Dependent: Livestock Profit (in '000)				
Biodiversity	5.5969*	4.9259	7.1740*	4.8374
Stocking rate	0.5726**	0.2549	0.6235**	0.2490
Number of Breeds	0.2622	1.4051	1.0255	1.4267
Labour	-0.1114	0.0692	-0.1265*	0.0679
Purchased Feed	-1.5556	2.1806	-1.5567	2.1946
Land	-0.0202	0.0441	-0.0375	0.0443
Biodiversity x Biodiversity	-1.8474	2.0569	-2.6006	2.0368
Stocking rate x Stocking rate	-0.0100*	0.0052	-0.0110**	0.0051
Number of Breeds x Number of Breeds	-0.0508	0.1603	-0.1338	0.1628
Labour x Labour	0.0011**	0.0005	0.0012**	0.0005
Purchased Feed x Purchased Feed	0.2686	0.5769	0.2875	0.5762
Land x Land	-9.80E-06	0.0002	2.39E-05	0.0002
Subsidy	0.0002	0.0002	0.0001	0.0002
Body size of animals	1.1712*	0.6431	2.0171***	0.7390
Sex ratio of animals	-0.3288	0.6536	-0.6606	0.6818
Constant	-1.3887	2.6220	-1.9982	3.3250
σ_u				2.8014
σ				5.7499
Rho (λ)				0.1918
H_0: All u_i=0			F(2,81)=3.29 (p=0.0422)	
N		**99**		**99**
F		**3.37*** **		**3.61*** **
R^2		**0.38**		**-**

Notes: Level of significance, ***=p<1%, **=p<5%, *=p<10%

The overall marginal effect of biodiversity on livestock profit is positive (see Table 19). Biodiversity has a positive and high marginal effect on livestock profitability relative to other livestock inputs. Since livestock profit is expressed in terms of thousands of euros, the value of biodiversity (the average shadow price) is estimated to be €5,023 per household per year. This value of biodiversity may be taken as one of the major findings of the study. Labour has a negative marginal impact on livestock profit in suckler beef and sheep enterprises but is positive in the mixed enterprise.

Table 19. Estimated marginal effects of stochastic profit frontier by livestock enterprises

Livestock enterprise	All	Beef	Sheep	Mixed
Biodiversity	5.0226	5.5038	4.7766	4.9234
Stocking rate	0.4426	0.4726	0.5160	0.3837
Number of Breeds	0.3229	0.4793	0.5956	0.0807
Labour	-0.0089	-0.0434	-0.0230	0.0172
Purchased Feed	-0.8770	-0.9280	-1.1646	-0.6800
Land	-0.0344	-0.0361	-0.0336	-0.0340
Subsidy	0.0001	0.0001	0.0001	0.0001
Body size of animals	2.0171	2.0171	2.0171	2.0171
Sex ratio of animals	-0.6606	-0.6606	-0.6606	-0.6606

The estimated values of elasticity of profit with respect to changes in inputs and other factors are presented in Table 20. The results indicate that the elasticity of profit varies significantly with livestock enterprise. The elasticity of profit with respect to changes in biodiversity is negative and inelastic in the suckler beef and sheep enterprises but it is positive in the mixed grazing enterprise. A 1% increase in biodiversity may lead to 0.1% decrease in profit suggesting that enhancing biodiversity may induce greater costs which will end up reducing profit.

The negative elasticity of breeds indicates that a change in the number of breeds may reduce long-run livestock profit. However, in the short-run increasing the number of breeds has no significant impact on livestock profit. In an effort to improve productivity and profit, many farmers have replaced traditional livestock breeds with higher yielding breeds. Ruto et al, (2008) indicate that the loss of traditional livestock breeds may result in the loss of an important genetic resource as a variety of genetic traits adapted to local conditions gradually becomes less common in the livestock population.

Labour is significantly correlated with livestock profit. The elasticity of profit with respect to changes in labour indicates that a change in labour input may affect profit negatively in sheep enterprises but positively in suckler beef enterprises. Hadley, (2006) has also found a negative elasticity for labour for UK beef and sheep farms. An increase in purchased feed does not increase profitability for the mixed farm enterprise. Since feed production is a labour intensive activity, local hay production may be a necessary condition to increase profit in the region.

Table 20. Elasticity of profit with respect to inputs and other factors

Livestock enterprise	All		Beef		Sheep		Mixed	
	Elasticity	Std error	Elasticity	Std error	Elasticity	Std error	Elasticity	Std error
Factors								
Biodiversity	-0.1196	3.0864	-0.5654	2.5389	-0.3773	3.8579	0.2615	2.8360
Stocking rate	-0.4192	5.1274	-1.8702	5.5817	-0.4841	5.6186	0.3605	4.4922
Number of Breeds	-0.1559	1.6889	-0.1737	1.9541	-0.0819	1.9289	-0.1909	1.4066
Labour	-0.1125	3.564	0.4171	3.8867	-0.2708	4.0536	-0.2887	3.1088
Purchased Feed	0.0371	1.4214	0.2714	1.7737	0.0451	1.5619	-0.0873	1.1201
Land	1.1220	9.2767	0.4057	2.0708	3.5605	16.6549	0.0351	3.6454
Subsidy	-0.2184	2.5715	-0.6487	2.1587	-0.3031	3.613	0.0517	1.9800
Body size of animals	0.5471	5.1932	-0.6198	5.1860	0.3220	4.4748	1.2770	5.5651
Sex ratio of animals	0.2025	3.0713	1.1124	3.1503	0.2754	3.1586	-0.3055	2.9290
Total	0.8831		-1.6711		2.6859		1.1135	

The elasticity of profit with respect to a change in subsidy is negative and inelastic in suckler beef and sheep enterprises. An increase in subsidy in these enterprises will not bring additional profit. The elasticity of profit with respect to changes in stocking rate is negative in suckler beef and sheep enterprises but positive in the mixed enterprise. The elasticity of profit with respect to land size is highly elastic and positive in sheep enterprise indicating land is relatively abundant as a factor of production and an important input for the enterprise. On the other hand, livestock profit is inelastic with respect to land size for suckler beef thus additional land will not result in a proportional increase in profit.

The results of the analysis indicate that the elasticity of profit is very sensitive to body size and sex- ratio of marketed live animals. A 1% increase in body size of animals (in TLU) may lead to 1.7% increase in profit in the mixed enterprise. The elasticity relationship between animals' body size and profit appears to be a negative one only for suckler beef. Profit is highly influenced by animal sex ratio predominantly in the beef enterprise. Increasing sex ratio at the moment will increase profit and efficiency in suckler beef enterprise. These results indicate that a change in body size and sex ratio of marketed animals may increase profit. The grand total of all elasticity of profit provides the most viable livestock enterprise. Farming sheep only appears to be the best way to maximize profitability in the upland regions under study.

4.5.3 Determinates of Biodiversity-Related Environmental Efficiency

This section attempts to analyse determinants of environmental efficiency and estimates the correlation between biodiversity-oriented environmental efficiency and profit efficiency. A truncated regression of environmental efficiency on profit efficiency is presented in Table 21. The results indicate that biodiversity-oriented environmental efficiency is negatively correlated with profit efficiency in all livestock enterprises except for mixed livestock. There is a trade-off between biodiversity-oriented environmental efficiency and the profit motive, i.e. farmers' motive for livestock profit leads to a fall in environmental efficiency**.** Leibenstein, (1978) also argue that profit is bounded to be inefficient as a result of excess profit motivation. Similarly, Kelly et al., (1996) and Bailey et al., (1999) investigated the trade-off between farm income and environmental performance of different cropping systems. The findings probably suggest that livestock market may play an important role for the trade-off between biodiversity-oriented environmental inefficiency and profit. Government plays an important role in market regulation and some regulation is required to make markets work. Government may be needed at a minimum to enforce property rights.

The truncated regression also indicates that there are no significant differences in environmental efficiency between part-time and full-time farms. Similarly, off-farm income was not significantly related to biodiversity-oriented Environmental Efficiency (EE) in almost all regressions considered. Subsidy has a significant negative impact on EE only in specialised farms. In specialised livestock farms, EE depends on profit efficiency, breeds, land size, subsidy, and education whereas in mixed farms EE depends on stocking rate, habitat fragmentation and number of plots. Generally, the results suggest that EE is more sensitive to changes in specialized livestock farms than in mixed farms (Table 21).

Environmental efficiency was also regressed on household characteristics such as age and education and their squares. The impact of farmer's age and education on EE was found to be nonlinear. The effect of education on EE is negative for linear coefficient but positive in the quadratic term except in specialised farms. EE is first rising and then falling with increases in age and education. However, farmer's age is not significantly related to EE in mixed farms.

Table 21. A truncated regression of biodiversity environmental efficiency by livestock specialisation

Variables description	All farms		Specialised farms		Mixed farms	
	Coefficient	**Std. err**	**Coefficient**	**Std. err**	**Coefficient**	**Std. err**
Dependent:- Environmental Efficiency						
Profit efficiency	-0.2074**	0.1108	-0.4388***	0.1805	-0.0070	0.0703
Stocking rate	-0.0041**	0.0023	0.0017	0.0048	0.0014*	0.0010
No of Breeds	-0.0101*	0.0067	0.0243*	0.0144	-0.0004	0.0028
Family Labour	0.0004	0.0004	0.0002	0.0007	-0.0002	0.0002
Purchased Feed	-4.07E-06	1.14E-05	-1.3E-05	0.00002	-3.39E-06	4.57E-06
Fertilizer	-1.6E-05*	1.09E-05	2.35E-05	3.14E-05	1.48E-06	4.04E-06
Land size	0.0002	0.0002	0.0006*	0.0004	0.0001	9.21E-05
Part-time farming (1=Yes; 0=No)	0.0087	0.0234	0.0257	0.03281	0.0067	0.0102
Subsidy	1.31E-07	2.15E-06	-5.21E-06*	3.09E-06	-1.44E-06	9.48E-07
Off-farm income	-0.0005	0.0004	0.0002	0.0005	-0.0002	0.0003
Perimeter to Area ratio (PA)	-0.0680	0.6217	-0.4465	0.7550	1.5832***	0.2992
Number of plots	0.0062***	0.0024	0.0205***	0.00378	-0.0043***	0.0010
Age	0.0026	0.0033	-0.0039	0.0043	0.0047	0.0047
Age square	-3.1E-05	3.39E-05	2.99E-05	4.66E-05	-4.1E-05	3.83E-05
Education	-0.1301*	0.1018	-0.4849***	0.1728	0.0144	0.0446
Education square	0.0479**	0.0265	0.1601***	0.0486	-0.0044	0.0119
Cons	0.2568**	0.1311	0.6112***	0.1904	0.0079	0.1499
Sigma	0.0897	0.0080	0.0821	0.0102	0.0284	0.0024
Number of obs.		129		61		68
LR χ^2		29.63**		83.26***		49.11***
Log likelihood		154.77		78.39		145.76

Notes: Level of significance, ***=p<1%, **=p<5%, *=p<10%

The main difference between truncated regression presented in Table 22 and the previous livestock specialisation regressions is mainly the inclusion of the habitat quality[7] variable shown in row 8 of Table 22. The results indicate that habitat quality is the most influential factor affecting EE on commonage lands. As expected, habitat quality has a positive effect on EE. In commonages, family labour has significantly enhanced environmental efficiency. In commonage farms, environmental efficiency significantly depends on profit efficiency, breeds, labour, fertilizer, habitat quality, part-time farming, off-farm income, age and education. In these farms, EE is negatively correlated with breeds, fertilizer and off-farm income.

In private farms, on the other hands, EE significantly depends on livestock profit, stocking rate, land size, and subsidy. In these farms, EE is negatively correlated with increase in livestock profit and subsidy. In private farms, profit efficiency and competitiveness, land size and subsidy are the driving force behind EE. Land size has a positive and significant impact on EE in private lands. On other hand, land size has no significant impact on EE on commonage farms. Increasing the number of breeds has a negative impact on EE in commonages. This might indicate that a shift to commercial breeds will affect EE. Results indicate that a negative relationship between environmental efficiency and use of fertilizer in the pooled regression.

[7] Habitat quality classification is based on the level of grazing damage and destocking rate of Irish Commonage Framework Plan (Conaghan et al., 2001). Private farm land is not included in this data set since GIS data set is confined to commonage only.

Table 22. A truncated regression of biodiversity environmental efficiency by property rights

Variables description	All lands		Commonage lands		Private lands	
	Coefficient	**Std. err**	**Coefficient**	**Std. err**	**Coefficient**	**Std. err**
Dependent:- Environmental Efficiency						
Profit efficiency	-0.2074**	0.1108	-0.1100*	0.0704	-0.0298**	0.0172
Stocking rate	-0.0041**	0.0023	0.0011	0.0017	0.0017***	0.0004
Number of Breeds	-0.0101*	0.0067	-0.02529***	0.0056	0.0008	0.0010
Family Labour	0.0004	0.0004	0.0012***	0.0003	-4.9E-05	5.68E-05
Purchased Feed	-4.07E-06	1.14E-05	3.06E-06	9.27E-06	-3.02E-06	1.85E-06
Fertilizer	-1.6E-05*	1.09E-05	-1.51E-05*	8.74E-06	-7.20E-07	1.66E-06
Land size	0.0002	0.0002	-0.0002	0.0002	0.0008***	8.31E-05
Habitat quality	-	-	0.0001***	2.45E-05	-	-
Part-time farming (1=Yes; 0=No)	0.0087	0.0234	4.25E-02**	1.78E-02	0.0044	0.0037
Subsidy	1.31E-07	2.15E-06	1.03E-06	1.72E-06	-8.25E-07**	3.30E-07
Off-farm income	-0.0005	0.0004	-0.0006**	0.0003	-2.2E-05	0.00007
Perimeter to Area ratio (PA)	-0.0680	0.6217	2.7529	2.7585	1.0547***	0.1415
Number of plots	0.0062***	0.0024	0.0142***	0.0038	-0.0057***	0.0004
Age	0.0026	0.0033	0.0029*	0.0022	0.0008*	0.0005
Age square	-3.1E-05	3.39E-05	-4.1E-05**	2.32E-05	-7.75E-06*	5.24E-06
Education	-0.1301*	0.1018	-0.1917***	0.0804	-0.0136	0.0166
Education square	0.0479**	0.0265	0.0610***	0.0209	0.0024	0.0044
Cons	0.2568**	0.1311	0.1745**	0.0883	0.1395***	0.0219
Sigma	0.0897	0.0080	0.0453	0.0049	0.0125	0.0010
Number of obs.		129		56		73
LR χ^2		26.63**		382.49***		314.61***
Log likelihood		154.77		97..78		216.45

Notes: Level of significance, ***=p<1%, **=p<5%, *=p<10%

4.5.4 Average environmental efficiency and profit efficiency

Most of the profit efficiency comparisons between mixed and specialized farms are based on traditional inputs (stocking rate, breed, labour, feed, and land) and outputs (beef, lamb) in which the technology is assumed to be different. The issue here is that environmental efficiency scores associated with different livestock enterprises may not be the same. Thus, an appropriate test of equality of average efficiencies across different livestock enterprises (Specialised sheep, cattle only, and mixed grazing) is required.

Average profit efficiency and environmental efficiency estimates of livestock enterprises is presented in Table 23. Profit efficiency is lower than environmental efficiency in all livestock enterprises. In all cases, mixed enterprises have higher environmental and profit efficiencies.

Table 23. Estimated biodiversity oriented profit efficiency and environmental efficiency by livestock enterprises

Biodiversity –-oriented	**All**	**Beef**	**Sheep**	**Mixed**
Environmental efficiency (%)	**84.3**	**83.7**	**83.6**	**85.0**
Profit efficiency (%)	**63.8**	**67.1**	**59.0**	**64.7**

In what follows, we explore whether livestock enterprise type or the nature of property rights has any influence on EE. Levene's (1960) test for equality of environmental efficiency scores indicates that there is no significant difference in biodiversity environmental efficiency between livestock enterprises. The implication is that agri-environmental policy is likely to have an equal impact on the level of environmental efficiency of farmers in all livestock enterprise types. However, our findings reveal that for property rights, Levene's test for equality on environmental efficiency indicates that there is indeed a significant difference in environmental efficiency between private and commonage farms. Similarly, commonage lands have higher environmental efficiency than private lands (Table 24). Therefore, any policy efforts to preserve environmental efficiency should afford priority to commonages. Levene's test for equality of profit efficiency scores indicates that there is a significant difference in profit efficiency between livestock enterprises at 5% level of significance. The implication is that although profit efficiency scores appear to be very close in values, Levene's statistical test rejects the notion of equal profit efficiency between livestock enterprises. High profit inefficiency therefore exists in sheep only enterprises and elimination of this inefficiency will result in improved livelihoods to Irish upland farmers.

Table 24. Estimated biodiversity oriented environmental efficiency by property rights

Land property right	**All**	**Private Lands**	**Commonage lands**
Biodiversity- oriented Environmental efficiency (%)	**84.3**	**83.2**	**85.8**

I now turn to the conclusions and recommendations in chapter five.

Chapter 5: Conclusions and Recommendations

5. 1 Conclusions

Biodiversity provide human-beings with food, fodder, bioenergy and pharmaceuticals and is essential to human well-being. Ecosystem services provided by biodiversity include pollination, biological pest control, maintenance of soil structure and fertility, nutrient cycling and hydrological services. Research assessments indicate that the value of these ecosystem services to agriculture is enormous and often underappreciated. Lack of markets is one of the main reasons for concern over the inadequate provision of ecosystem services. To make better decisions about use and management of biodiversity, economic valuation of biodiversity is necessary.

5.1.1 Chapter 2

A key aim of the first empirical study was to identify key drivers that affect the probability of land abandonment on private land and commonage in the Irish uplands. These include the impact of off-farm income and subsidies on the probability of land abandonment. The results of the ordered probit regressions indicate that off-farm income was positively correlated with the probability of land abandonment. Switching to part-time farming may lead to land abandonment. These findings indicate that off-farm activities compete for farming time and there is a trade-off between household time spent in off-farm and on-farm activities. Subsidies have significantly and negatively impacted on land abandonment. Subsidies help sustain livestock farming in areas where land abandonment is a critical problem. A possible explanation is that support payments increase the viability of farming and reduce the probability of land abandonment.

Household characteristics such as age and education are important factors affecting land abandonment. Farmer's age was found to be negatively and significantly correlated with the probability of land abandonment in all livestock enterprises. Land abandonment increases with younger farmers. Education was positively and significantly correlated with land abandonment. Older farmers often have little education and few opportunities to work in off-farm activities, thus forcing them to maintain livestock farming on marginal land. However, education was found to have less explanatory power than farmer's age as a reason for land abandonment.

The findings indicate that livestock management and common property rights are the main determining factors affecting land abandonment in the west of Ireland. Mixed livestock systems have significantly reduced the probability of abandonment on both private land and commonages. This shows that mixed livestock management was less vulnerable to the risk of land abandonment. Mixed livestock management was found to be a suitable management regime for the maintenance of grassland habitats in the west of Ireland. It is interesting to note that the results indicate that mixed grazing reduces the risk of land abandonment more in disadvantaged areas where land abandonment is an important phenomenon. It is important to gear government payments towards the promotion of mixed grazing.

The average predicted probability of land abandonment is 7% at household level; however it varies within livestock management regime and property right regime. The predicted future probability of land abandonment is highest for suckler beef enterprises, at approximately 12%, whereas abandonment is lowest in mixed livestock enterprises at 4%. A mixed livestock business may play an important role in reducing land abandonment and could be used as a possible restoration management strategy in commonage lands.

The ordered probit regression indicates that farm labour and farm income are found to reduce land abandonment while livestock costs are among factors that increase land abandonment. Efficient utilization of pasture resource and hay making are the principal determining factors for the sustainability of livestock production in the commonage. The production of haymaking should be given a priority. It is important to note that without proper pasture management, livestock production is not sustainable in upland regions. Farmer's own production of hay may increase the viability of livestock farming. In commonage marginal lands, there is an opportunity for hay making and pasture expansion where extensive areas of abandoned grassland could be available at relatively low economic cost. A policy option could be to initiate a hay meadow area payment scheme. This payment would be to exclusively cover production losses and provide an incentive to efficient farmers in the region.

Although livestock products are high in value, livestock revenue still generates a very low income due to low prices at the primary livestock markets in the west of Ireland. Increasing market prices and reducing feed costs may increase farm income and prevent land abandonment. Continued and better integration of farm households into the livestock economy may successfully raise incomes and thus support an agricultural presence and reduce land abandonment. Policies related to increasing farm income, increasing price of livestock at primary market, market niche creation, meat quality improvement and product value added schemes are important in reducing land abandonment. Measures aimed at meat products, especially high quality products, and services related to agricultural land such as tourism, need to be considered alongside environmental land management measures. Policy makers need to pay attention to local livestock breeds. Long-term established quality assurance schemes of highland beef and Connemara lamb will provide a comparative advantage for labeling livestock products in the west of Ireland.

5.1.2 Chapter 3

The main research aim of the second empirical chapter was to establish the effects of habitat fragmentation on biodiversity and livestock productivity in relation to livestock management regimes that involve private and commonage land in the west of Ireland. Habitat fragmentation involves the breaking up of a habitat or ecosystem into smaller parcels. The results indicate a strong negative relationship between habitat fragmentation and biodiversity on commonages for both sheep grazing and mixed grazing systems. Habitat fragmentation is found to be the single most critical threat to biodiversity loss in upland Irish commonages. This effect is most pronounced for specialised sheep production. However, there are differences in livestock enterprise type. The regression results suggest that habitat fragmentation has no significant impact on biodiversity for the mixed grazing system on private lands. It is interesting to note that similar results were found to previous studies. For example, ecologists often suggest that habitat fragmentation leads to reduced biodiversity (Fahrig, 2003), plant diversity (Chapin et al., 2000) and species richness (Davis & Margules, 1998). Habitat fragmentation may have a variety of negative consequences, overall loss of habitat and a significant negative impact on biodiversity (Wiens, 1995; Dirzo & Raven, 2003; Fahrig, 2003).

The elasticity of biodiversity with respect to habitat fragmentation is highly inelastic, particularly for specialised sheep production. On commonage land a 1% increase in habitat fragmentation (perimeter to area ratio) leads to a 2% decrease in habitat biodiversity for sheep enterprises. Study findings are in broad agreement with a number of previous studies that indicate a negative relationship between habitat fragmentation and biodiversity (Smith et al., 1996; Andrén, 1997; Jeffrey & Mitchell, 2000; Hobbs, 2001; Fahrig, 2003).

A trend towards specialised production has contributed to the rate of habitat fragmentation. For commonages, findings show that habitat fragmentation is positively correlated with livestock productivity in the pooled regression. In particular, a highly significant positive relationship for sheep enterprises is found. The study also shows that biodiversity is positively correlated with livestock productivity across all habitats.

The use of different breeds has a negative impact on livestock productivity on commonages particularly for specialised sheep production. For sheep on commonages the relationship between biodiversity and productivity is non-linear. At relatively low stocking rates higher levels of biodiversity increase productivity. At high stocking rates there is a negative relationship between biodiversity and productivity. Our findings are consistent with those of Grime, (1973) and Grace, (1999) who also observed a hump-shaped relationship between productivity and biodiversity for grasslands. The findings therefore show a potential trade-off between biodiversity and productivity. This effect is most significant for specialised sheep production. This pattern is not observed for mixed grazing. Instead a linear relationship between biodiversity and productivity for mixed grazing is found. Biodiversity increases linearly with productivity, indicating less of a trade-off between biodiversity and productivity in a mixed livestock system.

A further aim of this study was to determine factors affecting biodiversity loss and livestock productivity under different livestock management regimes and under different land property rights. Habitat fragmentation is one of the most significant causes of biodiversity loss in the Irish uplands. For specialised sheep production, larger farms and farms with a high labour intensity were negatively associated with biodiversity. Habitat quality was positively correlated with biodiversity for mixed grazing on commonage. The number of shareholding plots (fragmentation) was also positively associated with biodiversity. This effect is consistent across all enterprise types and property right regimes. Stocking rates have a positive effect on biodiversity for sheep production on commonage lands. On private lands (except for cattle enterprises), a negative relationship between biodiversity and stocking rates was found. The role of animal characteristics was most pronounced on private land where the results showed a highly significant negative relationship between animal body size and biodiversity. The effects were more pronounced for sheep production.

Another aim of this study was to evaluate the role of agri-environmental subsidies on biodiversity and livestock productivity. On commonage, subsidies have positively influenced biodiversity in sheep enterprises while subsidies have no impact on biodiversity in mixed enterprises. On private land, the results indicate that subsidies are significantly and positively correlated with biodiversity both in suckler beef and mixed enterprises. On commonage, subsidies have a negative impact on livestock productivity in sheep and mixed livestock enterprises. On private land, subsidies have a negative impact on livestock productivity but no significant impact on suckler beef and mixed enterprises. Subsidies were found to be positively and significantly correlated with biodiversity and livestock productivity in specialized livestock grazing management. Subsidies had no significant effect on biodiversity and livestock productivity in mixed grazing management.

A final aim of chapter three was to identify the livestock enterprise regimes that are less susceptible to the risk of habitat loss. Mixed livestock grazing was found to be less susceptible to the risk of habitat fragmentation. Mixed grazing management was found to be better than sheep production for biodiversity and livestock productivity in both private and commonage lands.

Mixed grazing management provides the highest total return on biodiversity and livestock productivity compared with specialized livestock grazing management. The biodiversity on commonage is better than the private property resources. With respect to policy recommendations, agri-environmental schemes should promote mixed grazing to enhance biodiversity conservation in the west of Ireland. The scheme should be modified to provide additional incentives for mixed grazing in order to achieve better biodiversity conservation and halt biodiversity losses.

5.1.3 Chapter 4

Changing land use practices have imposed real pressures on natural habitats and biodiversity in managed upland landscapes of Ireland in recent years. This is a subject of some concern to policy makers and the public at large because it is recognised that biodiversity loss could diminish the options open to future generations. Joint production of marketable produce and non-market services such as biodiversity in managed upland landscapes constitutes an important social goal as well as meeting CAP objectives. However, for many farmers the economic trade-offs between producing goods for the market and meeting conservation goals are all too real and land managers often pose the question: how does biodiversity affect farm profitability and is there a financial cost to biodiversity provision? This chapter bears these questions in mind and explores the role of biodiversity in the functioning of managed upland agro-ecosystems under private and common property regimes.

Thus, one of the aims of this chapter was to examine the relationship between biodiversity and profitability. The relationship between biodiversity related profit efficiency and environmental efficiency is examined comparing specialised versus mixed grazing farms; private versus commonage farmers; and full-time versus part-time farmers in the west of Ireland.

Results from the OLS and fixed effects stochastic profit frontier regressions reveal a significant positive relationship between profitability and biodiversity. However, the impact of biodiversity on profitability is non-linear. Diminishing marginal returns to biodiversity were observed. A key finding was that the overall marginal effect of biodiversity on livestock profitability is high relative to other livestock inputs. The average shadow price of the value of biodiversity was estimated to be €5,023 per household per year.

Possible trade-offs between profit efficiency and environmental efficiency was explored in a two-limit truncated regression model. The results show that biodiversity-oriented environmental efficiency is negatively correlated with profit efficiency in both specialised livestock enterprises and private land. There are differences in farm type. The results suggest that environmental efficiency is more sensitive to changes in farm management in specialized livestock farms than in mixed grazing farms.

A second objective was to examine whether the part time status affects either environmental efficiency or profit efficiency. The truncated regression indicated that there was no significant difference in environmental efficiency between part-time and full-time farms. Both off-farm income and part-time farming were not significantly related to biodiversity environmental efficiency.

A third aim of the study was to examine key determinants of farm profitability. The elasticity of profit with respect to changes in biodiversity was negative and inelastic for suckler beef and sheep enterprises but positive for the mixed grazing units. It was found that subsidies have no significant impact on biodiversity in a fixed stochastic regression model. The findings suggest that the marginal effects of subsidies on biodiversity are very small and inelastic. An increase in subsidies will not generate additional profit. Again differences in farm type are important; the elasticity of profit with respect to farm subsidies is positive for mixed grazing. The estimated value of elasticity of biodiversity with respect to body size of marketed animals is highly elastic. High values of body size elasticity indicate that biodiversity is highly sensitive to changes in body size of marketed animals.

The findings indicate that there is an inverse relationship between Environmental Efficiency (EE) and Profit Efficiency(PE). Subsidies on commonage have an insignificant impact on EE. In specialised livestock farms, EE depends on profit efficiency, breeds, land size, subsidy, and education whereas in mixed farms EE depends on stocking rate, habitat fragmentation and number of plots. In specialised farms, EE is negatively correlated with livestock profit and subsidy whereas in mixed livestock farms EE is negatively correlated with only number of plots.

In commonage farms, environmental efficiency significantly depends on profit efficiency, breeds, labour, fertilizer, habitat quality, part-time farming, off-farm income, age and education. In these farms, EE is negatively correlated with breeds, fertilizer and off-farm income. There is a significant difference in EE between full-time and part-time farming.

In private farms, on the other hands, EE significantly depends on livestock profit, stocking rate, land size, and subsidy. In these farms, EE is negatively correlated with increase in livestock profit and subsidy. In private farms, profit efficiency and competitiveness, land size and subsidy are the driving force behind EE.

From a policy perspective the interrelationship between the market and regulation is worthy of comment given the role of CAP farm subsidies in supporting farm incomes in the study area. Livestock markets may play an important role in the trade-offs between biodiversity oriented environmental efficiency and farm profitability. On the one hand, a number of key market driven variables such as specialised sheep production, body size and sex ratio of marketed animals appear to influence farm profitability. If these exert a strong influence on farm decision making, farmers may be expected to specialise in sheep production to stock larger animals and intensify production. The findings of this study support this trend; the results reveal that farming sheep only is the best way to maximise profitability. Biodiversity conservation, on the other hand, is promoted by a less specialised mixed grazing system that employs less production inputs. Findings from the two-limit truncated regression model showed that for specialised beef and sheep units promoting profit efficiency has a negative impact on environmental efficiency.

However, this is not the case for mixed grazing systems. The introduction of decoupling and agro-environment schemes under the CAP reforms were designed to support farm incomes and enhance the environment in its visual and amenity aspects (DAFM, 2004). However, few studies have shown whether this has been achieved (Feehan et al., 2005). This study finds that subsidies have no impact on biodiversity; profitability in specialized sheep and beef units or in mixed grazing are not improved. Farm subsidies have a significant negative impact on environmental efficiency for specialised beef and sheep units. Two of the goals of the CAP concerned with farm income support and the environment are not being realised for these specialised production units.

In summary, a key message from this analysis for policy makers concerned with biodiversity provision is that there is a need to consider the characteristics of marketed animals, including how farm decisions are affected by livestock market signals, plot level variables and the type of farm system. In particular the role of mixed grazing needs to be considered rather than focusing mainly on stocking rate and conventional inputs.

5.2 Key findings

The main findings of this research are:

- The ordered probit regressions indicate that off-farm income was positively correlated with the probability of land abandonment. Switching to part-time farming may lead to land abandonment.
- Subsidies have significantly and negatively impacted on land abandonment. Subsidies help sustain livestock farming in areas where land abandonment is a critical problem. A possible explanation is that support payments increase the viability of farming and reduce the probability of land abandonment.
- Mixed livestock systems have significantly reduced the probability of abandonment in both private lands and commonages. This shows that mixed livestock management was less vulnerable to the risk of land abandonment.

- Using a pooled regression, habitat fragmentation was negatively correlated with biodiversity for commonage land. On the other hand, the biodiversity regression results suggest that edge effect has no significant impact on biodiversity in the mixed grazing system. The mixed livestock grazing regime is less susceptible to the risk of habitat loss.
- Subsidies have positively influenced biodiversity in sheep enterprises but have no impact on biodiversity in mixed enterprises.
- In mixed enterprises, habitat quality is positively correlated with biodiversity in commonage lands.
- There is an inverse relationship between environmental efficiency and profit efficiency.
- There is no significant difference in environmental efficiency between full-time and part-time farming.
- Subsidies have an insignificant impact on environmental efficiency on commonage land.

5.3 Limitations of the research

The research is based on a cross sectional analysis of data from two different surveys. The primary data used is from a sample of 100 observations. The study also used a survey of 283 observations collected in 2004. The main limitation of the study is the small sample size. Thus, this section will review the literature to address the issue of small sample size in relation to goodness-of-fit.

A number of arguments are available to deal with small sample sizes. Larger samples are preferred because they tend to minimize the probability of errors, maximize the accuracy of population estimates, and increase the goodness-of-fit. In statistics, with known finite sample properties, the sampling distributions of the statistics change as a function of sample size. It is also apparent, when sample size large enough there will be more degrees of freedom in the analysis. When samples are large, more information is available and, therefore, more confidence can be expressed for the model to fit the population process. Hence, F statistics and t statistics explicitly adjust according to the sample-size differences.

In Monte Carlo studies, data are generated at various sample sizes and can be compared with the results of regressions. Monte Carlo studies can also help in determining appropriate sample sizes. For instance, Browne (1968) investigated the quality of solutions produced by different factor analytic methods. Browne found that solutions obtained from larger samples showed greater stability and more accurate recovery of the population loadings. Browne (1974) also suggests the influence of sample size is reduced when factor loadings were higher. Pennell (1968) examined the effects of sample size on stability of loadings and found that the effect diminishes as communalities of the variables increased. Based on such studies, a wide range of recommendations regarding sample size in regression analysis has been proposed. In a comprehensive Monte Carlo study, Browne (1968) examined the effects of sample size on various estimators in the factor analysis model. Guilford (1954) argued that N should be at least 200, and Cattell (1978) claimed the minimum desirable N to be 250. Comrey and Lee (1992) offered a rough rating scale for adequate sample sizes in factor analysis: 100 = poor, 200 = fair, 300 = good, 500 = very good, 1,000 or more = excellent.

The influence of small sample size may introduce inaccuracy and variability in parameter estimates, while the influence of model error is to introduce lack of fit of the model in the sample. In any given sample, these two issues produce lack of model fit and error in parameter estimates. Archer & Jennrich (1976) showed using Monte Carlo approaches that as the sample size increases, standard errors decrease. Similarly as sample size increases, the variability in factor loadings across repeated samples will decrease. Comrey & Lee (1992) placed the sample size question into the context of the need to make standard errors of regression coefficients adequately small so that ensuing factor analyses of those correlations would yield stable solutions. A variety of rules have been suggested for determining the sample size required to produce a stable solution when performing a factor or component analysis. Results of Tanaka (1987) suggest that small sample size may be more problematic when non-normal estimation methods are used. Tanaka's analysis shows that maximum-likelihood (ML) estimates were least affected in comparison to a variety of possible non-normal alternative samples.

Approaches other than Monte Carlo have been suggested to deal with small sample sizes. These approaches look for alternative goodness-of-fit indices that compare the observed data with the hypothesized model. Geweke and Singleton (1980) looked at the characteristics of the likelihood ratio chi-square test statistic assessing model fit in ML factor analysis. Using samples of size 10, 30, 100, 150, and 300, they found that the fit statistic behaved well in a sample of size after 30. The most exhaustive Monte Carlo examination of the effects of sample size on latent variable structural equation models was conducted by Boomsma (1983), who concluded that the ML estimator in latent-variable structural equation models broke down in samples of less than 100 subjects. Discouraging to many applied econometric users who face constraints of funding to acquire large sample data set, Boomsma (1983) suggested that the best modelling work required samples of at least a size of 200.

Another key limitation that prevents complete understanding of the dynamic nature of livestock productivity and biodiversity relationships is lack of panel data. In other words, potential changes of impacts, trade-offs and factors that affect biodiversity loss over time are not captured in this thesis.

All of the above literature shows the characteristics of small sample size and how it affects the precision of the estimations. Despite these limitations, the broad conclusions of the study still hold and they can be used for policy purposes and represent the study regions (Galway and Mayo in Ireland). As an additional check, the summary statistics of the 2004 survey dataset (with 282 observations) and 2010 survey dataset (with 100 observations), were compared and found to be almost the same. Larger sample size can only add more variables and strengthen the general findings. However, large sample size will significantly reduce the standard error of the beta coefficients and increase the precision of future estimation.

5.4 Further Research

There are a number of future research possibilities emanating from this thesis. Future research may increase the sample size to 200 observations and analyse if there is any change in the research outcomes. It would also be interesting to address the issue of time and to extend the research by asking the same questions to the same farmers and build a panel dataset. This can be used to investigate the dynamic nature of biodiversity in relation to the relationship between livestock productivity and environmental efficiency.

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Appendix A. The Survey material

Commonage Farm Survey

The recent changes in agri-environmental schemes such as REPS along with the wide scale economic downturn have contributed to significant changes in Irish agriculture farm management. In this context the Department of Economics at the National University of Ireland, Galway in association with Teagasc and the Department of Agriculture, Fisheries and Food are undertaking research to understand some of the issues being faced by farmers as a result of these events.

As a part of this research we intend to identify the current condition of your farm lands and also the management practices involved. We aim to use this information to link farm activity with the provision of public good on a wider scale in order to assess its contribution to society. By assigning a value to the public good produced, we wish to inform policy makers of benefits to society to ensure that the work of the farmers is not undervalued when making policies.

For the research to be valid your co-operation is essential and would be greatly appreciated. The interview will last about 45 minutes to an hour. You can be assured that any information given by you will be treated in the strictest confidence and will not be used by any other agency or for any other purpose.
We would therefore be most grateful if you could spare the time to answer the questions below.

Questionnaire Number: ________ Date: ____ Day____ Month_______

County ____________________ DED: __________________________

Interviewer initial: ____

Time Started: ________________

Time Ended: _________________

A. Private Farm plots and Commonage Characteristics

A1. GIS coordinate of homestead

GIS coordinate North ________________

GIS coordinate East ________________

Altitude ______________________

A2.	How many years have you been farming in this area?	Years

A3.	Are you an active commonage shareholder, i.e do you actually farm the commonage land for grazing?	0= No	1= Yes

		Private (Owned and Rented)	**Commonage**	**Total**
A4.	The total number of private plots and commonages you have operated in 2009?			

A5. Could you please tell us plot level information and area covered in **hectares** by private and commonage land types as described below? **(1 hectare = 2.4 acres)**

HERD ID: ________________________________ **or** **HOLDING ID:** ________________________________

PRIVATE LAND	**Private plots**							
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**
LPIS No. / Land ID								
Area (Hectares)								
*Land use type**								
Habitat type**								
Invasive Alien Species **(0=No 1= Yes)**								
Please indicate condition of the land***								
1=Low Land 2=High Land								
Distance to the main town **Specify Town____________**								
*Ownership*****								

***Land use type:** **1=Homestead; 2=Pasture or forage land; 3=Rough grazing; 4=Improved grazing; 5=Tree planting; 6=Peat cutting 7=Other;**
****Habitat type:** **1= Grassland; 2=Bog land; 3=Heath; 4=Forest; 5=Abandoned/scrub; 6=Other**

*****Condition of the land** **1=Over-Grazed 2=Slightly Overgrazed 3=Properly Grazed 4=Slightly Undergrazed 5=Undergrazed**

*****Ownership:** **1=Owned; 2=Rented in (leased in); 3=Leased out; 4=Shared; 5=Other**

Private plots continued…

PRIVATE LAND	**Private plots**							
	9	**10**	**11**	**12**	**13**	**14**	**15**	**16**
LPIS No. / Land ID								
Area (in Ha)								
*Land use type**								
Habitat type**								
Invasive Alien Species **(0=No 1= Yes)**								
If grassland, please indicate condition of the land***								
1=Low Land 2=High Land								
Distance from Main Town **Specify Town __________**								
*Ownership****								

***Land use type:** **1=Homestead; 2=Pasture or forage land; 3=Rough grazing; 4=Improved grazing; 5=Tree planting; 6=Peat cutting 7=Other;**
****Habitat type:** **1= Grassland; 2=Bog land; 3=Heath; 4=Forest; 5=Abandoned/scrub; 6=Other**
*****Condition of the land 1=Over-Grazed 2=Slightly Overgrazed 3=Properly Grazed 4=Slightly Undergrazed 5=Undergrazed**
*****Ownership:** **1=Owned; 2=Rented in (leased in); 3=Leased out; 4=Shared; 5=Other**

COMMONAGE	**Commonage land**			
	1	**2**	**3**	**4**
Commonage ID				
Commonage Name				
Total Area of Commonage (Ha)				
Your Share in Commonage (%)				
Total Number of Share Holders				
Total Number of Active Share Holders				
Habitat type**				
Invasive Alien Species (0=No 1=Yes)				
Condition of the land***				
1=Low Land 2=High Land				
Distance to Main Town				

****Habitat type: 1= Grassland; 2=Bog land; 3=Heath; 4=Forest; 5=Abandoned/scrub; 6=Other**

*****Condition of the land 1=Over-Grazed 2=Slightly Overgrazed 3=Properly Grazed 4=Slightly Undergrazed 5=Undergrazed**

A6.1 Feed produced from your farm

Please indicate the quantity of supplementary feed produced from private land during the year 2009. **[Indicate Units such as Bales, Tonnes, or Kilograms]**

Type of Feed	Unit	Feed produced Private Plots (UNITS) [Indicate Plot Number]							Amount Sold
1=Silage									
2=Hay									
3=Straw									
6=Other (Specify)									

A6.2 Feed purchased for your farm:

Please indicate the quantity of supplementary feed purchased last year (2009). **[Indicate Units such as Bales, Tonnes, or Kilograms and weight for bales and bags]**

Type of Feed	Feed Purchased				Total Cost
	Unit	Weight per Unit	Amount Purchased	Price per Unit	
1=Silage					
2=Hay					
3=Straw					
4=Concentrate					
5=Cereals					
6=Other (Specify)					

A7. Fertilizer Use:

Please indicate the total amount of conventional inputs applied in all your fields last year (2009).

Conventional Inputs	Unit	Total Amount Applied	Price Per Unit	Amount of Fertilizer Applied in Percentage (%) Private Plot Number [Indicate Plot Number]							
Fertilizer											**Total**
1=Lime											100%
2=Phosphate											100%
3=Nitrogen											100%
4=CAN											100%
10:10:20											100%
18-6-12											100%
7=Others											100%
8=Manure (slurry)											100%
9=Herbicide/Insecticide/Fungicide											100%
10=Other (Specify)											100%

A8. Please indicate the number of tractors you have? ______________

A9. Please indicate the horsepower of the tractors. _________ _________ _________

B. Description of Enterprise/ Livestock Grazing Management

B1. Please indicate how your livestock alternated between your commonage and private lands last year (2009) in terms of numbers and duration?

Livestock Type	Number of Head					
	Total Stock Numbers	Type of Breed		Period on Commonage (months)	Period on Private Land (months)	Period Housed (months)
		Breed	Number			
<u>Dairy Enterprise</u>						
Dairy Cows						
Female Calf Less Than 1 Year						
Replacement Heifers 1-2 Years						
Male Calf Less Than 1 Year						
<u>Suckler Beef Enterprise:</u>						

Suckler Cows						
Female Calf Less Than 1 Year						
Replacement Heifers 1-2 Years						
Heifers 1-2 Years (Fattened)						
Other Heifers Above 2 Years + (Non-suckler cows)						
Breeding Bull						
Male Calf Less Than 1 Year						
Male 1-2 Years						
Male Above 2 Years						
Sheep	**Total**	**Type of Breed**		**Period on**	**Period on**	**Period**

<u>Enterprise</u>:	**Stock Numbers**	**Breed**	**Number**	**Commonage (months)**	**Private Land (months)**	**Housed (months)**
Ewes						
Hoggets Less Than 1 Year						
Rams For Breeding						
Lambs						
<u>Other Enterprises</u>						
Pigs						
Working Horses						
Ponies						
Goats						
Chickens						
Ducks						
Geese						

B2. Please indicate the amount of honey produced and sold in 2009?

No. of bee hives ________ Amount of honey produced __________ Revenue from honey __________€

B3. Compared to 10 years ago (1999), how has your enterprise and management practice changed?

<table>
<tr><th rowspan="3">Livestock Type</th><th colspan="6">Change in grazing Management
(1= Decreased 2= Stayed the Same 3= Increased 4= N/A)</th></tr>
<tr><th rowspan="2">1=Change in stock</th><th colspan="2">Change in breed</th><th rowspan="2">4=Period on Commonage</th><th rowspan="2">5=Period on Private Land</th><th rowspan="2">6=Period housed</th></tr>
<tr><th>2=Rare breed</th><th>3=Indigenous Breed</th></tr>
<tr><td>1=Dairy Enterprise</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>2=Suckler/ / Beef Enterprise:</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>3=Sheep:</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>4=Horse/Ponies /donkey:</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

C. Costs of operation:

C1. Could you please estimate your total farm costs for the following categories in the last year (2009)?

Expenditure Category	Total in €
1=Vet & Medicine	
2=Artificial insemination / bull costs	
3=Ram Costs (for sheep farmers)	
4=Concentrate Feed	Cattle Sheep
5=Purchased hay, silage or straw	Cattle Sheep
6=Seed (Reseeding)	
7=Petrol, Diesel and oil	
8=Machinery maintenance and supplies	
9=Building maintenance and supplies	
10=Fencing costs	
11=Drainage	
12=Payment to Contractors	
13=Hired Labour	
14=Other	

C2. Please provide the number of hired workers, the average time worked per day and number of days worked on your farm by paid/unpaid non-family members **in 2009**.

Description of Work *	Number of Farm Workers	Average Number of Hours Worked	Total Number of Days Worked

*** Work Code: 1=Livestock related feeding and herding; 2=Stone walls; 3=Drainage; 4=Peat production; 5=Silage; Others Please Specify**

D. Market Participation

D1. In the last year (2009) how many livestock did you sell and purchase in the following categories?

Stock Category	Sales			Purchases	
	Number of head Sold	**Average Price/head**	**Place of Sale ***	**Purchases (number)**	**Average Purchase Price**
Dairy Enterprise					
Dairy Cows					
Replacement Heifers (1-2 years)					
Male calf (0-1 years)					
Female calf (0-1 years)					
Suckler / Beef Enterprise:					
Suckler Cows					
Breeding Bull					
Males calf (0-1yrs)					

Stock Category	Sales			Purchases	
	Number of head Sold	Average Price/head	Place of Sale *	Purchases (number)	Average Purchase Price
Males (1-2 yrs)					
Males (2yrs+)					
Female calf (0-1 yrs)					
Female (1-2 yrs)					
Female (2yrs+) (Non-suckler cow)					
Sheep Enterprise:					
Ewes					
Hoggets					
Rams					
Lambs					
Other Enterprises:					

Stock Category	Sales			Purchases	
	Number of head Sold	**Average Price/head**	**Place of Sale ***	**Purchases (number)**	**Average Purchase Price**
Horses					
Goats					
Others					

*1=Local Mart (Specify Mart); 2=Factory; 3=Butcher; 4=Cattle/Sheep Dealers; 5=Other

D2. Which of the following systems best describes your sheep enterprise?

1 = Producing store/unfinished lambs

2 = Producing finished lambs

3 = Others

D3. When did you (aim to) sell your lambs in 2009?

1 = Early lamb production (lambs sold before end of May)

2 = Mid-season lamb production (end of May until September/October)

3 = Late lamb production (between late October and Christmas)

4 = Others

E. Schemes

E1. Did you receive payment under the Single Farm Payment Scheme in 2009?

0= No 1=Yes

E2. If yes, could you tell us the following information regarding entitlements and payment in 2009?

Type of information	**The Total Number of Entitlements**
The number of eligible hectares / land entitlements	_____ **Ha**
	Payment
Total payment received	

E3. What payment did you receive under the Disadvantaged Area Payment Scheme in 2009?

_______________€

Involvement with REPS:

E4.1 Did you participate in the Rural Environment Protection Scheme (REPS) in 2009?

0= No 1=Yes

[If no, SKIP to E13]

E4.2 If 'yes', was it REPS 3 or REPS 4? _______________

E4.3 When will the contract end? Date _____________________

E5. What was your total REPS payment in 2009? **[For the whole year]**

___________________€

E6.1 **If your contract ended before 17th May 2010**, did you sign up for the new Agri- Environmental Option Scheme (AEOS)?

0=No 1=Yes

If No Why not?

____

E6.2 Once your current REPS contract ends, would you be interested in joining the new Agri-Environmental Option Scheme (AEOS)?

0=No 1=Yes 2=Not Sure/Don't Know

If No or Not Sure, why?

____

E7. When did you first join REPS?

____________Year

E8. If you did participate in REPS, please indicate which environmental measures and options you had picked.
Please circle only the measures and options they have picked.

1=Nutrient management (manure and silage application)

2=Grassland and soil management
Options
a=Traditional hay meadows
b=Species-rich grassland
c=Use of clover in grassland swards
d=Use of trailing shoe technology
e=Control of invasive species

3=Protect and maintain watercourses, water bodies and wells
Options
a=Increase water course margin
b=Exclude all bovine access to watercourses
c=Use of planted buffer zone

4=Retain wildlife habitat
Options
a=Creation of new habitat
b=Broadleaved tree planting
c=Nature corridors
d=Farm woodland establishment

5=Maintain farm and field boundaries
Options
a=Hedgerow Coppicing
b=Hedgerow laying
c=New hedgerow planting
d=Additional stone wall maintenance

6=Restricted use of pesticide and fertilizer

7=Establish biodiversity buffer strips surrounding features and historical & archaeological interest

8=Maintain and improve visual appearance of farm and farmyards
Options
a=Traditional Irish Orchards
b=Install bird and bat boxes

9=Tillage crop production
Options

a=Green cover establishment
b=Environmental management
c=Increased arable margins
d=Low input spring cereals
e=Minimum tillage

10=Training in environmental friendly farming practices

11=Maintain farm and environmental records

E9. If you did participate in REPS, please indicate the supplementary environmental measure undertaken?

Supplementary Environmental Measure	**Have you undertaken the measure?** **0=No 1=Yes**
1=Mixed grazing	
2=Traditional Irish Orchards	
3=Conservation of animal genetic resource/Rare Breeds	
4= Riparian Zones (long term set aside 20 years)	
5=Traditional sustainable grazing	
6= Incorporation of clover into grassland swards	
7=Conservation of wild bird habitats 9=Low Input Spring Cereals	
8=Lake Catchments	

E10. Have you changed your stocking rates as a consequence of joining REPS?

0=No 1=Yes

E11. If yes, how much were you asked to reduce your stocking rate by REPS ________________%

E12. Since joining REPS, how have your livestock numbers changed?
Management Change (1= Decreased 2= Remained the same 3= Increased)

Number of Sheep ________

Number of Cattle ________

Involvement with other schemes:

E13. Have you participated in the National Parks and Wildlife Service Scheme (NPWS)?

0= No 1=Yes

If Yes, please indicate when you joined ______________________

If you were in the NPWS in 2009 what was your total payment for that year? __________________€

E14. Did you participate in the Destocking Scheme in 2009?

0= No 1=Yes

E15. If yes, please indicate the total amount of payment received in 2009? ______________________€

E16. Did the Commonage Framework Plan or the Destocking Scheme affect your farming enterprise?

0=No 1=Yes

If 'yes', in what ways?____

E17. If yes, how much were you asked to reduce your stocking rate under the CFP or Destocking Scheme.

_______________ %

E18. Do you have any old hedgerows on your farm?

0=No 1=Yes

If yes, please indicate the length of old hedgerows on your farm ___________________ meters

E19. Did you participate in the Organic Farming Scheme in 2009?

0=No 1=Yes

If you did participate, what was the payment received in 2009? ___________€

E20. Did you participate in the Animal Welfare Recording and Breeding Scheme for your suckler herd in 2009?

0= No 1=Yes

If yes, please indicate the number of suckler cows for which the payments were made ________

E21. Please indicate the amount of payment received in 2009 under the Animal Welfare Recording and Breeding Scheme _________________________

If no, did you participate in 2010?

0= No 1=Yes

E22. Did you participate in the Grassland Sheep Scheme in 2010?

0=No 1=Yes

If no, would you like to participate in the Grassland Sheep Scheme next year?

0=No 1=Yes

E23. In relation to the Grassland Sheep Scheme, please indicate the total number of breeding ewes declared in the 2009 census? ____________

E24. Do you support the electronic tagging system for sheep?

0=No 1=Yes

If no, please indicate the main reasons.
____

____
____

E25. Did you participate in the Lamb Quality Assurance Scheme operated by Bord Bia in 2009?

0=No 1=Yes

If you did participate, what was the payment received in 2009?
___________________€

Condition of the uplands

E26. Have you noticed an increase in scrub (high heather, gorse, bracken) and/or unpalatable grasses on your grazing area(s)?

0=No 1=Yes

E27. Overall, what was the condition of your mountain grazing area 10 years ago?

[Circle only one of the chooses that apply]

1=Over-Grazed	2=Slightly Overgrazed	3=Properly Grazed	4=Slightly Undergrazed	5=Undergrazed

E28. Based on your own experience, historical practices on the commonage have resulted in which of the following outcomes regarding erosion on land and the environment?

1=Complete destruction

2=Severe damage and erosion (up to 50%)

3=Moderate damage and erosion (up to 25%)

4=Perfect Maintenance

5=Other (Please Specify……………………………………………………)

What are the most common types of Scrub Species you've noticed?
____

In your opinion what are primary reasons for Scrub encroachment:__________________________

____ __________________

E29. In your opinion, has the number of hill farmers declined in the last 10 years?

0=No 1=Yes

E30. If 'yes', what are the PRIMARY reasons for the change? **[Circle more than one if applicable]**

1=Age / lack of successors 5=Excessive farm regulations and paperwork

2= High costs of farm inputs 6=Poor prices for farm output

3= Lack of interest in farming 7=Farming becoming too much of a Burden

4=Reduction in farm income 8=Other (Please specify):

____

E31. Please indicate whether the area of **land abandoned** in your farm has changed in the last 10 years?

1= Decreased 2= No change 3=Increased

F. COOPERATION

F1. How important is your commonage land for your livelihood?

1=Not Important	2=Somewhat Important	3=Important	4=Extremely Important	5=Don't Know/Not Sure

F2. Preference for future commonage land use:

Please rank industries in the order of importance. ***Please rank giving 1 for your most preferred option and 2-5 as your least preferred option***.

Industry	Ranking
1=Recreational uses including Hill Walking and Mountain Biking,	
2=Grazing	
3=Forestry	
4=Renewable Energy including windmills	
5=Other industry (specify____________________)	

F3. Would you be willing to cooperate/ coordinate with other farmers in a recreational scheme?

0=No 1=Yes

If no, why not?

____

F4. Would you require payments for joining such a cooperative scheme?

0=No 1=Yes

G. Household Characteristics:

G1. Could you please provide the following information on all those living permanently in your household? (A household is defined as a group of people who live under the same roof and share the same budget.).

Family Status	**Age**	**Education level***	**Farm Participation****	**Ave. Farm labour hours / week**	**Off- farm employment / occupation*****
1. Male					
2. Female					
3. Daughter 1.					
4. Daughter 2.					
5. Daughter 3.					
6. Son 1.					
7. Son 2.					
8. Son 3.					
9. Mother					
10. Father					
11. Grandfather					
12. Grandmother					
13. Grandson					
14. Granddaughter					
15.Other					
16. Other					

Codes:

*** Education Level:**
1= Primary level 2= Secondary level 3=Third Level (College or university).

****Farm participation:**
1= Full-time farmer 2= Part-time farmer 3= Non-participation

***** Off farm occupation:**

0=No off-farm 1=Factory 2=Construction 3=Transport 4=Fishing 5=Tourism

6=Postman 7=Student 8=Retired 9=Health care 10=Farm contractor 11=other

G2. Marital status of household head:

1= Married 2= Single 3= Widowed 4= Separated

G3. What percentage of this income can be attributed to each of the following categories?

(Where offered please make note of the exact income figures)

Source of income	**% of total gross household income**
1=Farm (livestock, peat…)	%
2=Off-farm Employment	%
3=Tourist Activities (Rental accommodation, B&B, golf etc.)	%
4=State Transfers (farm subsidy, pensions, child benefit etc.)	%
5=Other (Specify________________)	%
Total	100 %

G4. In which of the following categories does your total household gross (pre-tax) income lie? Please include all sources of income including farm, off-farm employment, tourism activities and state transfers and any other cash income e.g. private pension etc.

	Total Income Range
1	< 15.000 €
2	15.000 – 30.000 €
3	30.000 – 45.000 €
4	45.000 – 60.000 €
5	60.000 – 75.000 €
6	>75.000 €

Thank You for your Cooperation and Patience: