The Economics of Agri-Environment Scheme Design: An Irish Case Study

Paula Cullen[12*], Mary Ryan[1], Cathal O'Donoghue[3], Paul Kilgariff[1]

[1] Rural Economy and Development Programme, Teagasc, Ireland
[2] Department of Economics, National University of Ireland, Galway
[3] NUIG Policy Lab, National University of Ireland, Galway

Contributed Paper prepared for presentation at the 92nd Annual Conference of the Agricultural Economics Society, University of Warwick, England

16-18 April 2018

*Corresponding author: Paula.Cullen@teagasc.ie

Abstract

Efficient agri-environment schemes (AESs), the European Union's main policy tool to improve the environmental performance of farms, address environmental concerns in a way which maximises the social benefit while minimising the cost. To design such schemes, policymakers are faced with a wide range of options. These include using voluntary or mandatory measures, top-down versus participatory approaches, collaborative versus coordinated participation, and whether to target the schemes or apply them horizontally. The efficiency of each of these options is dependent on the context and appropriateness of the application. Using Ireland as a case study, this paper assesses the evolving structure of AES design in the context of changing environmental targets, by creating an institutional framework to analyse past and current AESs and other measures. This is then compared to participation in AESs and the location of environmental public goods in order to determine the relative efficiency of policy.

Keywords agri-environment schemes, public goods

JEL codes Q18, Q51, Q57

Acknowledgements: The authors would like to acknowledge the support of the Teagasc Walsh Fellowship PhD Programme and the BEACON SFI project.

1. Introduction

Globally efforts have been made to increase or at least maintain the stock of environmental public goods. The Sustainable Development Goals and the Paris Agreement on climate change are just two of the numerous international agreements aimed at reducing the impact that humans have on the environment. Within the European Union (EU), a number of directives and regulations aiming to improve the environmental performance of Member States have been implemented with specific emphasis on biodiversity, water quality and climate stability. Accounting for nearly half of all land within the EU, agriculture has become a sector of central importance in many of these policies. The EU Common Agricultural Policy (CAP) which supports farmers throughout the EU, has moved from being directly concerned with the quantity of food production, to having numerous goals, not least the environmental sustainability of the agri-food sector. This was seen in the most recent reform of CAP in 2013 which included the sustainable management of natural resources as a priority focus. Since becoming compulsory for Member States in 1992, agri-environment schemes (AESs) have become the most important policy tool available to Member States to increase or maintain the stock of environmental public goods associated with agriculture.

While the design of AESs varies across Member States, schemes commonly involve farmers voluntarily participating and being compensated for the cost of undertaking management actions. The flexible nature of AES design has resulted in a wide range of schemes with numerous objectives that are sometimes conflicting. This raises a question as to whether they are being designed in the most efficient manner to achieve their goals. Literature has focused on the environmental improvements that can be attributed to AESs (Batáry, Dicks, Kleijn, & Sutherland, 2015; Finn & Ó hUallacháin, 2012; Jones et al., 2016; Kleijn et al., 2006). Little attention however, has been paid to the efficiency of design of AESs from an institutional economics perspective. A large number of options are available to Member States such as whether the schemes should be designed from the top-down or use a participatory approach, or whether they require coordination or cooperation between farmers. While each of these options have been looked at separately in terms of the actual impact in terms of participation and ensuing results (Lastra-Bravo et al. 2015; Wu & Babcock 1999; Gibbons et al. 2011; Newig & Koontz 2014 etc.), little work has been done to amalgamate the institutional design characteristics of AESs into one analysis. This paper aims to fill this gap.

Using Ireland as a case study, this paper outlines an institutional framework for the options available to policymakers in designing AESs by analysing current and past schemes. Ireland presents an important opportunity to assess these options as it has experience with different AES design approaches. Schemes in Ireland have ranged from being horizontal, top-down with payments made per hectare, to targeted participatory measures with payments for results. Ireland places a strong emphasis on the environmental sustainability of agriculture as evidenced by having the highest proportion in the EU of rural development expenditure on measures aimed at improving the environment and countryside during the last programme period (European Commission, 2013). In this paper we first identify the design options available to policymakers and their theoretical benefits and drawbacks in terms of efficiency. We then compare the characteristics of past and current policies to this institutional framework. This includes identifying the location of environmental public goods important to maintaining and increasing their stock and analysing scheme design in relation to efficiency. This allows us to identify the spatial concentration of environmental public goods and compare it to the past schemes and their participants.

The next section outlines the theory of public goods and the institutional characteristics of environmental policy. This is followed by a description of the methodology and the spatial and survey data used to conduct the analysis. The results are broken into three sections: an institutional analysis of past and current AESs, an analysis of the actual participants involved in schemes and finally an analysis of the concentration of environmental public goods. The paper ends with conclusions and policy recommendations.

2. Economic Theory

2.1. Public goods and agriculture

Agriculture and agricultural land provide a range of public goods. Public goods are defined in economic theory as displaying non-excludability (it is impossible to exclude individuals from consuming the good) and non-rivalry (the consumption of the good by one individual does not reduce the amount available to others). For this paper we focus on the environmental public goods identified by Cooper et al. (2009), who used a theoretical framework to identify the most important goods associated with agriculture within the EU. These goods are farmland biodiversity, water quality and availability, soil functionality, climate stability, air quality, agricultural landscape and resilience to fire and flooding. The goods are described in Table 1 along with a brief explanation of how agriculture impacts their quantity and quality. Many of these goods display both public and private good characteristics, largely due to factors of production, such as land, being privately owned. Some of these public goods are provided directly by agriculture while others such as climate stability and water quality could exist if the land was used in an alternative way. However, the provision of these public goods is linked, so if the land were put to an alternative use, the presence of one or more of the public goods could be lost (Cooper, Hart, & Baldock, 2009).

Table 1: Description of the public goods provided by agriculture[1]

Public Good	**Description**
Farmland Biodiversity	The species and habitats on agricultural land as well as the services they provide.
Water quality & availability	Agriculture uses water and impacts the quality of water through contamination with soil, effluent and inputs such as fertiliser, pesticides and herbicides. The loss of nutrients to water from agriculture is highly localised and dependent largely on the biophysical context and to a lesser degree on farm management practices (Roberts, Gonzalez-Jimenez, Doody, Jordan, & Daly, 2017).
Soil functionality	Soil is a resource for agriculture that is used to grow grass, trees and crops, break down wastes and provide nutrients. Agriculture can harm soil functionality if not managed appropriately. It is in farmers' interest to manage their soil well, hence soil has private good characteristics. However, society desires long term soil functionality for food production, biodiversity protection, water management and landscape (Cooper et al., 2009).
Climate stability	Climate stability is impacted positively through carbon storage and negatively by greenhouse gas emissions. Soils provide carbon storage. Peat soils contain the most organic carbon stocks. Unsustainable practices such as drainage, clearance and extraction result in the loss of this storage. Agriculture also emits greenhouse gases (GHG) including methane, carbon dioxide, nitrous oxide and ammonia. Due to the dominance of ruminant animals in the Irish agri-food sector, , agriculture accounted for 32% of the GHG emissions in 2016 (EPA, 2017a). Unlike water quality, the impact of GHGs on the environment is global, rather that localised.
Air quality	Air quality is non-excludable and non-rivalrous, a pure public good. It is impacted negatively through agricultural activities that emit greenhouse gases as well as burning and odours from livestock.
Agricultural landscapes	Agricultural landscapes, which have existed for many centuries, are increasingly being appreciated for their aesthetics. However, certain agricultural landscapes are valued more than others. Generally, more extensive landscapes are valued over intensive landscapes (Howley, Donoghue, & Hynes, 2012).
Resilience to fire	Sufficient grazed vegetation provides a barrier to fires reducing the risk of fires to crops, forests and houses.
Resilience to flooding	Agriculture can improve resilience to flooding through land management practices that improve water storage capacity.

The characteristics of public goods, including those associated with agriculture, lead to 'free-rider' behaviour. Due to non-excludability and non-rivalry there is no incentive for users to pay for them as they can get a 'free-ride'. The lack of paying consumers for public goods means that there is no incentive for private provision. If this results in an under-production of a good or over-consumption resulting in congestion, then it signifies market failure. In agriculture, the lack of a market for public goods means that farmers have few incentives to increase or maintain their provision, potentially leading to an undersupply. An undersupply of public goods associated with agriculture can be evidenced by the legislative targets and requirements set at an EU and national

[1] For a more in-depth description of these public goods and their relationship with agriculture in the European Union context see Cooper et al. (2009).

level as well as ground level issues identified by research associated with a lack of the good. Concern of the public for the state of the good also gives an indication of undersupply. This evidence is presented in Table 2.

Table 2: Evidence of an undersupply of environmental public goods associated with agriculture in Ireland[2]

Public Good	Evidence of undersupply
Farmland Biodiversity	Under the Habitats Directive Ireland must report on the status of 58 habitats and 61 species (non-bird) that are considered threatened in the EU. They must also report the status of 199 birds which are of concern to the European Union under the Birds Directive. Within Ireland 37 birds are on the Red List of conservation concern which means they have declined by over 70% or are threatened worldwide.
Water quality & availability	The Water Framework Directive (2000) commits Member States to achieve good qualitative and quantitative status of all water bodies. An Environmental Protection Agency (EPA) study in Ireland found that there was a 1% decline in the number of high or good ecological status river water bodies in 2010-2015 compared to 2007-2009. Agriculture is identified as a significant pressure in 'At Risk' water bodies. Nutrient losses from agriculture and domestic wastewater are considered to be the primary reasons why Water Framework Directive water quality objectives in Ireland will not be met (EPA, 2017b).
Soil functionality	Threats to soils in Ireland under current land use, management and climate conditions are low by international standards (EPA, 2016). However, this could be threatened by the increase in production that is required to meet Food Wise 2025[3] goals. Good soil management also reduces erosion which in turn improves water quality.
Climate stability	Global initiatives to improve climate stability through the reduction of greenhouse gases and increase in carbon storage are evidence of an undersupply. The Paris Agreement and the Kyoto Protocol share the same goal of restricting the rise in global temperature to below 2ºC. Ireland has agreed to reduce its emissions by 20% of 2005 levels by 2020.
Air quality	The EU has set limits for total emissions of sulphur dioxide, nitrogen oxides, ammonia and volatile organic compounds in the National Emission Ceilings Directive (2001).
Agricultural landscapes	Land abandonment resulting in a loss of agricultural landscapes which the public values, leading to an undersupply. The EU attempts to stem this loss through large scale income supports including the Basic Payment for all farmers and the Areas of Natural Constraint payment for farmers in marginal areas.
Resilience to fire	The climate of Ireland results in fire being of lower consideration than some parts of the EU, however, land abandonment or land-use changes would result in increased fire risks.
Resilience to flooding	Flooding is an increasing phenomenon in Ireland and climate change is expected to increase its prevalence and severity. Planning is ongoing to minimise the damage caused indicating a need for increased resilience which could be provided by changing agricultural practices.

Market failure resulting in an undersupply of public goods from agriculture indicates a need for public intervention. However, interventions only result in an efficient allocation if the benefits outweigh the costs. This requires an estimation of the value of benefits that could transpire. To calculate this, we must know the value that society places on the public goods as well as the

[2] For further evidence at a European wide level see Hart et al. (2011).
[3] Food harvest 2020 includes a goal to increase the value of primary production of the agriculture, fisheries and forestry sector by 65% by 2020.

importance of the good in terms of the provision of ecosystem services. This value must then be compared with the cost of intervening.

Due to the non-market nature of public goods, valuing benefits is difficult as there is no market price. There is a large literature using contingent valuation methodologies to place a value on public goods and also on how to combine valuations of different goods with the benefit transfer method (Brouwer et al. 1999; Loomis & White 1996; Noonan 2003; Richardson et al. 2015 etc.). These methodologies are not perfect (Diamond & Hausman, 1994), however, if their design is carefully considered and implemented acceptable values can be found (Carson, Flores, & Meade, 2001). Valuing every public good associated with agriculture in a country is a mammoth task especially due to the number of localised goods which would each have a separate value. These values would also change over time as land-use and people's opinions and knowledge change. This difficulty in obtaining an accurate estimate of the value to society provided by public goods associated with agriculture leads to difficulty in obtaining an efficient outcome as the level and cost of intervention needed is not clear.

2.2. Institutional Framework

The difficulty in valuing the benefits of increasing the supply of environmental public goods has led to interventions being largely based on achieving environmental targets rather than maximising benefits to society. Predominantly this has resulted in goals to at least maintain the current level of public goods or increase their supply. A number of policy options are available to policymakers to achieve these goals, each with their own benefits and drawbacks in terms of efficiency. While not an exhaustive list, a number of these opposing options are discussed below. The policy options represent the most significant in terms of the differences in the resulting schemes as well those that have actually been implemented within the EU and specifically within our case study country, Ireland. The first section outlines two principles within which all environmental policy falls, the polluter-pays-principle and the provider-gets principle. AESs are an example of the provider-gets principle, as farmers are paid for the costs they incur in making changes. AESs are also voluntary for farmers and the pros and cons of this are further discussed. The other sections in the framework relate to design options within schemes. These include top-down vs. participatory approaches, co-ordination vs. cooperation, horizontally schemes vs. targeted and finally we examine the way in which the payment rates are applied. Together these represent the major options available to policy makers in the current regulatory climate.

2.2.1. Polluter-pays vs. provider-gets

The first key decision to be made by policymakers is who will pay the cost, the public or the farmer? If it is the farmer, then the polluter-pays-principle (PPP) applies. In this case, the person who is likely to cause or has caused damage to an environmental good is the one who is responsible for preventing damage or compensating for it. It was first mentioned in a recommendation by the OECD in 1972 as a way to encourage the rational use of scarce resources by allocating the costs of pollution prevention and control measures (OECD, 1972). Since then it has become an important component of international environmental policy (Tobey & Smets, 1996). The PPP can be implemented using economic instruments or through regulation. This means that the cost falls on the polluter for any measures taken for maintenance or opportunity costs.

Although the principle originally was largely used in relation to the manufacturing industry and energy plants, it is now also implemented with the agricultural sector (Baldock, 1992). However, in agriculture environmental degradation arises typically from non-point sources, meaning it is difficult to determine the source, and may be spread over an extensive area, meaning it is difficult to monitor and to enforce penalties (Baldock, 1992). Economic and political considerations also come into play as farms are often small and family-run and do not have the ability to absorb extra costs in the same way as a large manufacturing firm. Politically and socially, agriculture is seen as an important industry to maintain, as indicated by the large scale subsidisation, leading to a reluctance in imposing costs which might adversely affect farmers' income and production (Tobey & Smets, 1996).

The need for an alternative principle, especially for agriculture has been argued since the conception of the PPP (Baldock, 1992; Hanley, Kirkpatrick, & Oglet, 1998; Tobey & Smets, 1996). Largely this centres on the concept of a reference level of environmental quality. The PPP puts the cost of maintaining the public good at this reference level onto the farmer. However, this means there is no incentive to increase the supply above this level, even if the farmer is able to do so. Also, farmers are not well placed to absorb this extra cost.

An alternate principle to the PPP is the provider-gets principle (Blöchliger, 1994). This principle suggests that those increasing the supply of environmental public goods should be compensated for doing so. Mauerhofer et al. (2013) state that there are three reasons for using this method in place of PPP: where there are no regulations in place to indicate the duty of the polluter, where the regulations are not sufficient; and where they have the benefit of being able to provide incentives for the best environmental practice.

2.2.2. Voluntary vs. mandatory

Under a mandatory policy a farmers' decision to undertake a measure is based on the cost of adoption, compared to the likelihood of receiving a penalty and the magnitude of the penalty if found to be non-compliant Thus the cost of mandatory measures falls on the farmer and hence is a PPP option. However, monitoring, enforcement and other transaction costs involved in the implementation of such measures may be high, especially given the non-point source nature of agricultural pollution. This has led to the increasing popularity of voluntary measures where contributions are made to farmers towards the costs of adoption.

Under a voluntary policy farmers weigh up the cost of adopting a measure against the payments received for doing so. Wu and Babcock (1999) find that voluntary measures are more efficient than mandatory measures if and only if the deadweight losses of government expenditures under the voluntary program are less than the difference between the private and public costs of government services plus the additional implementation cost of the mandatory program. This is likely if the deadweight loss of raising government revenue is small, the number of farms is large, and the implementation costs of the voluntary program are much less.

Voluntary environmental policy can be implemented in two ways: through inducing participation with the threat of a harsher outcome without participation, or through incentives. The first could be considered not truly voluntary. In relation to incentives, in practice these revolves largely around payments for environmental or ecosystem services (PES), a form of the provider-gets mechanism. Engel et al. (2008) point out three situations where inefficiencies can occur with these types of policies: payments offered are insufficient to induce a socially desirable level of adoption, the level of adoption is adequate but the cost is higher than the value of the services, or there are payments for adopting practices that would have been adopted anyway. The first two problems result in a social inefficiency, i.e. the marginal social cost is not equal to the marginal social benefit, leading to a reduction in social welfare. The third problem, known as lack of additionality, leads to a socially efficient outcome however it is not financially efficient as the socially efficient outcome would have been reached without expense. This is difficult to measure as we do not know what would have occurred without the scheme. The presence of additionality means that the scheme has had a positive environmental impact.

2.2.3. Top-down vs. participatory approach

Top-down approaches to environmental policy occur where one actor, generally the government, implements advisory, regulatory or economic policies. The alternative is participatory approaches where multiple actors (including those who are impacted by the decision) are involved. Participatory approaches can differ in the level of representation of interest groups, the amount of information that flows up or down and the influence that participants can have (Newig & Koontz, 2014).

Top-down approaches are limited by the lack of information and involvement of those who live and work in the areas where the environmental policy will be implemented (Van Den Hove, 2000). Including multiple actors allows for a pooling of information as well as integration of new information, as it becomes available throughout the implementation process (Van Den Hove, 2000). Participatory approaches are linked to the promotion of inclusivity in the planning and decision-making processes, with the objective of which increasing the likelihood of acceptance (Kapoor, 2001; Newig & Kvarda, 2012). Through these methods participatory approaches aim to improve effectiveness over top-down approaches. A key benefit of participatory approaches is that they enhance iterative programming where feedback loops result in in-situ improvements policies, allowing for more flexibility than top-down approaches (Kapoor, 2001).

The inclusion of multiple actors (each with their own interests), may result in conflicts over the nature of the problem and the potential solutions (Van Den Hove, 2000). This also may result in lower standards of improvement if the actors are more concerned with economic interests than environmental ones (Newig & Kvarda, 2012). The access to new information and knowledge is also not important if the issue requires more scientific and expert knowledge than a layperson. The inclusion of many actors may also result in the dilution of important information pertinent to solving the issue (Rydin, 2007).

2.2.4. Co-ordination vs. collaborative

Environmental policy is often targeted at multiple single actors making changes. Collaborative environmental actions involve a group of actors working together. Collaborative action between farmers is seen as generally beneficial in improving agri-environmental management, however there are limitations as identified by Prager (2015). Benefits can be identified in three areas: environmental, economic and social. The environmental benefits come from the larger scale management of landscape which reduces the likelihood of habitat fragmentation and maintains ecological networks, improving the performance of the management actions in increasing/improving biodiversity. This impact is due to the threshold effect where some ecosystem services operate at a larger scale than can be improved through the actions on just one farm in a local area (Dupraz, Latouche, & Turpin, 2009). Only some public goods are influenced by the localised threshold effect. Global goods such as climate stability are not impacted, however, local public goods such as water quality, collaborative participation is necessary to make actual improvements in the good.

2.2.5. Horizontal vs. targeted

Horizontal measures are open to farmers across the country or region in which the policy is in place. They generally cover a wide area and require farmers to make relatively small changes in practices (Matzdorf, Kaiser, & Rohner, 2008). Targeted measures are limited to certain zones and are usually implemented to manage specific species or ecosystems, requiring more substantial changes from farmers in practice. Theoretically, targeted measures will be more cost effective as they are only implemented in areas of need, resulting in the greatest benefit. This also reduces the risk of a lack of additionality, where little to no changes are made. This type of scheme will also be more likely to result in changes by reaching the threshold level above which improvements in the good will occur (Dupraz et al., 2009). However, identifying the farms to target may be difficult as this would need research, increasing the cost of implementation. Van der Horst (2007) also highlighted that public goods are not spatially compatible with each other, which may require separate targeting for different goods. Targeted measures would also not be effective where there is uncertainty and large time and space scales associated with the environmental issue, resulting in difficulty identifying those responsible for the public goods (Van Den Hove, 2000).

2.2.6. Payment: Action-based vs. results-based vs. hybrid

Payments made for conducting voluntary environmental measures on farms can be mostly divided into two groups: action-based and results-based. Action-based payments are made on the basis of undertaking farm management actions that are intended to increase the supply of environmental public goods. The payments are generally prescribed amounts for each measure applied horizontally. Heterogeneity among farms mean costs are lower for some which will result in some farmers being over-rewarded for participating and hence by more inclined to participate. Depending on the reason for the heterogeneity in costs faced, it may indicate that some farmers do not need to make many changes and presents another case of the lack of additionality. While socially efficient, this is not financially efficient as the money could have been employed to make greater changes in the stock of environmental public goods elsewhere. Derissen and Quaas (2013) find that this payment system is only optimal if there is an information asymmetry.

Alternatively, results-based payments are based on the improvement in the environmental public good, according to a baseline level set prior to the implementation of the measure. On the other hand, results-based schemes are financially efficient as payments are not made if there is no improvement. The difficulty in implementing these schemes is that a baseline level needs to be measured. This requires on-the-ground analysis of the current state of the land, which incurs added expense. There also needs to be an evaluation system in place to determine the level of payment based on the level of improvement in the environmental public good. Results-based schemes also allow farmers to undertake management actions that fit their context, and which will achieve the best results in the most cost-efficient manner (Gibbons et al., 2011). Results-based schemes do not suffer from a lack of additionality as farmers must prove improvements to obtain payment.

However, results-based payments suffer from issues surrounding environmental uncertainty, where even if a farmer undertakes perfect measures to improve the environmental public good, uncontrolled natural events can negate the attempts to improve the public good, resulting in low or no payments to the farmer. This indicates a transfer of risk to the farmers as it is they who lose if there are negative environmental consequences from an unexpected event such as flooding or a storm (Derissen & Quaas, 2013; Schroeder, Isselstein, Chaplin, & Peel, 2013). This may result in non-participation by risk-averse farmers resulting in lower overall participation and lower environmental improvement. Hybrid payments, which comprise a mix of payments for action and payments for results, are suggested as a solution to this problem. These reduce risk to farmers while still providing the incentives for direct environmental improvement as provided for by results-based payments. Derissen and Quaas (2013) find that hybrid payments are optimal for every situation other than when there is no symmetrical information.

In summary, there are numerous options available to policymakers in designing environmental policy for agriculture. Each option has its own benefits and flaws. The next section describes the methodology and data used to analyse the schemes that have been implemented in Ireland using the institutional framework outlined in this section.

3. Data and methodology

To assess the past and current environmental policy relating to agriculture in Ireland, we use a multifaceted analysis. First, we chart the progression of voluntary environmental schemes over time and compare characteristics to the institutional framework outlined in the previous section. We then investigate the potential impact that AESs could have relative to the spatial environmental public good concerns of Ireland identified at a townland level.

To conduct an institutional analysis of AESs in Ireland we first describe past and current schemes and measures. For the purpose of this study we limit the analysis to those that have a primary goal of improving the environmental performance of farms, are voluntary and subscribe to the provider-gets principle. This excludes schemes that have secondary environmental objectives or result in environmental improvements as a by-product of their primary objective (such as those which predominantly are aimed at providing income support for farmers). These schemes are examined against the institutional framework outlined in the previous section. Further analysis of their possible impact on the stock of environmental public goods is conducted through looking at the public goods in which they aim to improve as well as analysis of the scale and spatial aspects of participation.

To analyse the types of farms involved in AESs in Ireland we utilise the Teagasc National Farm Survey (NFS) database for the years from 1996 to 2016. The NFS provides yearly information on approximately 1000 farms in Ireland. It contains information on farm and farmer characteristics as well as their participation in the various AESs that have been in place in Ireland both past and present. This is combined with information from the Teagasc Agri-Environment Costs Survey conducted in 2012. This survey allows us to conduct a spatial analysis at county level of participants and non-participants in the schemes prior to Green-Low Carbon Agri-Environment Scheme (GLAS), the current scheme which is not included in this particular analysis due to the spatial targeting involved in its implementation.

This spatial analysis is then compared with the location of townlands where the supply of environmental public goods is vulnerable. Particular emphasis is placed on farmland biodiversity, water quality and agricultural landscapes, which are all localised public goods. Determining a spatial relationship of environmental public goods in Ireland requires combining the different goods into one map. This requires a valuation of each of the goods in relation to the others; however, the valuation of public goods is complex and poses challenges. To simplify this complexity, we assume that each of the public goods is equivalent in terms of value. In practice, this means that combining the public goods into one map involves giving each of the public goods of concern a value of one if a particular townland has been identified as the location of a public good that should be conserved or improved. The sum of these public good concerns thus provides a crude measure of the concentration of environmental public good concerns in a particular area.

The mapping resources to identify areas of importance for farmland biodiversity have been obtained largely from the National Parks and Wildlife Service (NPWS). Information on the distribution of birds, animal species and habitats that are of conservation concern within the European Union are reported as is required by the Habitats and Birds Directives. These directives also require the creation of Special Areas of Conservation (SAC), areas important for particular habitats or species protection, and Special Protection Areas (SPA), areas important for particular bird species, which together form the Natura 2000 network. Also identified are Natural Heritage Areas, which are important for the protection of certain habitats and species. Townlands in the top quartile for the number of birds, species and habitats reported under the Habitats and Birds directives or the presence of a SAC, SPA or NHA in a townland has been taken as identifying that townland as important for increasing or maintaining farmland biodiversity.

The identification of townlands important to water quality is found through using both Q-values (a measure of ecological river water quality), and Nitrogen and Phosphorus susceptibility maps, both of which are provided by the Environment Protection Agency (EPA). Q-values range between 1 and 5 where 1 indicates poor ecological quality, while 5 is the reference value, indicating pristine or high ecological water status. For this study we have taken townlands with rivers with a Q-value of 1, 2 and 5 to be of importance as those with values 1 and 2 need to improve under the Water Framework Directive, while maintaining pristine rivers is also important. Nitrogen and Phosphorus susceptibility maps measure the likelihood of nutrient losses to water in five categories from very low to very high. Townlands with a susceptibility of very high for either of these measures are identified as important to water quality for the purposes of spatially adding townlands of concern for water quality to areas of biodiversity concern.

We also include a measure of agricultural landscape environmental concern. While both biodiversity and water quality have an impact on this public good, cultural factors also have an influence. Therefore we have included the location of National Monuments as reported by the Heritage Council and commonage as identified by the NPWS. The National Monuments include areas of archaeological and historical importance which are protected. A townland receives a value of one for this good if it contains either of these.

As explained above, the concentration of environmental concerns (at townland level) relating to each of the public goods described is presented in Table 3 along with the number of townlands that meet the level out of a total of 50,109 townlands.

Table 3: Summary Statistics for Measures of Environmental Concern by townland

Public good	Measure	Level	Number of townlands
Farmland Biodiversity	Special Area of Conservation (SAC)	Townland contains SAC.	11,663
	Special Protection Areas (SPA)	Townland contains SPA.	5,088
	Natural Heritage Areas (NHA)	Townland contains NHA	150
	Number of habitats	Townland is in the top quartile for the number of different habitat types reported under Article 17.	10,252
	Number of species	Townland is in the top quartile for the number of non-bird species reported under Article 17.	11,494
	Number of bird species	Townland is in the top quartile for the number of bird species reported under Article 12.	11,610
Water Quality	Water quality (q-value)	Townland has river with q-value of 5, 2 or 1 representing high, poor or bad quality respectively.	9,685
	Nitrogen susceptibility	Townland has very high nitrogen susceptibility.	2,425
	Phosphorus susceptibility	Townland has very high phosphorus susceptibility.	8,826
Agricultural landscape	Commonage	Townland contains commonage land.	1,256
	National monument	Townland contains a national monument.	25,637

4. Results

4.1. Institutional analysis of past and current schemes

Agri-environment schemes are voluntary, provider-gets principle based, economic measures aimed at improving the environmental performance of farms. EU Member States have been required to implement agri-environment schemes since 1992 following Council Regulation EEC no. 2078/92. Objectives and design differ between Member States. Ireland has implemented three large-scale schemes: Rural Environment Protection Scheme (REPS), Agri-environment Options Scheme (AEOS) and the current scheme the Green Low-Carbon Agri-Environment Scheme (GLAS). Numerous other environmental measures have also been implemented over the years. All AESs, other schemes and measures are listed chronologically in Table 4 along with their institutional characteristics.

Table 4: Characteristics of voluntary schemes/measures in Ireland

Scheme	Year introduced	Top-down vs. participatory	Co-ordination vs. collaborative	Horizontal vs. targeted	Payment: Action-based vs. results-based vs. hybrid
Western Package Scheme	1981	Top-down	Co-ordination	Horizontal	Action-based
Afforestation Grant and Premium Schemes	1989	Top-down	Co-ordination	Horizontal	Action-based
REPS I-IV	1994	Top-down	Co-ordination	Horizontal	Action-based
Farm waste management scheme	2006	Top-down	Co-ordination	Horizontal	Action-based
Organic Farming Scheme	2007	Top-down	Co-ordination	Horizontal	Action-based
Forest Environment Protection Scheme	2007	Top-down	Co-ordination	Horizontal	Action-based
EU Life+ Programme	2007	Participatory	Collaborative	Targeted	Hybrid
AEOS I-III	2010	Top-down	Co-ordination	Horizontal	Action-based
GLAS	2015	Top-down	Co-ordination	Targeted/horizontal	Action-based
Low Emission Slurry Spreading Equipment Scheme	2015	Top-down	Co-ordination	Targeted	Action-based
Animal Welfare, Safety and Nutrient Storage Scheme	2015	Top-down	Co-ordination	Horizontal	Action-based
Beef Data and Genomics Programme	2015	Top-down	Co-ordination	Horizontal	Action-based
EIP-Agri Projects	2017	Participatory	Collaborative	Targeted	Hybrid

Tillage Capital Investment Scheme	2017	Top-down	Co-ordination	Horizontal	Action-based

The REPS and AEOS AESs were similar in that they were both top-down, horizontal schemes that involved co-ordinated actions undertaken by farmers who volunteered to participate. Farmers signed five-year contracts agreeing to undertake certain actions as well as follow a nitrates management plan, with threat of penalty for non-compliance. REPS involved the whole farm with payments made at a per hectare rate while the consequent schemes only involved completing actions and payments per action. GLAS was the first large scale top-down scheme that attempted to target the measures towards specific farms, based on areas of environmental concern as farmers with 'Priority Environmental Assets and Actions' including Natura 2000 sites, specific bird species, commonage, high status water areas and rare breeds had priority access to the scheme. GLAS also limited the payment available to farms to €5,000. All three of the AES schemes required farmers to follow Nutrient Management Plans in order to receive payment. Starting in REPS 4, and continuing into AEOS and GLAS, farmers with Natura 2000 designated land received a payment per hectare within the schemes for following a sustainable management plan.

The most recent progression in the design of agri-environment schemes is the European Innovation Partnership (EIP-Agri), where open calls for farmer-led, participatory project applications were sought in thematic areas including the preservation of agricultural landscapes, water quality, biodiversity and climate mitigation. Two of the projects implemented under this programme are the Hen Harrier and Freshwater Pearl Mussel scheme which are targeted at specific species with core target areas identified as important habitats where participants would be sourced from. The aim of the projects is to develop locally tailored solutions to problems with strong collaboration between a wide range of stakeholders.

There are a number of agricultural projects under The EU Life+ Programme, which began in 2007. The most significant and well-known is the Burren Life Programme which commenced in 2010. It aimed to increase the supply of a multitude of public goods including farmland biodiversity, water quality and agricultural landscape. Payments for the scheme were hybrid in nature, with some payments based on actions undertaken, while others were based on improvements in the quality of habitats and water. A key component of the programme was that it was 'locally-led', employing a collaborative approach between farmers and other stakeholders. The programme has been very successful and has paved the way for more collaborative schemes that base payments on results. One important factor in its perceived success was the proportionately similar implementation and administrative costs in comparison to the large-scale AESs which showed that targeted results-based schemes could be implemented without the costs outweighing the benefits (Cullen et al. Forthcoming).

A number of other measures have been introduced with specific priorities. The Beef Data and Genomics Programme that was implemented in 2015 included a requirement for applicants to complete a Carbon Navigator[4] with the aim of reducing greenhouse gas emissions on participating farms. Payments to aid farmers to convert to and maintain organic farming have been made since 1994. Between 1994 and 2006 these were made within the REPS scheme. In 2007 the Organic Farming Scheme was created as a separate scheme.

The Targeted Agricultural Modernisation Scheme (TAMS) is a grouping of capital grant schemes designed to incentivise private investment in physical farming assets in order to improve the economic and environmental performance of farms. In an early iteration of TAMS, the Farm Waste Management Scheme allowed farmers to improve their ability to meet the requirements of the Nitrates Directive by investing in assets to manage slurry and other farm waste. The current TAMS which was implemented in 2015 includes the Animal Welfare and Nutrient Storage Scheme and the Tillage Capital Investment Scheme which have primary goals of reducing nutrient loss to waterways while the Low Emissions Slurry Spreading Scheme aims to reduce greenhouse gas emissions from agriculture. Another TAMS scheme is the Organic Capital Investment Scheme which has a primary objective of reducing risk to converting or registered organic farmers, however secondary objectives include reducing nutrient loss and emissions.

The earliest schemes identified in Table 4 are all aimed at increasing forest cover on private agricultural land in Ireland. The Western Package Scheme, was the first EU funded afforestation scheme and was available only in western counties, had a slow uptake. This was replaced in 1989 with the Forest Premium Scheme, and various iterations of the current Afforestation Grant & Premium Scheme which was opened in 1992. This provided grants to plant land and maintain it in the first few years as well as payments to compensate for the agricultural opportunity cost of planting. Over time these schemes have been added to and now include the Forest Roads Scheme, aiming to improve access to the forests, the Woodland Improvement Scheme and the Native Woodland Scheme among others. These schemes are all top-down co-ordinated schemes that are applied horizontally.

The scheme characteristics found in Table 4 indicate that the dominant form of AES implemented in Ireland to date is top-down, co-ordinated, horizontal, action-based schemes. While these are still currently in place, largely in the form of schemes aimed at providing capital to improve the environmental performance of farms, there is also an increased use of targeted schemes aimed at specific areas, species or habitat types. There is also an increase in the use of collaborative approaches, funded under the Life+ Programme and EIP-Agri, indicating the evolving nature of AES scheme design in Ireland.

While the policies mentioned differ in their institutional characteristics, they all are aimed at improving the stock of environmental public goods on agricultural land. Table 5 indicates the public goods targeted for improvement by the different environmental policies. The majority of policies are targeted at the improvement or maintenance of public goods such as farmland biodiversity, water quality and availability, soil functionality and agricultural landscapes. The horizontal schemes all target numerous public goods, evidence of their multiple objectives.

---

[4] The Carbon Navigator is an online farm management package that quantifies the environmental gains that can be made on individual farms by setting targets in key areas such as grassland management. It allows farmers to see the reduction in GHG emissions from making changes such as lengthening the grazing season.

While climate stability was a primary goal of AEOS and the current large-scale scheme GLAS, however, the only optional measure available for farmers is the introduction of improved slurry spreading methods. Largely the capital grants under TAMS have concentrated on maintaining or increasing the stock of one public good each. These schemes generally provide the capital to either help farms achieve cross compliance standards in order to receive subsidies (or not incur penalties) or assist in entering horizontal schemes such as GLAS or the Organic Farming Scheme.

Table 5: Public goods and agri-environment scheme primary goals

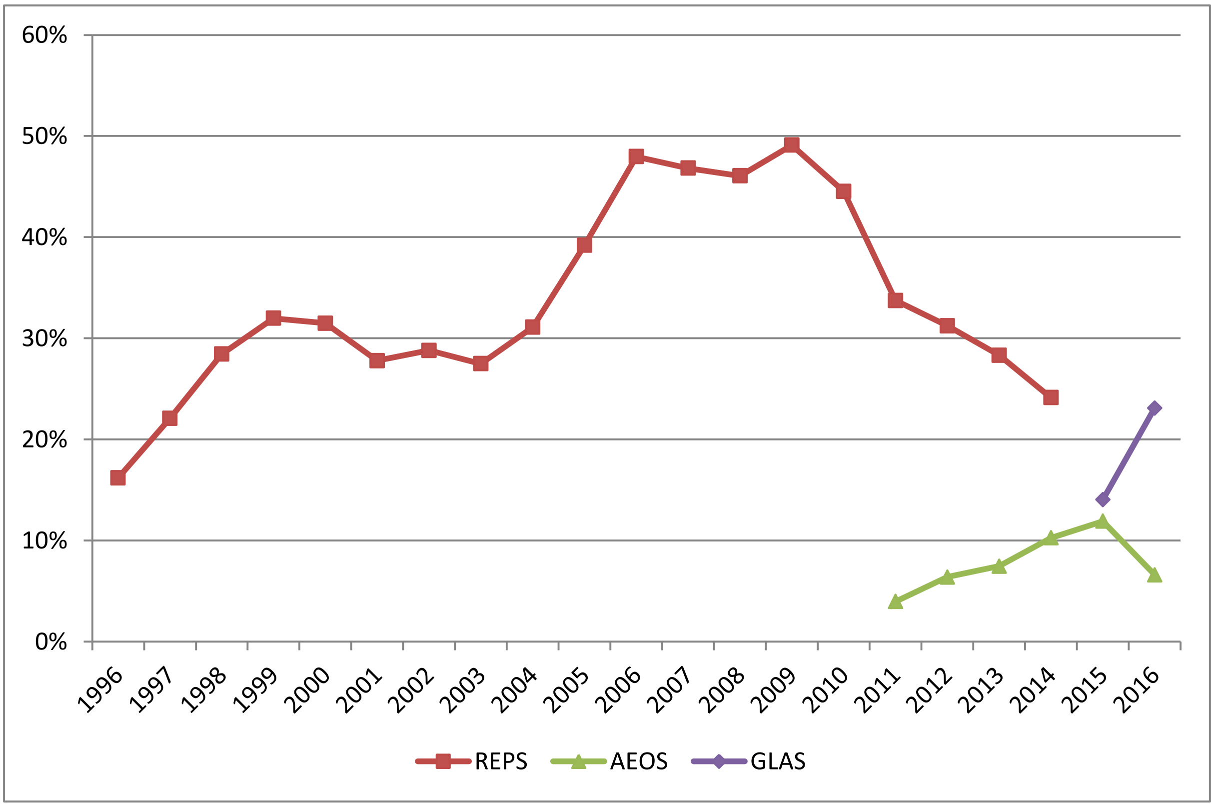
Type	Scheme	Farmland biodiversity	Water quality and availability	Soil functionality	Climate stability	Air quality	Agricultural landscapes	Resilience to fire	Resilience to flooding
Horizontal AES	REPS I-IV	✔	✔	✔			✔		
	AEOS I-III	✔	✔	✔	✔		✔		
	GLAS	✔	✔	✔	✔		✔		
	Organic Farming Scheme	✔	✔	✔			✔		
	Afforestation grant and premium scheme	✔	✔		✔	✔			
Targeted AES	Life+ Programme	✔	✔	✔			✔		
	EIP-Agri	✔	✔	✔			✔		
Capital scheme	Farm waste management scheme		✔						
	Low Emission Slurry Spreading Equipment Scheme				✔				
	Animal Welfare, Safety and Nutrient Storage Scheme		✔						
	Tillage Capital Investment Scheme		✔						
Other	Beef Data and Genomics Programme				✔				

In relation to targeted schemes, the EIP-agri programme is comprised of a number of localised targeted schemes. These individual schemes such as the Hen Harrier and Pearl Mussel schemes are aimed at improving only one public good, in their case farmland biodiversity. In contrast, schemes under the Life+ programme such as the Burren Life Programme target multiple public goods in a holistic way as they are aimed at general improvement of the environmental performance of farms and increasing the amenity value of a specific area.

4.2. Agri-environment scheme participant analysis

The three large scale action-based AESs in Ireland, REPS, AEOS and GLAS, share a number of characteristics as identified in the previous section. Participation in these schemes differed significantly as illustrated in Figure 1 which charts scheme participation from 1996 to 2016 based on the National Farm Survey, a representative sample of Irish farms. REPS had by far the highest participation rate, reaching almost 50% in 2009, following which no new contracts were issued. This scheme was a whole farm scheme and the payments available to each farm were higher than the schemes that followed. AEOS had relatively low participation levels, however, there was significant overlap between the two schemes with the more lucrative REPS contracts on-going for most of the AEOS period. GLAS began in 2015, and in 2016 over 20% of the NFS farms were involved, still well below the REPS level. GLAS is split into three tranches with new entrants yearly. With only two data points available little can be said on the participation trend.

Figure 1: Percentage of farms in the National Farm Survey involved in an agri-environment scheme (1996-2016)



Extensive farms are likely to require fewer practice changes to participate in an agri-environment scheme in Ireland. This lower level of change is also generally associated with a lower opportunity cost of participation, which is lower for extensive than for intensive farms. As horizontal schemes pay the same amount to all farms for specific measures, theoretically this will result in higher participation of extensive farms in schemes as they will likely be overpaid relative to intensive farms. This has been shown in past research (Hynes & Garvey, 2009; Murphy, Hynes, Murphy, & O'Donoghue, 2014). This indicates a possible financial inefficiency in the schemes as the costs to the public result in fewer changes than if intensive farms joined.

Figure 2

Figure 2 shows the percentage of extensive and intensive farms that participated in REPS, AEOS and GLAS between 1996 and 2016. Extensive farms are those with stocking rates below 1.4 livestock units per hectare as this was the level below which farmers could obtain an 'extensification' payment. In order to reduce the mapping complexity in this analysis, stocking rates above 1.4 livestock units per hectare are designated as intensive. Prior to 2010 a higher percentage of extensive farms participated in REPS than intensive, however post 2010 this relationship changed. This could be due to the wind-down of REPS indicating that contracts were ending for the extensive farmers who had joined the scheme earlier.

Figure 2

Figure 2 also shows higher extensive farm participation rate for those involved in AEOS. In this case the participation rate for extensive farms was at least twice as high as intensive farms for all years of the scheme. A similar result is seen for GLAS farms.

Figure 2: Scheme participation of extensive and intensive farms in Ireland

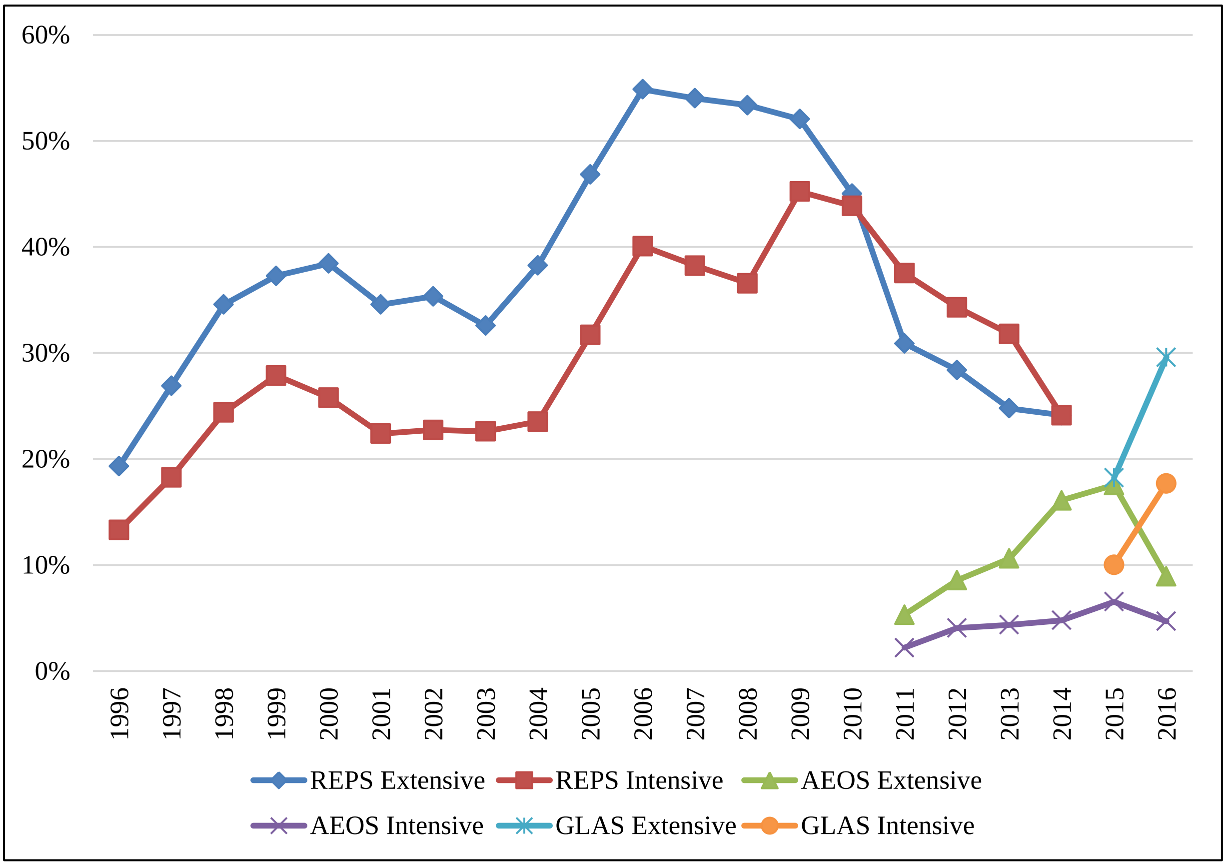
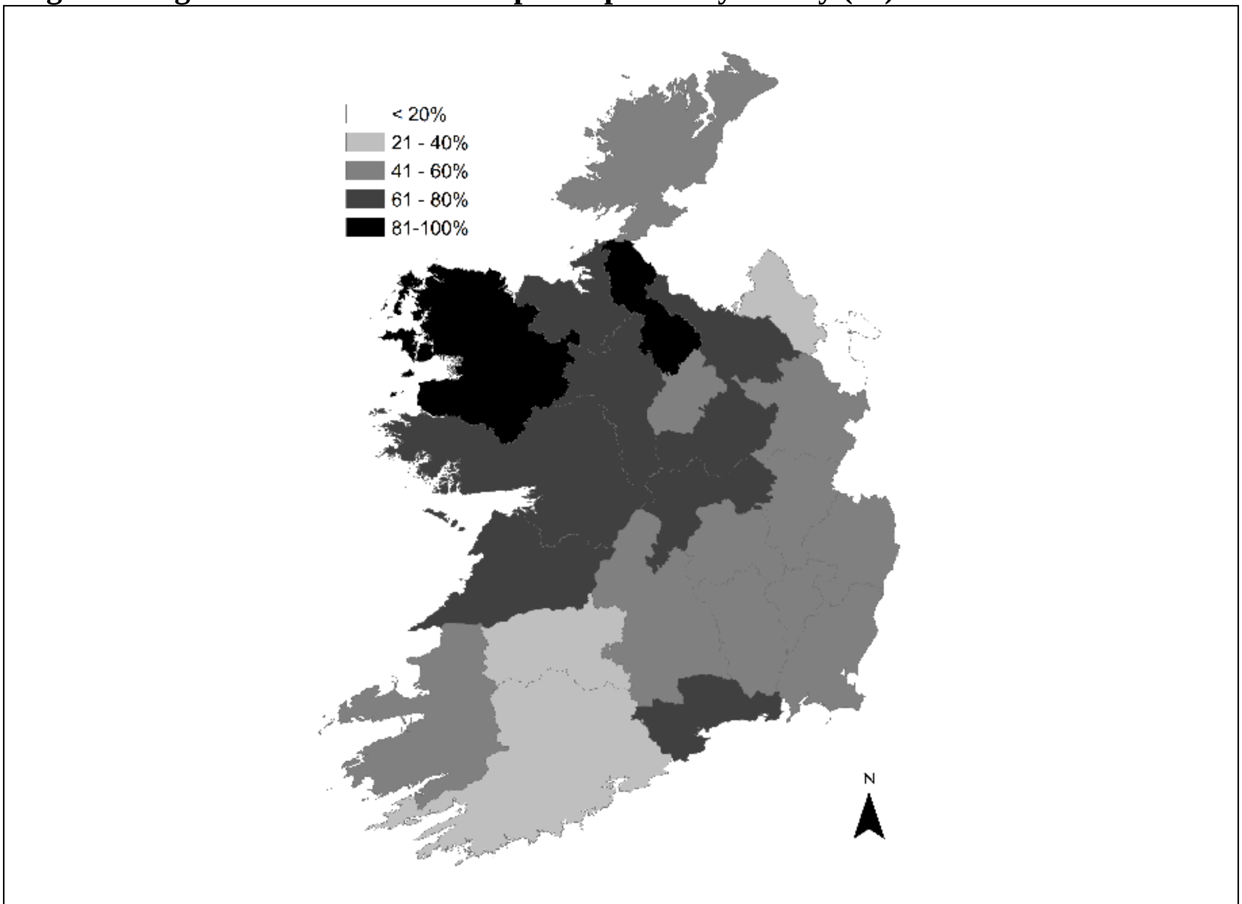


Figure 3 shows the percentage of farms in each county that are current or past participants in AESs as of 2011, prior to the implementation of the more spatially targeted GLAS scheme, using data from the Teagasc Agri-Environment Cost Survey. A pattern is evident with lower participation rates in the south-west. Higher participation rates are found in the north-west of the country with Leitrim and Mayo having the highest level of participation at 86% and 84% respectively within the sample. When compared to the average stocking rate throughout the country (shown in Figure 5), we again see the relationship between extensive farming and AES participation. Areas in the north-west which have lower average stocking rates have higher rates of AES participation, and the opposite is true for the south and east of the country. This raises potential scheme inefficiencies as scheme design may not have adequately addressed the question of additionality. Due to the flat rate nature of payments for the schemes, it is likely that participation is skewed towards farms with lower opportunity costs. The unintended consequence of this however is that the achievement of environmental improvement may be lower than if intensive farms participated and made more significant changes.

Figure 3: Agri-environment scheme participation by county (%)



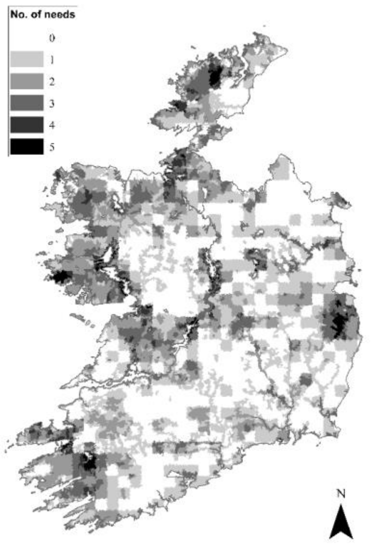
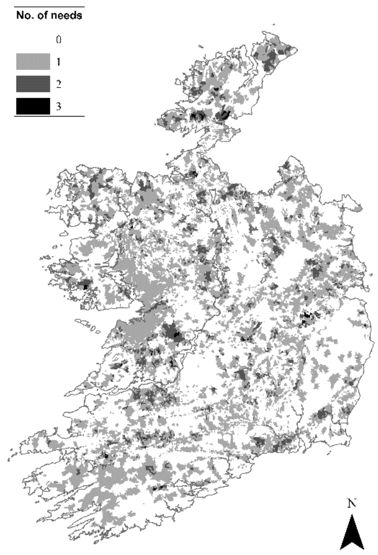
4.3. Spatial analysis of environmental public goods

The locations of concentrations of environmental public goods of importance to sustainability within Ireland are identified in Figure 4. Specifically, we have identified areas of importance associated with biodiversity, water quality, and agricultural landscape. If a townland meets any of levels indicated in Table 3 for each public good it takes a value of one for that good. For example, if a townland contains special protection area, then it receives a value of one for farmland biodiversity, leading to a range of zero to three for the three different types of public goods. The image shows us that largely these concerns are spatially discrete. While there are certain areas that have a high concentration of locations of environmental concern, such as the west of the country, there are also multiple townlands with a high concentration of different environmental public goods of importance adjacent to next to townlands with none. Groupings of townlands with high concentrations of concerns can be seen in the west of the country as well as in the south-east.

Figure 4: Concentrations of environmental public goods of importance in townlands in Ireland by type (farmland biodiversity, water quality & agricultural landscape)

The lack of a clear pattern in Figure 4 can largely be explained by the variation in spatial density of the different environmental public goods. Figure 5 breaks down these concerns into their separate categories. Each townland is categorised based on the concentration of different environmental public goods of importance as identified by each of the different measures in Table 3. Figure 5A shows the concentration of farmland biodiversity concerns at townland level. A significant portion of the country has no townlands with areas of concern for farmland biodiversity based on the measures used. These areas are largely located in the south and north-east of the country. There is a large concentration of townlands of importance for farmland biodiversity in coastal areas, specifically in the north-west, and south-west. By comparison areas with water quality concerns (Figure 5B) are more evenly distributed throughout the country, although once again there is a concentration in the west. The townlands with agricultural landscape environmental concerns i.e. commonage and national monuments are presented in Figure 5C. There is little pattern to areas with only one concern, (largely townlands having a national monument), when commonages are added, there are more areas of concern in the west. Together these images show that the different types of environmental concerns in Ireland are not spatially consistent and have diverse patterns.

Figure 5: Environmental public goods of importance in Ireland by type

Figure 5A: Concentration of farmland biodiversity of importance	Figure 5B: Concentration of water quality concerns	Figure 5C: Concentration of cultural agricultural landscape public goods
		

Comparing the concentrations of environmental public goods of importance (Figure 4) to participation rates of different counties in agri-environment schemes (Figure 3), we can see that high participation rates are not always in locations with a large number of important environmental features. While the highest participation rates in the north-west correspond to high concentrations of environmental concerns, this trend does not hold for areas with 61-80% participation rates as many of these townlands contain only one public good of environmental concern. Similarly, in the south of the country, where AES participation is low, there is a mix of high and low concentrations of environmental public goods of importance.

5. Conclusions

Agri-environment schemes and other environmental measures in Ireland have been evolving over time from top-down horizontal co-ordinated actions-based schemes to more targeted approaches (with increasing use of participatory approaches) to scheme design and collaborative implementation. The large-scale schemes (REPS, AEOS and GLAS) tended to be taken up by extensive farmers relative to non-extensive farmers as has been found in previous studies (Hynes & Garvey, 2009; Murphy et al., 2014). This is likely to have been due to lower opportunity costs of participation for extensive farmers. This suggests there may have been issues with a lack of additionality, where farmers did not have to make many changes to join the scheme and hence the environmental improvements attained were lower than if intensive farmers had joined in greater numbers. The success of the Burren Life

Programme and the acceptable levels of administrative and implementation costs suggests that targeting schemes to their spatial needs is possible and can be achieved efficiently (Cullen et al., 2018).

Spatially, our analysis shows that areas important to the improvement or conservation of the stock of environmental public goods, are not in discrete locations and are spread throughout the country. While there is a concentration of important areas in the west of the country, there are also numerous townlands with a high concentration next to townlands with low concentrations, suggesting that the environmental public goods analysed are localised. This suggests that optimally the targeting of schemes for these goods should be done at a small scale. On the other hand, climate stability is a global good, and hence changes made at any level will have an impact. There are also a number of other public goods related to agriculture which have not been discussed in this paper such as social public goods including rural vitality that are of concern to policy makers, however these are currently beyond the scope of this analysis. Another key implication of the spatial analysis of environmental public goods is the spatial inconsistency between the goods, each displaying different patterns in their occurrence. This suggests that although schemes generally have a large number of goals relating to different environmental public goods, it may be more efficient in terms of targeting to separate them.

The spatial analysis was limited to the data available on the public goods. Future work will expand the number of public goods analysed preferably to include measures for all of the public goods mentioned. This will allow for a broader understanding of the locality of environmental public goods that are important for maintaining and improving their stock. Preferably this would then be compared to actual participation in all schemes and measures mentioned, however, limited participation data is available. We also have not weighted the public goods in terms of their value to the public, instead assuming they have the same value. This is unlikely and the development of an index with different weightings for each good may increase the value of the analysis through further indication of the concentration of value of environmental public goods in certain areas.

Designing new AESs is challenging. There is no single ‘best’ option available to policymakers in terms of scheme efficiency, as each options has benefits and flaws. Ideally, policies should address the spatial disparity in the concentration of specific environmental goods. However, if this is too costly, then horizontal schemes may be more efficient. Efficient environmental policy requires identifying the correct mix of policies in order to address the specific problems faced. Solving localised problems may require targeted collaborative schemes, while addressing larger scale issues such as climate change, will require co-ordination of large numbers of farmers, either through regulation or economic means.

References

Baldock, D. (1992). The Polluters Pays Principle and its Relevance to Agricultural Policy in European Countries. *Sociologia Ruralis*, *32*(1991), 49–65.

Batáry, P., Dicks, L. V., Kleijn, D., & Sutherland, W. J. (2015). The role of agri-environment schemes in conservation and environmental management. *Conservation Biology*, *29*(4), 1006–1016. https://doi.org/10.1111/cobi.12536

Blöchliger, H. J. (1994). Main results of the study. *The Contribution of Amenities to Rural Development, OECD, Paris, France*.

Brouwer, R., Langford, I. H., Bateman, I. J., & Turner, R. K. (1999). A meta-analysis of wetland contingent valuation studies. *Regional Environmental Change*, *1*(1), 47–57. https://doi.org/10.1007/s101130050007

Carson, R. T., Flores, N. E., & Meade, N. F. (2001). Contingent valuation: Controversies and evidence. *Environmental and Resource Economics*, *19*(2), 173–210. https://doi.org/10.1023/A:1011128332243

Cooper, T., Hart, K., & Baldock, D. (2009). *Provision of public goods through agriculture in the European Union*. London.

Cullen, P., Dupraz, P., Moran, J., Murphy, P., O'Flaherty, R., O'Donoghue, C., … Ryan, M. (2018). Agri-environment scheme design: Past Lessons and Future Suggestions. *Eurochoices - Point de Vue (Forthcoming)*.

Derissen, S., & Quaas, M. F. (2013). Combining performance-based and action-based payments to provide environmental goods under uncertainty. *Ecological Economics*, *85*, 77–84. https://doi.org/10.1016/j.ecolecon.2012.11.001

Diamond, P. A., & Hausman, J. A. (1994). Contingent Valuation: Is Some Number Better than No Number? *Journal of Economic Perspectives*, *8*(4), 45–64. https://doi.org/10.1257/jep.8.4.45

Dupraz, P., Latouche, K., & Turpin, N. (2009). Threshold effect and co-ordination of agri-environmental efforts. *Journal of Environmental Planning and Management*, *52*(5), 613–630. https://doi.org/10.1080/09640560902958164

Engel, S., Pagiola, S., & Wunder, S. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics*, *65*(4), 663–674. https://doi.org/10.1016/j.ecolecon.2008.03.011

EPA. (2016). *Ireland's Environment: An Assessment 2016*. Environmental Protection Agency (EPA), Wexford.

EPA. (2017a). *Ireland's Provisional Greenhouse Gas Emissions - 1990-2016*. Environmental Protection Agency, Wexford.

EPA. (2017b). *Water Quality in Ireland 2010-2015*. Environmental Protection Agency, Wexford.

European Commission. (2013). *Rural Development in the EU - Statistical and Economic Information Report 2013*.

Finn, J. A., & Ó hUallacháin, D. (2012). A review of evidence on the environmental impact of Ireland's Rural Environment Protection Scheme (REPS). *Biology & Environment: Proceedings of the Royal Irish Academy*, *112*(1), 1–24. https://doi.org/10.3318/BIOE.2011.19

Gibbons, J. M., Nicholson, E., Milner-Gulland, E. J., & Jones, J. P. G. (2011). Should payments for biodiversity conservation be based on action or results? *Journal of Applied Ecology*, *48*(5), 1218–1226. https://doi.org/10.1111/j.1365-2664.2011.02022.x

Hanley, N., Kirkpatrick, H., & Oglet, D. (1998). Principles for the Provision of Public Goods from Agriculture : Modeling Moorland Conservation in Scotland. *Land Economics*, *74*(1), 102–113.

Hart, K., Baldock, D., Weingarten, P., Osterburg, B., Povellato, A., Vanni, F., … Boyes, A. (2011). *What tools for the European agricultural policy to encourage the provision of public goods?* London: Institute for European Environmental Policy.

Howley, P., Donoghue, C. O., & Hynes, S. (2012). Exploring public preferences for traditional farming landscapes. *Landscape and Urban Planning*, *104*(1), 66–74. https://doi.org/10.1016/j.landurbplan.2011.09.006

Hynes, S., & Garvey, E. (2009). Modelling Farmers' Participation in an Agri-environmental Scheme using Panel Data: An Application to the Rural Environment Protection Scheme in Ireland. *Journal of Agricultural Economics*, *60*(3), 546–562. https://doi.org/10.1111/j.1477-9552.2009.00210.x

Jones, J. I., Murphy, J. F., Anthony, S. G., Arnold, A., Blackburn, J. H., Duerdoth, C. P., … Skates, J. (2016). Do agri-environment schemes result in improved water quality? *Journal of Applied Ecology*. https://doi.org/10.1111/1365-2664.12780

Kapoor, I. (2001). Towards participatory environmental management? *Journal of Environmental Management*, *63*(3), 269–279. https://doi.org/10.1006/jema.2001.0478

Kleijn, D., Baquero, R. A., Clough, Y., Díaz, M., De Esteban, J., Fernández, F., … Yela, J. L. (2006). Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, *9*(3), 243–254. https://doi.org/10.1111/j.1461-0248.2005.00869.x

Lastra-Bravo, X. B., Hubbard, C., Garrod, G., & Tolón-Becerra, A. (2015). What drives farmers' participation in EU agri-environmental schemes?: Results from a qualitative meta-analysis. *Environmental Science & Policy*, *54*(December), 1–9. https://doi.org/10.1016/j.envsci.2015.06.002

Loomis, J. B., & White, D. S. (1996). Economic benefits of rare and endangered species: summary and meta-analysis. *Ecological Economics*, *18*(3), 197–206. https://doi.org/10.1016/0921-8009(96)00029-8

Matzdorf, B., Kaiser, T., & Rohner, M. S. (2008). Developing biodiversity indicator to design efficient agri-environmental schemes for extensively used grassland. *Ecological*

Indicators, *8*(3), 256–269. https://doi.org/10.1016/j.ecolind.2007.02.002

Mauerhofer, V., Hubacek, K., & Coleby, A. (2013). From polluter pays to provider gets: Distribution of rights and costs under payments for ecosystem services. *Ecology and Society*, *18*(4). https://doi.org/10.5751/ES-06025-180441

Murphy, G., Hynes, S., Murphy, E., & O'Donoghue, C. (2014). An investigation into the type of farmer who chose to participate in Rural Environment Protection Scheme (REPS) and the role of institutional change in influencing scheme effectiveness. *Land Use Policy*, *39*, 199–210. https://doi.org/10.1016/j.landusepol.2014.02.015

Newig, J., & Koontz, T. M. (2014). Multi-level governance, policy implementation and participation: The EU's mandated participatory planning approach to implementing environmental policy. *Journal of European Public Policy*, *21*(2), 248–267. https://doi.org/10.1080/13501763.2013.834070

Newig, J., & Kvarda, E. (2012). Participation in environmental governance: legitimate and effective. In K. Hogl, E. Kvarda, R. Nordbeck, & M. Pregernig (Eds.), *Environmental Governance - the challenge of legitimacy and effectiveness* (pp. 29–45). Edward Elgar Publishing.

Noonan, D. S. (2003). Contingent Valuation and Cultural Resources: A Meta-Analytic Review of the Literature. *Journal of Cultural Economics*, *27*(3/4), 159–176. https://doi.org/10.1023/A:1026371110799

OECD. (1972). Recommendation of the council on guiding principles concerning international economic aspects of environmental policies. *Annex I, Adopted at the Council's 239th Meeting (May 26,1972). "Note on the Implementation of the Polluter Pays Principle," paragraph.*, *Vol. 2*.

Prager, K. (2015). Agri-environmental collaboratives for landscape management in Europe. *Current Opinion in Environmental Sustainability*, *12*, 59–66. https://doi.org/10.1016/j.cosust.2014.10.009

Richardson, L., Loomis, J., Kroeger, T., & Casey, F. (2015). The role of benefit transfer in ecosystem service valuation. *Ecological Economics*, *115*, 51–58. https://doi.org/10.1016/j.ecolecon.2014.02.018

Roberts, W. M., Gonzalez-Jimenez, J. L., Doody, D. G., Jordan, P., & Daly, K. (2017). Assessing the risk of phosphorus transfer to high ecological status rivers: Integration of nutrient management with soil geochemical and hydrological conditions. *Science of the Total Environment*, *589*, 25–35. https://doi.org/10.1016/j.scitotenv.2017.02.201

Rydin, Y. (2007). Re-examining the role of knowledge within planning theory. *Planning Theory*, *6*(1), 52–68. https://doi.org/10.1177/1473095207075161

Schroeder, L. A., Isselstein, J., Chaplin, S., & Peel, S. (2013). Agri-environment schemes: Farmers' acceptance and perception of potential "Payment by Results" in grassland-A case study in England. *Land Use Policy*, *32*, 134–144. https://doi.org/10.1016/j.landusepol.2012.10.009

Tobey, J. A., & Smets, H. (1996). The Polluter-Pays Principle in the Context of Agriculture and the Environment. *The World Economy*, *19*(1), 63–87. https://doi.org/10.1111/j.1467-9701.1996.tb00664.x

Van Den Hove, S. (2000). Participatory approaches to environmental policy-making: The European Commission Climate Policy Process as a case study. *Ecological Economics*, *33*(3), 457–472. https://doi.org/10.1016/S0921-8009(99)00165-2

van der Horst, D. (2007). Assessing the efficiency gains of improved spatial targeting of policy interventions; the example of an agri-environmental scheme. *Journal of Environmental Management*, *85*(4), 1076–1087. https://doi.org/10.1016/j.jenvman.2006.11.034

Wu, J. J., & Babcock, B. A. (1999). The relative efficiency of voluntary vs mandatory environmental regulations. *Journal of Environmental Economics and Management*, *38*(2), 158–175. https://doi.org/10.1006/jeem.1999.1075