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Economic and environmental sustainability performance of environmental policies in agriculture

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Economic and Environmental Sustainability Performance of Environmental Policies in Agriculture

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This report reviews the literature on the effects of agri-environmental policies on environmental sustainability and economic performance in agriculture. Examining these twin impacts is essential for understanding the scope for "win-win" policies which improve both types of performance, and where trade-offs between economic and environmental objectives may arise. The review considers findings on several underlying questions: i) whether agri-environmental policy instruments successfully deliver on their objectives to improve the environmental performance of agriculture, and ii) whether agri-environmental policy instruments slow down productivity growth or if they contribute to stimulating productivity growth and improved environmental outcomes. As part of this latter question, this review considers the impacts of agri-environmental policies on innovation, economic performance and structural change in agriculture. It brings together literature from across a range of disciplines, including evidence from over 160 papers. As a whole, the reviewed literature identifies significant "room for improvement" in both the effectiveness of agri-environmental policies for improving agricultural sustainability and their economic efficiency, particularly in relation to hybrid instruments (e.g. cross-compliance) and voluntary agri-environmental schemes (AES).

Keywords: Agri-environmental policy, environmental sustainability, economic performance, Porter Hypothesis, innovation, AES

JEL Codes: Q15, Q18

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Table of contents

Table

Executive Summary

This report reviews the literature on the effects of agri-environmental policies on environmental sustainability and economic performance in agriculture. Examining these twin impacts is essential for understanding what scope there might be for “win-win” policies which improve both types of performance, and where trade-offs between economic and environmental objectives may arise.

This review is part of a suite of work which aims to help consolidate the knowledge base on which the OECD’s agricultural innovation and sustainable productivity country reviews are built. It also informs efforts to model the complex and dynamic effects of agri-environmental policies, in particular by considering policy impacts on agricultural innovation, structural change, productivity and sustainability performance, thereby enhancing the OECD’s analytical capacity in relation to policies for sustainable and productive agriculture.

At the farm level, the empirical literature does not establish a simple relationship between environmental performance and economic performance in agriculture. It does however identify a number of factors that positively (or at least non-negatively) impact this link (i.e. increase the likelihood of a positive relationship between environmental and economic performance). These include: farm size, presence of demand for environmentally-differentiated goods, and the ability to respond “proactively” or “dynamically” to external pressures to improve environmental performance; that is, to voluntarily change production processes towards being more environmentally sustainable in anticipation of external factors such as regulation or changing demand, or to embrace change (especially regulatory change) rather than resisting it. This finding suggests that policies should be designed with such pre-existing relationships in mind: for example, policy-makers could couple introduction of new regulations with innovation policies targeted at helping farmers to embrace change (becoming more “dynamic”), or with a consumer-side policy aimed to stimulate demand for environmentally sustainable products.

Looking at policy performance, the literature identifies significant “room for improvement” in both the effectiveness of agri-environmental policies for improving agricultural sustainability and their economic efficiency, particularly in relation to hybrid instruments (e.g. cross-compliance) and voluntary agri-environmental schemes (AES). While there was considerable heterogeneity found across different contexts, in general action-oriented (also called practice-based) measures often performed poorly on effectiveness and efficiency criteria. Site-specific economic and biological conditions cause substantial heterogeneity in the environmental effectiveness and cost-effectiveness of agri-environmental policies; therefore, policies need to be targeted to specific conditions and should take into account bio-physical relationships, farmer decision-making and broader economic relationships. Where agri-environmental policies do *not* succeed in substantively improving environmental performance, there is a real risk that rather than creating a “win-win”, unintended outcomes could occur. For example, the policy could be spending public funds for very little tangible benefit and inappropriately subsidising economic performance; alternatively, the policy may be funding insignificant gains in environmental performance but causing significant *decreases* in economic performance, such that the benefits are far outweighed by the costs.

Few studies assess the productivity impacts of agri-environmental regulations, and available evidence suggests that these impacts are mixed. Importantly, however, there are several cases identified where regulations can have *positive* effects on productivity, at the same time as improving environmental performance. One important pathway via which this occurs—known as the Porter Hypothesis—is that regulation can spur innovation that ultimately provide economic benefits which outweigh compliance costs.

Based on the literature surveyed in this review, hybrid instruments (e.g. environmental cross-compliance) appear generally not to have statistically significant impacts on participating farms' productivity. Where impacts do occur, they can be positive or negative but are generally small in magnitude. When coupled with the evidence that such instruments have often been unsuccessful in substantially improving environmental performance, the conclusion reached is that, while such mechanisms appear promising in theory, in practice they have often been unsuccessful at stimulating real change.

There is little evidence that economic agri-environmental policy instruments (AES, market-based approaches) negatively impact participating farmers' economic performance. This is in line with economic theory, which posits that, since these mechanisms are voluntary, a rational profit-maximising farmer would not enter if they expected their economic performance to suffer. Nevertheless, there are some isolated cases where voluntary mechanisms do appear to have had a negative impact on participating farmers' productivity or economic performance more generally; these cases can provide lessons for policy-makers on what to avoid. Given the evidence of patchy success on achieving environmental effectiveness (and cost-effectiveness), it appears that these mechanisms have room for improvement but appear capable of delivering "win-win" outcomes for both environmental and economic performance of agriculture, although challenges such as mitigating selection bias and ensuring additionality remain.

The possibility that policy impacts differ with and without (or before and after) innovation means that the dynamic impact of policies may differ from static impacts and that "win-win" outcomes may only occur over time (i.e. lagged effects). Innovation, particularly the ability for a farm to "embrace change" (i.e. innovatively respond to external stimuli), has been found to be one of the factors determining whether regulations positively or negatively impact economic performance. Stimulating innovation can also be key to the design of successful policy instruments. For example, results-oriented mechanisms are considered to have the potential to stimulate on-farm innovation and adaptation of environmental management practices to local conditions, and thereby achieve lasting improvements in agricultural environmental sustainability. However, empirical evidence on the degree to which results-oriented mechanisms actually spur innovation is scant, not least because results-oriented mechanisms are still in their infancy. Further, there are many definitions of exactly what constitutes a result-oriented mechanism. Nevertheless, there is much optimism, and burgeoning empirical evidence, that result-oriented measures will be both more effective and more efficient than practice-based mechanisms. Additional work in this area is needed to clearly define the spectrum of different payment structures and consider which type(s) of payments are optimal under which conditions.

Lastly, the literature shows that AES tend to decelerate the pace of structural change by allowing land retirement, fallow or low-management land uses to become a (more) profitable land use option for farmers. However, some types of AES can also incentivise expansion of cultivated area. Thus, impacts are likely to be stronger in terms of land use change than total farm numbers or entry and exit decisions. In terms of impacts on farm size, there are indications that the structural impacts of agri-environmental policies are likely to be positive for small farms but uncertain for medium-sized and large farms.

This review attempts to consolidate evidence to answer the dual questions of:

- whether policies which aim to stimulate improved environmental performance of agriculture necessarily come at the cost of retarding economic performance (for example, by lowering the farm productivity);
- whether in fact there may be the possibility of "win-win" policies which stimulate improved environmental and economic performance.

These questions are of vital importance of policy-makers who are committed to ensuring that agriculture achieves the dual objectives of food security and sustainable natural resource use.[1]

The review seeks the answer to these questions by considering findings on several underlying questions: i) whether agri-environmental policy instruments successfully deliver on their objectives to improve the environmental performance of agriculture, and ii) whether agri-environmental policy instruments slow down productivity growth, or instead contribute to stimulating both productivity growth and improved environmental outcomes. As part of considering this latter question, this review considers the impacts of agri-environmental policies on innovation, economic performance and structural change[2] in agriculture.

This review is not intended to comprise a systematic discussion of the all available empirical evaluations of agri-environmental policies: such a task is far beyond the scope of the present exercise. Rather, this review brings together literature from across a range of disciplines, and seeks to provide a coherent narrative on the general findings relevant to the central questions. While this review makes no claim to be exhaustive, it brings together findings from over 160 papers, mostly peer-reviewed journal articles.

The review was conducted by first constructing an overall conceptual approach, which is implemented via the structure of this report. This approach consists of:

- Defining the policy instruments which form the focus of the review—Section 1.
- Considering broadly whether there is any evidence of a fundamental relationship between environmental and economic performance in agriculture (i.e. without considering the impact of specific policy interventions)—Section 2.
- Reviewing the evidence on the environmental impacts of broad categories policy instruments—Section 3.
- Reviewing the evidence on the economic impacts of broad categories policy instruments, including by considering impacts on productivity, innovation and structural change—Section 4.
- Bringing together these findings into a coherent narrative for policy-makers—Section 5 and Executive Summary.

Having defined this structure, for each section the review was approached by conducting online searches of OECD publications (OECD iLibrary) and the peer-reviewed literature via Google Scholar and other databases such as EconLit. Bibliographies in key papers were also searched to identify further papers of interest. In addition, certain grey literature (for example, policy evaluations published by governments) were also used. Discussion of the Porter Hypothesis in Section 4 draws in part from a literature review undertaken for the OECD by Stefan Ambec, Céline Nuages and Arnaud Reynaud (Toulouse School of Economics). In general, reviewed literature relates to policies in OECD countries, and are from English-language publications (including English translations of texts in other languages, where available).

[1] This question is also relevant for other objectives such as the maintenance of successful rural economies, but this objective is not the central focus for this review.

[2] This literature focusses in particular on three dimensions of structural change: the entry or exit of farms, change in farm size, and modification of land use. See Section 2.4.3.

1. Agri-environmental policy instruments covered in this review

This review focuses on the impacts of a range of commonly-used environmental policy instruments used in agriculture (Table 1). These instruments can be broadly categorised into *regulatory instruments* which directly impose environmental requirements on farmers; *hybrid instruments* which tie environmental requirements as a condition for participation in other (non-environmental) policy measures; and *economic instruments* which incentivise farmers to provide environmental services (improve their environmental performance). While policy instruments may affect several sectors of the economy, this review provides evidence relating to environmental policies that are specific to the agriculture sector (referred to as "agri-environmental policies") wherever possible, but in some instances supplements this with references to studies on the impacts of broader environmental policies.

Table 1. selected agri-environmental policy instruments

Category	Instrument	Examples
Regulatory instruments	Environmental standards	Input standards, technology standards, performance (output) standards
Hybrid instruments	Environmental "cross-compliance" requirements	Cross-compliance mechanisms, baseline eligibility requirements
Economic Instruments	Agri-environmental payment schemes	Payments for environmentally sustainable on-farm practices, public investment in structural adjustment towards "greener" agricultural systems
	Environmental taxes and tariffs	Performance taxes, input taxes, water tariffs
	Tradeable allowances	Emissions trading schemes, tradeable offset schemes, water markets, in-lieu-fee programmes
	Publicly-funded investment in property rights for the environment	Purchase of water rights from agricultural enterprises, with purchased rights being allocated to the environment

Source: Adapted from OECD (2010[4]) and Hardelin and Lankoski (2018[5]).

1.1. Regulatory instruments

Environmental regulations constitute a key component of policy packages in most OECD countries to reduce environmental pressures from agriculture. They can be differentiated to account for heterogeneity of site-productivity of ecosystem service provision (for example proximity to surface watercourses). They can also protect ecosystem services used by agriculture such as land resources and habitat conservation useful for biological pest control (Hardelin and Lankoski, 2018[5]).

Regulatory instruments specify environmental standards which constrain the choice set of farmers with the intent to force improved environmental performance. They often prescribe a specific level of environmental performance or technology to be used for an individual actor (e.g. farm) that must be met, or restrict the use of environmentally-damaging inputs or production processes.

Technology standards (including *input standards*[3]) place restrictions on the technologies that can be used for production or consumption. They rely on an engineering approach to environmental protection: the environmental problem is expected to be fixed, or at least mitigated, by specifying the technology (or, more broadly, production processes) that can be used. Examples abound: these include banning the more toxic

[3] The distinction between technology and input standards depends on the definition of "technology". We take a broad approach where technology refers to the production process generally; as such, input restrictions are one form of technology standard.

pesticides, regulations specifying the maximum amount of inputs (e.g. fertiliser) that may be applied per area, maximum stocking density or livestock exclusion regulations, land clearing moratoria, etc.[4]

Performance or emission standards are other technological restrictions that are defined in quantitative terms rather than applying to the technology itself. They can be expressed in absolute value when pollutant emissions or inputs are capped (e.g. limits on wastewater discharges from concentrated livestock facilities), or in relative terms, i.e. per unit of output (e.g. CO_2 equivalent of greenhouse gas (GHG) emissions per kilowatt hour).

Technological and performance standards are imposed on economic agents (producers and consumers), and restrict their possibilities of complying with environmental regulations. They require some monitoring efforts and penalties in case of non-compliance. For these reasons, they are often referred to as "command-and control" instruments. By imposing greener inputs or production processes, technological standards are expected to enhance the adoption of new technology. However, they provide few incentives to innovate or to improve the technology beyond that imposed by the regulation. Performance standards are better in this respect because they do not impose a specific technology to comply with the standard. Firms are free to pick the technology that is better suited to their needs and characteristics. Furthermore, firms can also develop their own technology, or improve the existing technology, to meet the performance standards.

1.2. Hybrid instruments: Cross-compliance mechanisms

Hybrid instruments embed regulatory-type requirements into a broader policy mechanism that a farmer can choose to participate in. Common examples include:

- Cross-compliance requirements under Pillar I of the European Union Common Agricultural Policy (CAP): in this case, a farmer is required to implement certain environmental "best management practices" (BMPs) in order to receive EU direct payments.
- Baseline eligibility requirements for participation in environmental markets (trading schemes): in this example, a farmer must achieve a certain level of environmental performance or implement prescribed BMPs before undertaking further activities which are eligible to generate credits (e.g. water quality credits, carbon credit, etc.)[5] that can be sold in a trading scheme. In some cases, buyers participating in environmental markets are also required to achieve a prescribed level of pollution abatement "on-site" before becoming eligible to purchase abatement credits in a trading scheme.

[4] It is acknowledged that banning pesticides may not necessarily lead to an increase in ecosystem services from agriculture, for example, if the ban of a particular chemical leads to increased use of another chemical that may also pose a risk to the environment. Nevertheless, pesticide bans are a commonly-used policy tool in OECD countries.

[5] Such baseline requirements are a common feature of water quality trading programmes operating under the federal Clean Water Act (CWA) in the United States. The US Environmental Protection Agency (which administers the CWA in partnership with state, local and tribal authorities, describes seller baselines as follows: "A seller's baseline is the level of discharge it is otherwise required or expected to attain prior to generating credits. A nonpoint source seller would be expected to meet its TMDL [total maximum daily load] load allocation or, if there is no TMDL, it would be expected to meet any state and local requirements before it can generate credits." (US EPA Office of Water, n.d.[161]).

1.3. Economic instruments: Taxes, subsidies and emission allowances

Economic instruments (also called "market-based instruments") do not attempt to prescribe the means to achieve a specific level of environmental harm for an individual actor (e.g. farm). Rather, they create or alter the relative incentives faced by these actors, such that at least some actors voluntarily choose to improve their environmental performance (e.g. reduce nutrient runoff, conserve soil and water resources, etc.). The level of incentive can in theory be calibrated to achieve a particular level of environmental performance at an aggregate level or can directly be defined in some case of cap-and-trade systems.

Economic instruments used in OECD member countries' agri-environmental policies include:

- Environmental taxes or tariffs
- Environmental subsidies (e.g. subsidies for sustainable practices on-farm, public investment in structural adjustment towards "greener" agricultural systems[6])
- Environmental tenders or auctions: payments for ecosystem services
- Tradeable allowance programmes (e.g. emissions trading schemes, tradeable offset schemes, etc.)
- In-lieu-fee programmes: this is a hybrid measure that is somewhat like a tax in some aspects and also like an offset programme.[7]

Beyond these regulatory instruments and economic instruments (and hybrids), other agri-environmental policy instruments are available and may affect the outcomes of interest. Examples include public investment in R&D to improve agricultural sustainability and publicly-funded technical assistance and extension services (OECD, 2010[4]). However, the relationship between these policy instruments and farm environmental and economic performance (as well as other impacts such as structural change) is less studied and as such are in general excluded from this review.

It is further acknowledged that environmental policy instruments that are not specific to agriculture can have significant impacts on outcomes in the agriculture sector. Important examples are the impact of "green" energy and fuel policies (e.g. carbon tax applied on fuels, reforms phasing out fossil fuel subsidies, biofuel mandates and public investment in green technologies), climate change mitigation and adaptation policies (e.g. policies stimulating investment in rural information-communication technology (ICT) infrastructure), forestry and native vegetation policies or other policies affecting land use.

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[6] One example is government support to permanently retire agricultural working lands—e.g. government-funded conservation easements (note the extent to which this causes structural adjustment depends on the context; subsidies for 'set asides' could also be considered as a subsidy for an on-farm practice; it depends on the degree of change in land allocation and whether this change causes farm exit). Another example would be a policy which subsidised farmers for purchasing "green" technology—e.g. farm machinery that used less fuel, irrigation systems that use less water. It is acknowledged that the ultimate environmental impact of such measures depends on the programme design.

[7] In-lieu-fee programmes operate somewhat like offset programmes, in that they start from the point of requiring regulated entities to offset environmentally damaging activities undertaken on-site. However, rather than (or "in lieu") of contracting for a specific offset, the entities pay fees into a central fund, and the fund manager subsequently and separately contracts for conservation activities which offset (perhaps in aggregate) the activities undertaken by the fee-paying entities. Thus, from the perspective of these entities, the programme operates like a tax, as these entities are under no obligation to secure offsets. From the point of view of suppliers of offsets, this programme operates similarly to a traditional offset scheme, albeit likely with fewer buyers.

2. Is environmental performance of agriculture correlated with economic performance?

A fundamental starting point for assessing the potential for policies to create synergies or trade-offs between environmental performance and productivity (or economic performance more broadly) in agriculture is to understand whether, and how, these performance types are linked before any policy intervention occurs. While stylised facts about the contribution of agriculture to natural resource degradation[8] may imply a trade-off between environmental and economic performance at the aggregate level, correlations in specific contexts may be dependent on a number of factors which are relevant for policy-making.[9]

Many different conceptions that can broadly fit within the notions of “environmental performance” and “economic performance” exist in the literature. Environmental performance can encompass participatory metrics such as implementation of some kind of farm-level environmental management strategy (without linking this to a particular physical impact); implementing specific production methods (e.g. conservation tillage or other “best management practices”); generally reducing input use (e.g. land retirement, decreasing water, fertiliser or nutrient inputs etc.); or achieving measured environmental gains (e.g. reducing nutrient runoff to downstream water bodies). Similarly, economic performance encompasses measures of productivity, competitiveness, economic efficiency (technical, allocative and dynamic), and financial profitability.

The OECD’s agricultural sustainable productivity framework focuses specifically on productivity growth when considering economic performance, as this is a holistic measure that considers the creation of economic value for a given level of inputs, and moreover is a concept which lends itself to empirical quantitative analysis. However, this focus on productivity growth does not necessarily allow for explicit analysis of aspects which may be of concern to policymakers, and which are the focus of several studies reviewed in this section—in particular profitability and producer surplus measures which relate to productivity but which are also influenced by the distribution of economic surplus throughout agri-food supply chains. In this review, where possible information on the specific measures and concepts used in the reviewed studies will be cited.

Barba-Sánchez and Atienza-Sahuquillo (2016[6]) identify that the benefits to a farm of good environmental performance can come through different angles, such as cost reduction, improved consumer demand or “social licence”, and product differentiation (e.g. organic products). As such, even though pursuing environmental performance may be costly to a farmer (i.e. improving environmental performance may generate cost increases rather than reductions), there are other avenues which may mitigate these additional costs and lead to improved economic performance overall. Bernués et al. (2011[7]) similarly note that environmental and social aspects of production processes (e.g. animal welfare, location of production, “environmentally friendly production”) are increasingly important for consumers, and therefore that improved environmental performance can result in improved economic performance by stimulating

[8] For example, FAO (2003[164]) identifies agriculture as having major contributions to air pollution, climate change, land degradation, over-extraction of freshwater resources, water logging, salinisation, water impairment by fertilisers, pesticides and livestock wastes, and loss of biological and ecological diversity.

[9] Even if there *is* a positive correlation between environmental and economic performance, this does not necessarily preclude the need for policy intervention. In this case, the task of the policy is somewhat different: for example, rather than seeking to improve environmental performance without significantly harming economic performance, the policy may instead concentrate on incentivising the “poor performers” to improve and related distributional questions.

demand for products which are differentiated by environmental quality. This argument is supported by López-Gamero, Molina-Azorín and Claver-Cortés (2010[8]) and Schaltegger and Synnestvedt (2002[9]).

Echoing the theoretical literature, the empirical literature does not conclusively establish that there is a broad, simple relationship between environmental performance and economic performance. Rather, this relationship appears to be mediated via a number of different factors, as discussed below. While there are many studies investigating these relationships, relatively few are specific to the agriculture sector. Therefore, this section draws on this broader literature to assess general results, and cites evidence specific to agriculture where available.

Two large studies exploring evidence on the linkages between economic performance and environmental performance are Horváthová (2010[10]) and Lankoski (2010[3]).

Horváthová (2010[10]) conducted a statistical meta-analysis of existing research into the relationship between "environmental performance" and "financial performance" (44 studies), not limited to the agricultural sector. Results varied, and it was found that the methodology employed can affect the likelihood of finding a negative link. In particular, studies employing correlation coefficient analysis or portfolio studies were found to be more likely to find a negative link. Horváthová noted the possibility of omitted variable bias in these studies. In contrast, the author found that a positive link is more likely to be found in studies which use a qualitative environmental variable. In terms of (non-methodological) explanatory variables, Horváthová found that a positive link between environmental performance and economic (financial) performance was more likely in common law countries (the United States, Canada and the United Kingdom) than in civil law countries. While the author attempted to test whether industry type significantly impacted the link, results were inconclusive and this was recommended as an area for future research.

Lankoski (2010, p. 32[3]) reviewed studies in the economics and management literatures which empirically examine the links between environmental performance and firm competitiveness. Lankoski found a large degree of heterogeneity, stating that: "[v]arious reviews have identified altogether almost 50 different methodological or measurement problems in the body of research on the firm-level relationship between corporate responsibility (including environmental responsibility) and competitiveness factors explaining the differences in results". However, the author concludes that, on balance, this literature has found that the impact is "slightly positive, or at least not negative".

In an analysis of the Indian sugar industry, Murty, Kumar and Paul (2006[11]) find a positive relationship between water conservation efforts and firm efficiency, and note the potential for complementarities—such as cost savings from water recovery and recycling—between output (sugar) production and pollution load reductions, particularly where innovation relates to process change rather than "end-of-pipe" treatment technologies.

A number of authors suggest that "environmental proactivity"—defined by Barba-Sánchez and Atienza-Sahuquillo (2016, p. 2[6]) as "the voluntary adoption of measures which help reduce the environmental impact"—is a key mediating factor in the relationship between economic and environmental performance. Empirical examinations of this hypothesised relationship generally seem to support it, although not universally. For example, Liu, Guo and Chi (2015[12]) performed a meta-analysis of 68 studies examining this relationship, and found overall that environmental proactivity positively impacts firms' economic and environmental performance. Further, they found that the effect on economic performance is stronger for Chinese firms than Western firms and also depends on other factors such as regulatory context, managerial mind-set and stakeholder norms. Examining the Spanish wine industry, Barba-Sánchez and Atienza-Sahuquillo (2016, p. 10[6]) similarly find that "not only does [environmental proactivity] reduce resource consumption and waste generation, thereby minimizing the environmental impact of wineries, but these environmental results also have a positive impact on perceived corporate performance."

However, Albertini's (2013[13]) meta-analysis of 52 studies examining the link between corporate environmental performance and financial performance shows a more complex picture: the author emphasises that environmental proactivity (termed "corporate environmental management" in her study) is a "meta-construct" which itself consists of many variables which have different impacts on both environmental and economic performance. Further, measures of environmental performance and financial performance differed widely across the studies included in the meta-analysis, and these differences were found to significantly impact results.

López-Gamero, Molina-Azorín and Claver-Cortés (2010[8]) find evidence of a two-way relationship between environmental proactivity and firm performance (measured in terms of growth in added value, economic development and financial profitability), and that the precise nature of the link can depend on factors such as a firm's competitive advantage and size. These authors argue that firm size may matter because larger firms can absorb more risk and have a greater ability to hire and train new staff, thus allowing them to more effectively mitigate risks associated with environmental proactivity.

Ramanathan et al. (2017[14]) used nine case studies of UK and Chinese firms to examine the linkages between environmental performance, environmental regulation and "private benefits of sustainability". They found that a key factor mediating the relationship between environmental performance and economic performance was whether firms responded "dynamically"[10] to external pressures to improve environmental performance, whether that pressure came from regulation or other sources such as consumer preferences, strategic positioning or economic pressures. Their results show that "firms that take a dynamic approach to proactively managing their environmental performance are generally able to improve the private benefits of sustainability (e.g. by reducing consumption of energy and raw materials that result in reduced waste or pollution, or enjoying better market performance) better than those firms who do not prioritise environmental performance as highly." (p. 89).

Buckley (2012[15]) similarly finds evidence from a study of Irish farmers' views of implementing the EU Nitrates Directive (NiD) that farmers differ in the attitudes and response to environmental regulation. Some farmers (termed "benefit accepters") have a positive view on environmental regulation, and embrace change in a proactive manner; others ("constrained productionists") are unconvinced about the appropriateness of NiD requirements (and yet other groups are relatively unaffected). This research suggests that farmers' attitudes about the environment and about environmental regulation may affect their willingness and ability to respond in a positive or "dynamic" way, which in turn may ultimately affect the relationship between their environmental and economic performance.

In summary, the literature indicates that the link between environmental and economic performance, even in the absence of regulations or environmental policies, depends on a number of factors. On balance, factors which are found to positively (or at least non-negatively) impact this link include environmental proactivity, firm size, presence of demand for environmentally-differentiated goods, and a firm's ability to respond "proactively" or "dynamically" to external pressures to improve environmental performance. However, many methodological issues exist, and one implication of the possibility that results vary across industries is that there is a need for more studies specific to agriculture to examine the nature of these relationships for farmers.

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[10] In a follow-up article, Ramanathan, Ramanathan and Bentley (2018, p. 136[90]) expand on their notion of what it means for a firm to respond "dynamically" to regulation: "Rather than just oppose legislation and try to slow its passage, a firm can see positive results if it embraces the regulations and can actually use it as the basis of competitive advantage...the innovative dynamic firm can use it as an opportunity to move into new product markets, move to leaner and greener production processes, which reduce unnecessary energy consumption and material inputs, as well as turning mandatory recycling into a profitable remanufacturing process."

3. Agri-environmental policy impacts on agricultural sustainability

Agri-environmental policies are fundamentally based on the view that the environmental performance of the agriculture sector (or a sub-section thereof) needs improvement. Therefore, before examining the extent to which agri-environmental policies can improve economic performance, it is relevant to consider evidence on the extent to which environmental policies do indeed positively impact the environmental performance or "sustainability"[11] of the agriculture sector. If agri-environmental policies do not succeed in substantively improving environmental performance, there is a real risk that rather than creating a "win-win", unintended outcomes could occur: for example, the policy could be spending public funds for very little tangible benefit and inappropriately subsidising economic performance; alternatively, the policy may be funding insignificant gains in environmental performance but causing significant decreases in economic performance, such that the benefits are far outweighed by the costs.

There is a rich literature assessing the performance of agri-environmental policies; a comprehensive review of this literature is beyond the scope of this paper. However, the following sub-sections provide a broad overview of key factors identified in the literature which affect the performance of agri-environmental policies, in terms of i) their success (or otherwise) in achieving their stated environmental objectives (the "environmental effectiveness" criterion—see Section 3.1), and ii) their related cost-effectiveness (the "cost-effectiveness" criterion—see Section 3.2) (OECD, 2010[4]).

3.1. Environmental effectiveness of agri-environmental policies

Environmental effectiveness is "the capacity of the instruments to achieve stated environmental goals or targets of practices" (OECD, 2010[4]). According to Börner et al. (2017[16]), environmental effectiveness is determined by four main factors:

- Programme costs (i.e. transaction and implementation costs)
- The direct changes in land or resource use induced by the programme, compared to the counterfactual without the programme
- Indirect changes in land or resource use (e.g. spillovers onto land outside of the programme), compared to the counterfactual
- The effects these changes in land and resource use have on the actual provision of environmental goods and services.[12]

Evidence on environmental effectiveness of agri-environmental policy measures can be drawn from both the ecological and economics literature. This literature demonstrates a wide degree of success when assessed by this measure, with some instruments only having limited success in achieving their stated goals, while others achieve them.

[11] It is acknowledged that there is no consensus on the definition of "sustainability", and that several forms of sustainability can be considered—e.g. environmental sustainability, social sustainability, economic sustainability. In this review, the term "agricultural sustainability" is used synonymously with environmental performance of the agriculture sector.

[12] Beyond this, Hardelin and Lankoski (2018[5]) identify that political feasibility is another important factor.

3.1.1. Regulatory instruments

Water quality regulation

Van Grinsven et al. (2012[17]) study the environmental impacts resulting from implementation of the EU Nitrates Directive (NiD) over the period 1995 to 2007. They find that:

> *The most significant environmental effect of the implementation of the NiD since 1995 is a major contribution to the decrease of the soil N balance (N surplus), particularly in Belgium, Denmark, Ireland, the Netherlands and the United Kingdom. This decrease is accompanied by a modest decrease of nitrate concentrations since 2000 in fresh surface waters in most countries. This decrease is less prominent for groundwater in view of delayed response of nitrate in deep aquifers. In spite of improved fertilisation practices, the southeast of the Netherlands, the Flemish Region and Brittany remain to be regions of major concern in view of a combination of a high nitrogen surplus, high leaching fractions to groundwater and tenacious exceedance of the water quality standards…[However,] [d]ifferences between procedures in member states to assess nitrogen balances and water quality and a lack of cross-boundary policy evaluations are handicaps when benchmarking the effectiveness of the NiD.*

Marconi and Raggi (2015[18]) study the impacts of agri-environmental measures in Emilia-Romagna, Italy on the use of nitrogen-based mineral fertilisers, over the period 2000 to 2010. They find that constraints associated with the EU Nitrates Directive had a greater influence on fertiliser application rates than participation in voluntary agri-environmental measures (AEMs); on average, fertiliser reductions in Nitrate Vulnerable Zones (NVZs) were ten times greater than reductions estimated for land enrolled under 'integrated production', a local AEM. However, this positive finding about the efficacy of NVZs as a nutrient control mechanism contrast somewhat with an earlier study undertaken in England, which showed that "69% of NVZs showed no significant improvement in surface water concentrations even after 15 years…[and]…in comparison to a control catchment 29% of NVZs showed a significant improvement but 31% showed a significant worsening" (Worrall, Spencer and Burt, 2009, p. 21[19]).

In the United States, discharges from certain kinds of concentrated animal feeding operations ("CAFOs", e.g. feedlots, poultry and hog operations) are regulated as point-sources under the National Pollutant Discharge Elimination System under the *Clean Water Act* (CWA). These regulations have changed over time since the initial passage of the CWA in 1972. In relation to a revision of these rules which took effect in 2003, the US Environmental Protection Agency estimated that the revised rules would reduce CAFO nutrient loadings by 23%, sediment loadings by 6%, and metals discharges by 5% compared to pre-regulation baseline loadings (Copeland, 2010[20]). However, in an assessment conducted in 2018 of changes in CWA regulations as applied to US dairy CAFOs, Yu, Du and Phaneuf (2018[21]) find no significant change in management practices after regulatory changes in 2003 and 2008, and conclude that the impact of these regulations on dairy CAFO farm management is limited. They attribute this lack of change to weak enforcement and oversight from the US Environmental Protection Agency, and emphasise that a lack of comprehensive data on dairy CAFOs caused difficulties in enforcement. Given their analysis focussed only on dairy CAFOs, their findings do not preclude the possibility that the regulations stimulated change for other types of agricultural operators, or that environmental improvements occurred via other avenues than changes to dairy farm management practices.

Land clearing and land grazing regulations

Regulations aiming to directly limit the use of land for agricultural purposes—such as regulations on livestock stocking densities, livestock exclusion and limiting land clearing for agricultural (or other) use—are an important part of the agri-environmental policy framework in some OECD countries such as Australia, Canada, the United States, and Israel. Tal (2009[22]) reviews stocking density regulations in these countries (as well as some non-OECD countries) and finds that the environmental effectiveness of such regulations vary considerably across jurisdictions. The author identifies that implementation issues—particularly issues relating to regulatory capture and adequate monitoring and enforcement over sometimes areas—can cause stocking density regulations to be too lenient to achieve substantial environmental benefits.

3.1.2. Hybrid economic instruments: cross compliance mechanisms

Since 2005, the CAP has used "cross compliance" as a mechanism designed to produce a minimum level of environmental sustainability for farms receiving payments under Pillar I of the CAP (direct payments). Cross compliance "links most CAP payments to compliance with rules relating to the environment, public, animal and plant health, and animal welfare and to maintain agricultural land (especially when it is no longer used for production purposes) in good agricultural and environmental condition" (European Court of Auditors, 2008, p. 8[23]). Further, "greening measures" apply a higher level of requirements to a smaller subset of payments.

In 2016, the European Court of Auditors reviewed the cross compliance mechanism to evaluate its effectiveness, and found that "the information available did not allow the [European] Commission to assess adequately the effectiveness of cross-compliance" (European Court of Auditors, 2016, p. 7[24]). Similarly, Pe'er et al. (2017, p. 3[25])found that:

> *There are local and regional successes of targeted CAP instruments (primarily agri-environment-climate measures, AECM), but they fail to scale up to the EU level and the CAP as a whole. Main inhibitors are limited budget, low uptake, and poor design and implementation of AECM. Greening design and implementation is insufficient to reverse negative trends due to broad exemptions, low requirements for crop diversification, lack of management criteria and the inclusion of ineffective options for Ecological Focus Areas (EFA), comprising 75% of EFA area. Climate measures are insufficient, hardly targeting livestock production and nitrogen fertilizer use as the main sources of greenhouse gas (GHG) emissions. Effects on soil and water are partly positive, partly negative."*

While the latter study extends beyond assessment of the cross-compliance and greening mechanisms to include an assessment of the voluntary payments (Agri-environment-climate measures), these reports generally find that cross compliance has been ineffective at substantially improving the environmental performance of agriculture.

In addition to these *ex post* assessments, the impacts of cross compliance requirements have been studied via scenario modelling which assess a variety of CAP reform scenarios, *with* and *without* cross-compliance requirements (see, for example, Schmid, Sinabell and Hofreither (2007[26]); Galko and Jayet, (2011[27]); Brady et al. (2009[28]); Pelikan, Britz and Hertel (2015[29]); Cimino, Henke and Vann (2015[30]); Gocht et al. (2016[31]); Cortignani and Dono (2019[32]); Solazzo et al. (2016[33]); and Gocht et al. (2017[34])). The results of these analyses are discussed in detail in OECD (2019[35])[13] but the key findings obtained from comparing scenarios which entail decoupling of agricultural support *without* accompanying mandatory constraints versus decoupling *with* mandatory constraints (cross-compliance, greening) are:

- Decoupling *without* constraints shows generally a greater positive impact on water quality variables than decoupling *with* constraints, generally because the constraints modelled prevent

[13] I.e. the companion literature review for this report. That review considers the impacts of agricultural policies on productivity and sustainability performance in agriculture.

conversion of agricultural land to alternative uses such as forestry which generally have lower water quality pressures than agricultural land uses.

- Results are more mixed in terms of greenhouse gas emissions, both compared to baseline and when comparing alternative decoupling scenarios.
- Decoupling without constraints had a worse effect on biodiversity than decoupling with constraints, but both alternatives are worse than the baseline scenario, which includes (some) coupled support. This effect is primarily due to landuse homogenisation (towards extensive grasslands) and land abandonment.
- In relation to specific greening requirements:
    - Introduction of ecological focus area (EFA) requirements produces small improvements for greenhouse gas and water quality indicators for the European Union as a whole; however there is strong spatial heterogeneity of impacts; environmental gains were mostly in intensive agricultural areas, due to increased idle land in these regions. However, EFA requirements also stimulated changes in crop prices which in turn caused intensification in more marginal areas, with consequential increases in water quality pressure from agriculture (nutrient balances).
    - Crop diversification requirements were found to have almost no environmental impact.
    - The grassland measure, because it promotes homogenisation of land types, tends to have a negative impact on biodiversity; however it reduces water quality pressures and soil erosion.

To conclude, these analyses generally show that existing mandatory constraints (particularly the EU GAEC condition) does not clearly produce improved environmental outcomes, but rather are often ineffective.

Another example of a hybrid instrument is the "Conservation Compliance" provisions which link eligibility for some US Farm Bill support payments to erosion control on highly erodible land (HEL). Claassen et al. (2004[36]) assesses the performance of Conservation Compliance finds that Conservation Compliance accounts for approximately 25% of the decreases in soil erosion achieved over the period 1982 to 1997. Further, the authors found that this mechanism has likely deterred conversion of non-cropped highly erodible land and wetland to cropland, which further contributes toward erosion control but also likely provides other benefits such as habitat retention and nutrient cycling via wetlands. However, Arbuckle (2013[37]) notes that Conservation Compliance has been the subject of ongoing critiques, including that the requirements do not do enough to prevent erosion, do not address other environmental issues, and that there is inconsistent and insufficient monitoring of compliance. One key critique has been that the ability of this hybrid mechanism to deliver environmental benefits is fundamentally linked to the size and scope (i.e. land coverage) of the US Farm Bill payments they are linked to (Arbuckle, 2013[37]; Claassen, 2012[38]). In recent years, payments linked to conservation requirements have been declining, and if they are further reduced or even eliminated this would reduce compliance incentives on many farms. Prior to 2014, subsidised crop insurance was not linked to conservation requirements. The decisions in the 2014 and 2018 Farm Bills to make subsidised crop insurance subject to cross compliance requirements has strengthened conservation incentives; according to Claassen et al. (2017, pp. 38-39[39]), "where crop insurance premium subsidies are high…compliance incentives are higher under the 2014 Act…If the link between crop insurance premium subsidies and Compliance were severed, the Compliance incentives would decline on many farms."

3.1.3. Economic instruments: environmental taxes and tariffs

Environmental taxes on fertilisers and pesticides have been used in a few OECD countries. According to Hardelin and Lankoski (2018[5]), within the European Union, pesticide taxes have existed in Denmark, France, Norway, Sweden and Finland; and fertiliser taxes have been implemented by Sweden but are no longer in place. The environmental impacts of these input use taxes have been relatively limited, due to a combination of a low tax level set by regulators, combined with a low price-elasticity of demand for pesticides and fertilisers.

Picazo-Tadeo and Reig-Martínez (2007[40]) study the effectiveness of a range of policy options to reduce consumption of nitrogen in citrus farming. The authors find that input taxes (e.g. nitrogen taxes) for (water) pollution control appear to have lower environmental effectiveness compared to nutrient input restrictions, due to being relatively less efficient (cost-effective). In general, very high taxes are needed to induce reductions in fertiliser use to environmentally-sound levels, and they therefore have a higher impact on farm profits than nutrient input restrictions.

Martínez and Albiac (2006[41]) find that emissions taxes appear to be the first-best instrument for addressing nutrient emissions from agriculture, however they may be difficult to implement in practice due to difficulties in tracking non-point source emissions. Turning then to "second-best" options such as input taxes and tariff mechanisms, the authors find that input taxes appear to be more efficient than water pricing mechanisms (taxes on irrigation water) for addressing water pollution issues. This finding is supported by Semaan et al. (2007[42]), who analyse nitrate pollution control policy options for irrigated agriculture in Southern Italy. Further, Martínez and Albiac (2006[41]) find that the relative performance of input standards versus input taxes in achieving environmental objectives may depend on soil type.

Environmental tariffs are a similar policy tool to environmental taxes. They can be used to provide a price signal about the use of natural resources in agricultural production systems. One key resource for which tariffs are often proposed is water. Water tariffs are not often used to manage water scarcity for agriculture, but are used in some cases to recover costs of water irrigation infrastructure (OECD, 2009[43]). Tariffs could in theory also be used to address water quality issues. However, Martínez and Albiac (2006[41]) and Semaan et al. (2007[42]) find that water pricing appears to be an inefficient mechanism for (water) pollution control, and that more direct mechanisms (e.g. nutrient restrictions, input or emissions taxes) are more effective.

3.1.4. Economic instruments: agri-environmental payments and environmental markets

A significant body of work assesses the myriad of voluntary agri-environmental schemes (AES) arising under the European Union Common Agricultural Policy (EU CAP). In a meta-analysis of the literature assessing success in improving biodiversity, Batáry et al. (2015[44]) find that "[r]esearch over the last 20 years shows that European AES have been generally beneficial for farmland biodiversity, leading in the majority of cases to a moderate increase in numbers of species present" (p. 1014). In an empirical assessment of AESs in Germany, Spain, France, Italy and the United Kingdom, Arata & Sckokai (2016[45]) similarly find that "[w]ith the exception of Spain, participation in AESs seems to be effective in promoting more sustainable agricultural practices" (p. 183).[14]

However, Batáry et al (2015[44]) find evidence that the success of AES, in terms of achieving environmental goals, depends on the scheme design being appropriate to the specific region in which it is to be implemented. Specifically, the authors found that the application of Western European-style AES, designed

[14] In Spain, a negative impact on farm income of participating in AES was also found. The authors noted that the Spanish AES consists of mandatory, nation-wide measures, and had a relatively low payment per hectare (compared to other countries studied). Interestingly, Spanish farmers had the highest dependence on total CAP subsidies, but the share of AES payments in total farm subsidies is among the lowest. The authors suggest the low environmental performance of the Spanish AES may be attributable to implementation difficulties and co-financing issues (resulting in payment levels being set "too low", but also that Spain has a higher proportion of crop farmers relative to livestock farmers, and that AES on crop farms appear to be less effective.

for intensively farmed landscapes, were ineffective or even negative for promoting biodiversity when applied in Eastern European contexts in which extensive farming is dominant. In addition, they find that targeted schemes which focus on the needs and spatial distributions of specific species of interest tend to be more effective than untargeted ones, a finding corroborated by others such as Lankoski (2016[46]).

Further, the authors find that in terms of achieving biodiversity goals, AES which promote "out-of-production" practices [15] are much more effective than "in-production schemes [which] support environmentally sensitive approaches to the management of land that is used to grow crops or feed livestock" (p. 1011[44]). They also find that AES have most success in achieving their stated goals when applied to "intensively farmed landscapes of intermediate complexity" (p. 1014[44]), a result supported by Kampmann et al.'s (2012[47]) empirical study of AES in Switzerland, which explicitly assesses scheme performance for different farm landscape types.

Kleijn and Sutherland (2003[48]) undertake a review of four literatures studying the impacts of voluntary agri-environmental policy instruments ("agri-environmental schemes", AES) in Europe: theoretical models on incentives for eco-innovation, econometric studies based on observed data, survey analysis based on stated information and technology case studies. They find that this literature provides very mixed evidence on the environmental effectiveness of policies aimed at improving biodiversity on farmland, and that "the most striking conclusion" from this literature is that there is a lack of robust research on the environmental effectiveness of AES.

Burton and Schwarz (2013[49]) review the theoretical and empirical literature on European AES, distinguishing between AES which are "action-oriented" and those which are "result-oriented".[16] They note (somewhat in contrast to Batáry et al. (2015[44])) that action-oriented AES often have "rather poor" environmental effectiveness, and offer several explanations for this. By prescribing a menu of practices for famers, action-oriented AES restrict the ability for farmers to innovate. This can limit or even preclude the achievement of stated environmental policy goals, particularly in a context where programme incentives are insufficient to achieve the goals using the existing specified practices. Further, such restrictiveness can foster resentment and act as a disincentive to participate, leading to insufficient participation which fundamentally limits the policy's environmental effectiveness.

While acknowledging that result-oriented schemes are still in their infancy with the European Union, Burton and Schwarz (2013[49]) state that "[a]mongst researchers in this field there is a general belief that result-oriented approaches will be able to deliver better ecological outcomes than action-oriented approaches" (p. 631). The key explanation offered is that result-oriented schemes will allow farmers to innovate using "context specific, heterogeneous and subtle" knowledge to improve their environmental service provision—a mechanism which echoes Batáry et al.'s (2015[44]) emphasis on the need to appropriately target policies to specific local conditions. Also, under a result-oriented scheme, farmers will be incentivised to maximise environmental results rather than simply enrolling the minimum of marginal land needed to meet the requirements of an action-oriented scheme, regardless of the environmental results produced.

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[15] I.e. practices on land not actively used to produce crops, forage, etc., e.g. wildflower buffer strips.

[16] The authors define "action-oriented" schemes as "paying farmers not for the provision of outcomes, but the delivery of land management practices", and results-oriented schemes as "agri-environmental schemes that pay land managers, not for performing specific management actions (such as mowing on set dates or restricting fertiliser use to set limits), but for achieving set environmental outcomes (such as particular species mixes or the promotion of an endangered species)" (pp. 629-630). Examples of results-based schemes for biodiversity from the EU context are available from the European Commission (see http://ec.europa.eu/environment/nature/rbaps/index_en.htm, accessed February 2019).

However, the authors acknowledge that existing prototype result-oriented schemes feature high administrative and transaction costs, and are often very small. They also point out that over time, result-oriented schemes could be used to foster collaboration between farmers and environmental groups, perhaps leading to collective or co-operative mechanisms for providing environmental services. Therefore, the dynamic potential of these schemes to achieve specified environmental goals may be greater than the initial potential.[17]

The lack of robust analysis of agri-environmental measures (Kleijn and Sutherland, 2003[48]; Sauer, Walsh and Zilberman, 2012[50]) may in part reflect the difficulty of establishing a counter-factual against which to evaluate policy impacts. Also, in some cases agri-environmental payment mechanisms are relatively new, and therefore more time will be needed before *ex post* analyses can be conducted. Therefore, it is likely that in future such robust evaluations, particularly *ex post* evaluations, may become more common. One recent study by Cisilino et al. (2018[51]) uses a difference-in-difference matching approach to evaluate the environmental effectiveness of organic farming payments under the EU CAP Rural Development Programme in the Marche region of central Italy. The authors found that farmers receiving organic payments used less nutrients per hectare compared to the reference group, and that biodiversity, as measured by a diversification index, improved. However, expenditure on and quantity of agri-chemicals was not significantly different, which the authors attribute to the fact that inputs allowed under organic rules[18] might be more expensive, and the required interventions more frequent.

Beyond the European context, there is also mixed evidence on the success of AES in achieving improved environmental performance in agriculture. One positive result is from south-eastern Australia, where Lindemayer et al. (2012, p. 25[52]) found that "agri-environment scheme investments such as fencing to reduce degrading processes like overgrazing by domestic livestock can have a positive effect on some key vegetation attributes. Such changes in vegetation structure can, in turn, influence bird responses, including those of a range of species of conservation concern." In contrast, Michael et al. (2014[53]), also studying an AES in south-eastern Australia, found that, due to strong habitat specificity, management interventions incentivised by the AES may result in increased populations of already-common lizards but not significantly increase herpetofaunal (reptiles and amphibians) diversity, at least in the short term.

Börner et al. (2017[16]) undertake a broad review of payments for environmental service (PES) initiatives, which includes AESs where the government is the 'buyer' as well as other market-based mechanisms (see also discussion on environmental market mechanisms below). This review covers PES involving agriculture as well as other sectors such as forestry. The authors find that, as of 2017, empirical evaluation of PES is still in its early stages. However, some key findings emerge from their review:

- Programme design and performance often differ between user-financed and government-led initiatives: user-financed programmes more often adopt targeting criteria and strong conditionality rules, and preliminary evidence is that they have higher environmental effectiveness than government-led programmes.
- Many programmes are established on the basis of a "shaky scientific background", which undermines effectiveness.

[17] See also OECD (2013[162]), which provides a systematic review of the conditions for collective action in agri-environmental contexts, including a discussion of the benefits of such mechanisms and challenges to successful implementation.

[18] The authors do not describe what restrictions are placed on input use (including agri-chemicals) organic farmers under organic rules, but for further information see Council Regulation (EC) No 834/2007 of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092/91 https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32007R0834&from=EN, accessed July 2019.

- Environmental effectiveness is associated with the involvement of trustworthy intermediaries, sufficiently long contracts, social co-benefits and voluntary participation.
- PES initiatives may involve tradeoffs, between different environmental objectives or between environmental and other objectives (e.g. social welfare or equity aspects); however, further research is required to understand what causes win-lose versus win-win outcomes.

Payments for watershed services

Brouwer, Tesfaye and Pauw (2011[54]) undertook a meta-analysis of *payments for watershed services* (PWS) schemes worldwide. They found that scheme success (in terms of "environmental achievement" and participation) was influenced by a range of key factors, including "the selection of service providers, community participation, the existence and monitoring of quantifiable objectives, and the number of intermediaries between service providers and buyers" (p. 380[54]). Other significant explanators of environmental performance were: contracts to reduce sediment concluded with downstream hydropower improved performance relative to contracting with other types of "buyers"; a higher number of intermediaries was correlated with poorer performance; voluntary programmes had a negative effect relative to mandatory programmes; and payments in cash instead of "in kind" had a positive effect. These importance of these potential issues is corroborated by Connor and Kaczan (2013[55]), who find that the challenges for erosion of environmental flows arising from increased efficiency of water use (facilitated by water markets) remains unsolved in the Australian context.

Börner et al. (2017, p. 367[16]) find that PWS evaluations generally show that PWS initiatives suffer from "low willingness-to-pay among poor service users, state control of environmentally sensitive lands, high transaction costs, or weak institutions and organisational capacity among both service providers (e.g. tenure insecurity) and users (e.g. monitoring and enforcement infrastructure)".

Environmental markets

Water markets have been used in water-scarce countries as a key tool to reallocate (consumptive) water extraction rights between competing users, including agriculture. Bjornlund (2004[56]) examines the role of formal and informal water markets in south-eastern Australia, and finds that both market types assist irrigators to adjust both to intra-and inter-seasonal climatic variability, as well as to economic pressures and policy reform. However, the author also notes several potential pitfalls which (if not adequately managed), could result in water markets having negative impacts on the environment. These include: the potential to activate previously-unused water access rights; the potential for water markets to precipitate changes in the location of extraction which may have (unintended) third party impacts on the environment (or other users) by changing seepage and evaporation losses, dilution flows (important for salinity); and the potential for water use to move towards more efficient users, again with the possibility of (unintended) negative environmental impacts due to reductions in return flows.

Grafton et al. (2011[57]) examine water markets in Australia's Murray-Darling Basin, the western United States, the Limarí Valley in Chile, South Africa and the People's Republic of China. The authors similarly find that water markets provide an effective mechanism for allocating water between competing water users, but that there are several preconditions for markets to be able to meet environmental sustainability objectives (including but not only in relation to agriculture). These pre-conditions are:

- Adequate scientific data to determine hydrological requirements of water-based environmental resources
- Adequate provisions for environmental flows
- Adaptive management of environmental flows, including the capacity to monitor the environment
- Water quality considerations in water planning and markets

- Complementary basin and catchment-level planning.

Another environmental market mechanism, often involving agriculture, is *biodiversity offset markets.*[19] Bull et al. (2013[58]) evaluate implementation to date (circa 2013) and synthesises theoretical and practical challenges which remain unsolved.[20] The authors find that there is "limited quantitative information available on the outcomes of offset projects". Practical problems of how to monitor offset sites and use monitoring data to evaluate environmental performance are compounded by difference in methodology, objectives and scope across different offset sites or between different programmes.

OECD (2016[59]) studied the design and implementation of biodiversity offset mechanisms[21] across a range of sectors, including agriculture. The authors found that despite the proliferation of such mechanisms, evidence on their environmental effectiveness is mixed. They attribute this not to any fundamental flaw in the instrument itself, however, but rather to how programmes have been designed and implemented in practice. Key design and implementation issues identified include: establishing thresholds for what can and cannot be offset; identifying the type of biodiversity covered by the mechanism; ensuring additionality; having robust monitoring, reporting and verification; reducing transactions costs; and ensuring appropriate compliance and enforcement mechanisms. It is worth noting that many of these issues are not limited to biodiversity offset mechanisms, but are key for all types of agri-environmental policy mechanisms.

Vatn et al. (2011[60]) undertake a broad survey of market mechanisms to protect biodiversity, focussing particularly on payments for ecosystems services (PES), including PES procurement auctions and biodiversity offset markets. They similarly find that "little information exists on the effectiveness of PES on biodiversity conservation and sustainable use" (p. viii[60]). They note that procurement auctions are as yet a nascent policy option and that therefore there is very little in the way of evidence about environmental impacts.

Voluntary land retirement instruments

Land retirement instruments operate by directly taking agricultural land out of production, either for a fixed term (e.g. 5 or 10 years) or in perpetuity. Land retirement can contribute to improved environmental performance on the retired land (by reducing pressures from agriculture on that land) and potentially also for adjacent land (e.g. creating buffer zones, creating habitat corridors or agglomerations, etc.). However, land retirement could in theory also lead to negative environmental performance, for example by increasing pressures on remaining agricultural work lands (i.e. intensification) or because the retired land is not

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[19] Vatn et al. (2011, p. ix[60]) define biodiversity offset markets (and a related concept *habitat banking*) market-based instruments of the cap-and-trade variety: "Governments define a development cap, such as a percentage of land declared not available for development, or a conservation objective such as 'no net loss' of biodiversity. In principle, trading TDR/offsets in a market can then achieve the cap/objective at lower cost. Habitat banking opens up the scope for finding trades with even greater differences in opportunity cost by allowing credits to be banked over time. Experiences are limited to a few countries, mostly high and middle income and little empirical evidence is as yet available on the cost-effectiveness as compared to traditional regulation".

[20] The authors discuss the following challenges for biodiversity offset market mechanisms:

- *Theoretical/conceptual*: currency (choosing metrics for measuring biodiversity); how to define requirements for "no net loss"; how to demonstrate equivalence between biodiversity loss and gains; longevity; time lag (how to deal with temporal gaps between development and offset gains); uncertainty; defining critical biodiversity thresholds.
- *Practical*: how to ensure compliance; and how to measure ecological outcomes; how to deal with various kinds of uncertainty.

[21] This study considers a range of biodiversity offset mechanisms, including but not limited to market-like mechanisms.

adequately managed (i.e. the problem of "land abandonment", which can cause environmental issues such as providing a haven for invasive species and pests, increasing wildfire risk, increased erosion in areas where re-vegetation does not naturally occur, etc.).

Land retirement has been a significant part of agri-environmental policy in the United States for many years. A number of studies evaluate the environmental impacts resulting from the US land retirement programmes such as the Conservation Reserve Program (CRP) and the Wetlands Reserve Program (WRP).

In 2007, Giudice and Haroldson (2007[61]) attempted to assess the effects of the CRP on populations of grassland birds, but found limited evidence of improvements. Their study documented the difficulties of isolating effects of the CRP in a situation characterised by complex agricultural systems, large scales, and changes over time. They also noted that, in assessing overall policy impacts, there is a need to account not only for changes on retired land, but also for the effects of Farm Bill commodity support which incentivises conversion of pasture, haylands and small grains to row crops.

In a more recent study, Gleason et al. (2011[62]) evaluates the impacts of both of these policies for the Prairie Pothole Region (PPR) of the United States. The authors find that, together, these programmes have resulted in more than 2 million hectares of wetland and grassland habitats in the PPR, and that "the restoration of wetlands and grasslands under USDA conservation programmes has enhanced the distribution and quality of habitat for many wildlife species" (p. S75[62]). They point out, however, that the environmental benefits on lands which have been drained for many years is limited by the availability of seeds and germination requirements, and also that the benefits for wildlife could be further enhanced if species-specific habitat relationships are better taken into account in future (this latter point is supported by Riffell et al. (2010[63]). Gleason et al. note that the offsite benefits of these programmes has yet to be sufficiently evaluated.

Waddle, Glorioso and Faulkner (2013[64]) assessed the impacts of the US WRP on amphibian populations in the Mississippi Alluvial Valley, and found that 9 of the 11 frog and toad species studied had higher probabilities of occurrence on WRP land compared to agriculture.

3.1.5. Summary of environmental performance impacts

In summary, there appears to be significant "room for improvement" in the effectiveness of hybrid and market-based agri-environmental policies in achieving their stated objectives. Environmental regulations appear to be more successful in achieving measurable improvement in environmental outcomes; however environmental performance can suffer from regulations being set too low (possibly due to vested interest or regulatory capture during the policy setting or enforcement phases), or due to monitoring and compliance schemes which are less-than-fully effective. Moreover, governments have often wary of using command-and-control measures due to possible trade-offs with economic performance; this is discussed in a later section of this paper. Overall, regulations are often seen as "a safety net to prevent environmental damage and guarantee essential ecosystem processes are maintained", particularly in countries such as Australia and New Zealand where a more neo-liberal approach has been taken towards agriculture (Valentine et al., 2007, p. 315; Burton and Paragahawewa, 2011).

While there is considerable heterogeneity across different contexts, in general action-oriented (also called practice-based) measures often perform poorly on effectiveness criteria. There is much optimism, and some empirical evidence, that result-oriented measures, as yet in their infancy, will be much more effective.

Overall environmental effectiveness of voluntary agri-environmental policies is also undeniably linked to which, and how many, farmers choose to participate, in that a lack of participation can fundamentally undermine the policy's effectiveness (Batáry et al. (2015[44]), Burton and Schwarz (2013[49])). There is an extensive literature on the economics and motivation leading to farmer participation in voluntary policies and about adoption of specific “best management practices” promoted by these policies. However, Engel (2016[65]) notes that maximising participation is not equivalent to maximising scheme effectiveness.

3.2. Cost-effectiveness of agri-environmental policies

Cost-effectiveness can be defined as finding the least cost option for meeting a specified objective or outcome. Cost-effectiveness can be assessed at various scales (e.g. farm-level, landscape-level, regional-level, etc.), but should include all relevant costs; i.e. indirect costs and administrative costs (transactions costs) should be included as well as direct costs (see Balana, Vinten and Slee (2011[66]) and OECD (2010[4])).

Studies analysing the cost-effectiveness of agri-environmental policies demonstrate significant heterogeneity across the specific programmes or schemes which implement them. However, in general there is significant agreement in the literature that existing agri-environmental instruments, particularly voluntary schemes, have considerable room for improvement in terms of cost-effectiveness (Shortle et al., 2012[67]; Engel, 2016[65]; Batáry et al., 2015[44]; Lankoski, 2016[46]; Hardelin and Lankoski, 2018[5]; Coderoni and Esposti, 2018[68]; Dal Ferro et al., 2018[69]). Key areas for improvement are discussed in this section. Due to the paucity of literature specifically evaluating the cost-effectiveness of agri-environmental regulations or hybrid mechanisms such as cross-compliance, this section focusses on the rich literature studying the cost-effectiveness of voluntary economic agri-environmental instruments.

Engel (2016[65]) identifies several factors that can make voluntary agri-environmental schemes (AES) more cost-effective. These include: cost targeting (“favouring low cost sites over high cost sites”); benefit targeting (“focus[sing] on ecological priority areas”) or cost-benefit targeting (“selecting (targeting) sites on the bases of benefit and cost considerations”); indexed payments (where a high-quality index is available[22]); exploiting spatially correlated benefits by using spatially coordinated approaches (e.g. an “agglomeration bonus”); and including payments based on group performance [23] when aggregate performance is important (e.g. in provision of water quality).

Kampmann et al. (2012[47]) report on the cost-effectiveness of Ecological Compensation Areas (ECAs) in the Swiss Alps region of Europe. They find that the highest cost-effectiveness for this measure is achieved when implemented on “simple” landscape type, and that it is “more efficient to protect biodiversity values than to restore them” (p. 575).

Dal Ferro et al. (2018[69]) use a GIS-based modelling approach to study the impacts of various agri-environmental measures under the EU CAP on crop yields and nitrogen use efficiency for farms in the Veneto region of Italy. Their study showed that agri-environmental measures need to be considered at a site-specific level that includes consideration of pedo-climatic variability. Given that the same practices can yield different environmental results in different cases (e.g. different soil types and farm management contexts), it follows cost-effectiveness of specific agri-environmental measures differ considerably across contexts.[24]

[22] “Only if the index is strongly correlated to opportunity costs is indexing likely to be more cost-effective than other approaches.” Engel (2016, p. 158[65]).

[23] “Note that making payments conditional upon group performance does not necessarily imply that the payment is also paid out to the group as a whole.” (Engel, 2016, p. 153[65]).

[24] The same could be said of costs of implementing agri-environmental measures; however these authors do not study this aspect of cost-effectiveness.

Toderi et al. (2017[70]) note that most AES consist of governments contracting with individual farmers. Analysing nine case studies in the Marche region of Italy, the authors provide evidence of a "scale mismatch" and identify that individualistic approaches hamper environmental effectiveness and therefore reduce cost-effectiveness where landscape-scale environmental improvements are needed. Further, the authors find that 'bottom-up' design processes which involve local stakeholders can result in collaborative, landscape-scale but site-specific agri-environmental initiatives which are more effective than existing programmes (see also Mantino et al. (2018[71])).

Galati et al. (2015[72]) point out that AES payments are generally designed to cover the cost of net income lost (e.g. net costs incurred and income foregone), without accounting for ecosystem benefits. They argue this approach is 'not an effective use of public funds' because it does not incentivise farmers to adopt practices that will benefit the environment. Instead, the authors recommend an 'incentive efficiency' approach which takes into account both ecosystem benefits and (net) loss of income, in both the selection of payment recipients and the calculation of the payment amount.

Batáry et al. (2015, p. 1014[44]) find that, overall, AES are "an expensive way to do conservation". They argue that, as a result, "AES should only be employed in parts of the world, such as Europe, where a high proportion of the unique or declining biodiversity depends directly on farmland or farming activities".

Chabé-Ferret and Subervie (2013[73]) assess the French implementation of the EU AES. The authors find that there is a large potential for adverse selection[25] in these schemes, and that different AES have different levels of additionality.[26] In particular, while all schemes studied showed at least some "positive additional effects", those schemes which impose the strongest requirements, such as those subsidising conversion to organic practices, were having the strongest positive additional effects and the least amount of windfall payments to non-additional activities.

Vergamini, White and Viaggi (2015[74]) compare design of voluntary agri-environmental payments in the European Union, the United States and Australia. They find that adverse selection and information rents are a key source of inefficiency (limitation on cost-effectiveness). Agri-environmental auctions are identified as an innovative policy tool that can address problems stemming from asymmetric information between farmers and the policy-maker, because the auction process reveals information via competitive bidding. In a related paper, Vergamini, Viaggi and Raggi (2017[75]) propose a methodology for optimising design of a differentiated payment mechanism, which delivers improved cost-effectiveness by integrating spatial information and simultaneously considering incentive compatibility (for participants) and cost targeting.

Lankoski (2016[46]) developed a theoretical framework and empirical illustration using data from Finland to analyse the cost-effectiveness of voluntary agri-environmental programmes focussing on biodiversity conservation. This work compared a variety of payment systems: uniform payments; three types of conservation auctions with environmental targeting; uniform payment with environmental targeting; and two types of differentiated payments with environmental targeting. The author found that "cost-effective policy design to address heterogeneous agricultural and environmental conditions requires the combination of differentiated payment level and environmental targeting" (p. 31[46]). Across the payment systems analysed, uniform payments were found to be less efficient than other payment types, and auctions with environmental targeting are the most efficient. Importantly, this work found that cost-

[25] OECD (2008, p.16) explains the concept of *adverse selection* in the context of a voluntary agri-environmental scheme as follows: "If a payment is offered to farmers, those who would incur high costs in the provision of the service may not participate, even though their participation could result in the largest gain to society as a whole."

[26] OECD (2012, p. 11) defines *additionality* as "the extent to which the policy was a necessary condition for obtaining the targeted result." Chabé-Ferret and Subervie (2013[73])then define "windfall effects" as payments (or "windfall" gain) made in respect of actions which are *not* additional.

effectiveness gains achieved via environmental targeting more than offset the increase in policy-related transaction costs, compared to non-targeted payment systems. However, this work also highlighted that the potential cost-effectiveness gains from using auction systems are uncertain, as there is significant potential for farmers (auction participants) to extract information rents.

While the findings of this literature are diverse, one synthesised finding is that there is a range of factors in agri-environmental policy design which reduce cost-effectiveness by introducing transaction costs or inefficiencies. Because transactions costs reduce the overall size of the economic surplus, reforms to agri-environmental policies which reduce (private) transactions costs will therefore, other things equal, help reduce trade-offs between improving environmental performance and economic performance. However, transaction costs are only one component of overall cost-effectiveness, and sometimes 'other things' are not equal: for example, (Lankoski, 2016[46]) showed that improved environmental targeting of biodiversity conservation policies improved efficiency overall despite an increase in policy-related transaction costs. Also, the most efficient (i.e. cost-minimising) distribution of transaction costs may not minimise private transaction costs, which may be a source of trade-offs.

Lack of additionality is generally considered to be a major source of inefficiencies in voluntary agri-environmental instruments (Chabé-Ferret and Subervie, 2013[73]; Thamo and Pannell, 2016[76]; Claassen, Duquette and Smith, 2018[77]). This logic seems obvious, in that funds are spent for environmental conservation activities that would have taken place even without the policy. However, García-Amado et al. (2011[78]) challenge this notion, arguing that paying for non-additional activities might provide other benefits such as strengthening community support for the programme, instilling a conservation ethic, or having an equity value by rewarding good environmental practices. The possibility of such other benefits requires further empirical investigation. Chan et al. (2017, p. 113[79]) also note that "strict additionality requirements may act as a signal that nonmonetary motivations for conservation are not valued by excluding conservation-minded people who may already be engaged in conservation activities for non-monetary reasons", and therefore act as a deterrent to scheme participation.

3.2.1. Cost-effectiveness of results-oriented schemes compared to action-oriented schemes

There is quite a broad literature which theorises that performance-based or results-based[27] agri-environmental policies are more cost-effective than (uniform) practice-based policies. For example, Burton and Schwarz (2013[49]) express concern that existing practice-oriented AES are not sufficiently cost-effective: "Factors such as 'adverse selection' of lower yielding land for entry into environmental programmes and selection of options (from menu-type schemes) for ease of management rather than ecological objectives suggest that currently we may not be getting the best environmental return for our investment" (p. 629[49]).

Burton and Schwarz argue that result-oriented AES (roAES) are more cost-effective than action-oriented AES, largely due to reducing information asymmetries and their associated costs. However, they also identify two key challenges with roAES which have yet to be overcome: i) increasing financial risk for farmers, and ii) the challenge of developing effective indicators of results. Further, they acknowledge that existing pilot or "prototype' roAES have relatively high transaction costs to administer the scheme and monitor results, lack economies of scale and often have learning costs because there is no prior experience

[27] Definitions and nomenclature differ considerably in the literature. Broadly speaking, we differentiate between "practice-oriented" schemes where payment is for specified on-farm practices, versus "results-oriented" schemes which pay, at least in part, not for defined practices for demonstrated performance or results, whether at the output level (e.g. reduced nutrient loadings), field level (e.g. maintaining habitat on-farm), or a broader scale such as catchment or landscape level. Future OECD work will explore the full spectrum of such policies.

to build on. While some of these costs may decline with experience, the authors conclude that the "cost reduction value of result-oriented schemes remains largely theoretical" (p. 632[49]). Matzdorf and Lorenz (2010[80]) empirically analyse a result-oriented agri-environmental scheme in Germany and find that this kind of mechanism has improved cost-effectiveness compared to an action-oriented policy. The authors argue that the mechanism promoted flexibility and innovation, increased intrinsic motivation for conservation, and promoted continuity in terms of continuing to enrol a similar or larger amount of land in the programme. However, the authors also identified potential negative effects of result-oriented mechanisms, including high transaction costs and increased financial risk for participating farmers.

Moxey and White (2014, p. 397[81]) comment that "conventional action-oriented agri-environmental schemes have been widely criticised for a number of reasons including poor targeting, a lack of payment differentiation, short-termism and inadequate monitoring". However, rather than advocating for a wholesale switch towards result-oriented mechanisms, the authors identify (p. 398[81]) that "[t]he benefits of a result-oriented approach are likely to be conditional on other supporting arrangements and—although an empirical matter—it may be that changes to other aspects of scheme design [i.e. other than moving from practice-based to result-based] can deliver easier efficiency gains in some cases."

4. Economic impacts of agri-environmental policy

As the previous sections have shown, environmental performance is not necessarily correlated with economic performance. Given this, a key policy question for many governments is how to design and implement policies that incentivise agricultural management practices which would simultaneously stimulate agricultural productivity growth and sustainable resource use, and whether there would be synergies or trade-offs between productivity and sustainability objectives.

While recognising that there may be room for improving the performance of existing agri-environmental policies in terms of achieving improved environmental performance, this section examines evidence from the literature on the ways in which agri-environmental policies impact agriculture's economic performance. The aim here is to identify factors which affect the relationship between environmental policies and agricultural innovation and productivity.

The OECD's framework *Analysing Policies to Improve Agricultural Productivity Growth, Sustainably* (OECD, 2015[1]) acknowledges the important role of innovation in mediating the impact of policies on both environmental and economic performance (productivity growth). The possibility that policy impacts differ with and without (or before and after) innovation means that the dynamic impact of policies may differ from static impacts and that "win-win" outcomes may only occur over time (i.e. lagged effect). Accordingly, this section considers jointly the available evidence of the impacts of agri-environmental policies on innovation and economic performance of the agriculture sector.

A further important source of dynamic impacts is that policies may induce structural change in the agriculture sector, such as changes in farm size, inputs devoted to specific agricultural commodities, farm employment, farm numbers, etc. Structural change may in turn affect the productivity and overall economic performance of both individual farms and the sector as a whole. Therefore, the final part of this section examines evidence on the structural impacts of agri-environmental policies.

4.1. Impacts of environmental regulations on innovation and productivity: The Porter Hypothesis

4.1.1. The Porter Hypothesis

More than twenty years ago, Professor Michael Porter suggested that pollution was generally associated with a waste of resources, or with lost energy potential: "Pollution is a manifestation of economic waste and involves unnecessary or incomplete utilisation of resources … Reducing pollution is often coincident with improving productivity with which resources are used" (Porter and Van Der Linde, 1995, p. 105[82]). Based on this reasoning, Porter argued that "properly designed environmental regulations can trigger innovation that may partially or more than fully offset the costs of complying with them". This has come to be known as the Porter Hypothesis (PH). In other words, it is possible to reduce pollution emissions and production costs at the same time, resulting in "win-win" situations.

The PH is controversial. First, the evidence initially provided to support it is based on a small number of company case studies, in which firms were able to reduce both their polluting emissions and their production costs. As such, it can hardly be generalised to the entire population of firms. Second, some economists may suggest that, if there are opportunities to reduce costs and inefficiencies, companies should identify them by themselves without the need for government intervention. However, over the last twenty years, many studies have proposed analytical justifications for the PH, and in some cases this justification was in turn used to justify government intervention. It could be that the interests of companies and their managers are not aligned. There might be several reasons for this, such as risk aversion, time-inconsistency, or asymmetric information. Regulations force firms to adopt innovations that are profitable for the firm but not for its managers. As Ambec and Barla (2006[83]) argued, the PH can be valid if a market failure exists in addition to the environmental externality. Examples include knowledge spill-overs or market power. For instance, Simpson and Bradford (1996[84]) investigated the impact of environmental regulation in a model with firms competing on international markets. They show that more stringent environmental regulations commit a domestic firm to an aggressive cost-reducing programme, thereby enjoying a first-mover advantage.

4.1.2. Empirical evidence on the Porter Hypothesis

On the empirical side, Jaffe and Palmer (1997[85]) presented three distinct variants of PH. In their framework, the "weak" version of the hypothesis is that environmental regulation will stimulate certain kinds of environmental innovations, although there is no claim that the direction or rate of this increased innovation is socially beneficial. The "narrow" version of the hypothesis asserts that flexible environmental policy instruments, such as pollution charges or tradable permits, give firms a greater incentive to innovate than do prescriptive regulations such as technology-based standards. Finally, the "strong" version posits that properly designed regulation may induce innovation that more than compensates for the cost of compliance and improves the financial situation of the firm.

Many researchers have tested the different versions of the PH empirically, although most of this literature focuses on sectors other than agriculture.[28] Overall, the empirical literature provides evidence for the weak version but not for the strong one. Ambec et al. (2013[86]) conduct a broad review the theoretical foundations and empirical literature examining the Porter Hypothesis. They find that, on balance, the literature investigating the link between environmental regulation and innovation has concluded that there

[28] Vercammen (2011, p. 3[112]) notes that "[t]he economics of regulation occupies only a minor niche in the agricultural economics literature, so it is not surprising there are comparatively few studies on regulating environmental aspects of agriculture.

is a positive link, although "the strength of the link varies" (p. 10), and it is not clear whether the benefits of innovation (cost reductions, new products or markets, etc.) are sufficient to offset or outweigh the direct costs of regulation. They point out, however, that in general previous studies have not sufficiently taken into account the potential for dynamic impacts—particularly potential time lags before innovation yields cost savings.

Cohen and Tubb (2017[87]) undertook a meta-analysis of 103 publications which estimate the relationship between environmental regulation and productivity or competitiveness, at the firm, regional or country level. While they found considerable heterogeneity of results, overall they found the most likely result is that the relationship is insignificant. However, when disaggregating the data they found that "there appears to be a higher chance of finding a negative finding at the facility, firm or industry level, and a higher chance of a positive finding at the state, regional or country level" (p. 18[87]). Further, they found that "flexible regulations are much more likely to exhibit positive and statistically significant results than command and control regulations...studies that include a lagged regulatory variable are more likely to find positive and significant results" (p. 31[87]). These results support the emerging consensus (Lankoski, 2010[3]) that there is no simple positive (or negative) relationship between environmental regulation and economic performance (whether conceived of in terms of productivity, competitiveness, efficiency or financial profitability), but rather that this relationship is mediated by a number of different factors and needs to be empirically assessed on a case-by-case basis.

Giraud-Héraud et al. (2016[88]) consider the Porter Hypothesis in the context of sustainable food policies. They find that, in addition to conventional explanations for the PH found in the literature (e.g. organisational rigidities and market failures), the role of food policies (e.g. education, nutrition regulation, standards) in assuaging consumer food safety concerns and "suspicion" towards new food products can enhance the private benefits of innovation, thereby having a positive effect on the relationship between food policy (including regulation), innovation and firm performance. This adds an important dimension to the PH debate, which is that regulations (and non-regulatory policy measures) can affect the benefits of innovation and ultimately firm performance not only via acting directly on the firm, but also via their effects on consumer demand.

Lanoie, Patry and Lajeunesse (2008[89]) test the PH for the manufacturing sector in Quebec, Canada. They find that the dynamic impact of environmental regulation on productivity is "less detrimental and even positive" (p. 128[89]). Further, they find that a greater degree of external competition is a "driving force" behind firms responding innovatively to regulation.

Ramanathan et al. (2017[14]) demonstrate using a case study approach that regulations can lead to the adoption of environmental management practices by regulated entities, which can positively impact firm financial performance. The authors stress the importance of regulatory flexibility, which, depending on firms' own resources and capabilities, can lead to innovation and positive private benefits; they posit that inflexible regulations can *only* result in a reactive approach in which firms concentrate on pollution control and do not innovate. In their view, the flexibility of regulations and the ability of a firm to respond in a "dynamic", innovative way are key elements mediating the relationship between environmental regulation and firm performance (Ramanathan, Ramanathan and Bentley, 2018[90]; Ramanathan et al., 2017[14]).

4.1.3. Evidence on the productivity impacts of agri-environmental regulation

A few studies assess the impact of agri-environmental regulation on firm performance, but without explicitly assessing the innovation mechanism proposed by Porter. For example, Metcalfe (2001[91]) investigated the potential for detrimental impacts of increasingly stringent water quality regulation on US hog industry production; he found no significant production impacts in aggregate. However, when disaggregated, results indicate that regulatory costs are significant for small operations (<1000 head) but not for large ones. Park et al. (2000[92]) produced a similar result. Piot-Lepetit and Moing (2007[93]) conduct a productivity analysis for the French pig sector for the period 1996-2001. They find a positive “win-win” effect of environmental regulation and farm productivity.

Van der Vlist, Withagen and Folmer (2007[94]) study the impact of environmental regulations relating to energy use on the technical efficiency of small and medium Dutch horticultural firms, for the period 1991 to 1999. They find that increased stringency increased technical efficiency, and attribute this to the regulation “stimulating efficiency improvements”. It is not clear whether these improvements stemmed from innovations or from other sources.

Ferjani (2011[95]) examined the impact of environmental regulations on Swiss dairy farms during the period 1993-2001. While he also reports “mixed” findings, he is unable to reject the hypothesis that firm productivity can be enhanced by “environmental agreements”. However, the specifics of the environmental regulations (also referred to as “agreements”) are not provided, so it is difficult to assess which particular policy settings this study relates to. Further, while Ferjani motivates the study with reference to the PH, the role of innovation is not explicitly studied.

Sneeringer and Key (2011[96]) explore the effects of size thresholds (where larger firms face higher environmental stringency) in US environmental regulation of the livestock industry. Using regression discontinuity analysis, they find evidence of livestock firms responding to the thresholds by adjusting farm size to be “just below” the threshold. Further, they find that avoidance is stronger for new entrants than for incumbents, which they account for by noting that sunk capital costs of incumbents may limit their ability to adjust to the size thresholds. While the authors do not examine the implications of strategic responses to regulatory size thresholds, insofar as economies of scales exist in this industry (which has been shown elsewhere to be the case (Key and Mcbride, 2007[97])), this behaviour could have a negative effect on livestock industry productivity.

More generally, Hardelin and Lankoski (2018[5]) identify that information necessary to undertake cost-benefit assessments for environmental regulations in agriculture is lacking. This includes information about the costs of compliance, including potential negative impacts on farm productivity, but also information on the potential benefits.

4.1.4. Alternatives to the Porter Hypothesis

Mohr and Saha (2008[98]) point out that the idea that firms may benefit from regulation is not incompatible with the notion that regulations are costly. Beyond the most common explanation reconciling these ideas—that regulations can induce firms to innovate to find lower cost methods of producing the same output (a cost-effectiveness argument)—the authors identify several other mechanisms whereby regulation may positively impact firm performance (in terms of profitability), while also inducing innovation. First, when regulations create the potential for scarcity rents (e.g. in a cap-and-trade system), firms may able to shift costs to consumers and increase profits compared to the unregulated case. Faced with different relative costs under regulation due to the ability to shift costs to consumers, a firm may also find it profitable to innovate where previously it did not. Second, where a negative production externality exists (e.g. in a common pool resource case), regulation addressing the externality may increase profits, and similarly alter the net cost of innovation for firms. Third, where consumers have preferences for sustainable products but can only imperfectly observe the sustainable characteristics of those products (i.e. information asymmetries exist), a firm may benefit from regulations that increase information throughout the supply chain, and again may also find it more worthwhile to innovate. Finally, where “green” technologies display public good characteristics, regulation may allow firms to co-ordinate a shift towards using green technology even where non-adoption is a Nash Equilibrium. All of these mechanisms plausibly explain a positive link between environmental regulation and firm performance (profitability), with innovation as a possible (but not necessary) "side benefit". While these mechanism differ from that envisaged by Porter, and the extent to which any of them actually operate in practice is yet to be empirically established, they are important for policymakers to consider. They are important firstly because, as with Porter's mechanism, they challenge the notion that environmental regulations inherently involve a trade-off between environmental and economic performance. Additionally, they emphasise the importance of distributional impacts of agri-environmental regulations and the role of regulations in sustaining co-ordinated outcomes (either between producers and consumers or simply between competing producers).

4.1.5. Instrument choice: regulatory instruments versus economic instruments

In theory, an economic instrument can be set to meet the same pollution reduction as a regulatory instrument such as an emission standard. [29] Thus, economic instruments can lead to the same environmental performance as regulatory instruments in the short run. However, in the long run, economic instruments can provide incentives to go further as pollution abatement technology improves. As abating pollution becomes cheaper, firms will abate more than they would with the equivalent standard. In contrast, in a regulatory context, firms have no incentive to go beyond the requirements of the standard. Furthermore, in the case of an emissions tax, the tax provides higher incentives to invest in innovation than the standard because the gain from improving the technology is higher: the firms save not only on the cost of reducing pollution but also on the tax paid on emissions. Similarly, firms could purchase fewer permits on the market or even sell their own emission endowments by cutting emissions beyond their own emission rights. This is a seminal result in environmental economics: economic instruments dominate regulatory instruments (standards) for enhancing environmental innovation, and is oft cited in the literature.

However, Kemp and Pontoglio (2011[99]) challenge this view. They find that design features are more important for innovation than instrument choice, particularly the following features: policy stringency; predictability (regulatory certainty), including the credibility of policy commitments to future standards; (appropriate) differentiation with regard to industrial sector or the size of the plant; timing: particularly in relation to the use of phase-in periods; possibilities for monitoring compliance and discovering noncompliance; enforcement (inspection and penalties for non-compliance); and combination with other policy instruments. The authors also critically distinguish between "incremental innovation" ("minor modifications of existing processes or products") and "radical innovation" ("technological discontinuity based on a break with existing competencies and technologies"), and find that "there is more evidence of regulations stimulating radical innovation than of market-based instruments doing so" (pp. 33-34[99]).

Requate (2005[100]) concludes from an extensive survey of the literature that "one can draw the main conclusion that instruments which provide incentives through the price mechanism, by and large, perform better than command and control policies" (p. 193[100]), in incentivising adoption and development of advanced abatement technology. However, Requate also argues that environmental technological progress may "crowd out" other kinds of welfare-enhancing technological progress (in a context of scarce R&D resources); i.e. that resources spent spurring pollution-abating technical innovation may be better spent elsewhere. This implies that even if regulations succeed in spurring innovation which more than offsets regulatory costs (as envisaged by the PH), a broader consideration of the relative benefits of the policy is yet warranted.

[29] Note that a regulatory instrument is not necessarily mandatory for all farms—see the examples of *hybrid instruments* in Section 1.2. It is also possible to mandate that all actors participate in an economic instrument such as an emissions trading scheme. The notion of "regulatory" thus refers more to the nature of the constraint on participating entities' choices rather than to whether participation in the policy instrument is mandatory or voluntary.

4.2. Impacts of voluntary and hybrid agri-environmental policy instruments on innovation and productivity

Mandatory policies targeting the source of negative environmental externalities in agriculture (for example, a tax on nitrate run-off) have often been difficult to implement due to the absence of information and data needed to identify the sources of pollution and to measure their contribution to the pollution generated. Some mandatory measures establish "bright lines" for agriculture (e.g. banning certain toxic chemicals, or specifying certain minimum or maximum requirements). However, efforts to improve agriculture's environmental performance have often focused on changing observable farm management practices, and have generally been either wholly voluntary in nature (e.g. payments for best management practices or for ecosystem services) or are in the form of a hybrid measure which ties a mandatory environmental condition to a different (non-environmental) payment mechanism (i.e. cross compliance mechanisms). This has been the case in part due to the norm of assigning property rights to the environment to landowners and in part due to the difficulties of measuring non-point pollution and attributing responsibility to particular polluters at reasonable cost. The current implicit assignment of property rights to farmers allows them to use land to maximize economic returns (Rabotyagov, Valcu and Kling, 2014[101]). This has implicitly protected the agricultural sector from policies that would penalize pollution; in general, there is no mandatory requirement for farmers to reduce pollution.[30] The absence of a mandatory requirement to reduce pollution implies that farmers are not expected to bear any costs of abatement. Instead they are encouraged to reduce nutrient loadings through technical assistance, moral suasion and subsidies to induce voluntary adoption of best management practices (Stephenson and Shabman, 2017[102]).

In contrast to the literature on the impact of environmental regulations on innovation and economic performance (which is extensive but rarely focuses specifically on the agriculture sector—see Section 4.1 above), there is a much smaller body of literature empirically assessing the economic impacts of *voluntary* agri-environmental policies.[31] According to Sauer, Walsh and Zilberman (2012, pp. 6-7[50]), "only a few studies so far have attempted to empirically measure the actual impact of being subject to agri-environmental schemes on producer behaviour at individual farm level using statistical or econometric tools". This literature generally focusses on the economic impacts of certain types of voluntary measures applied in specific contexts. In particular, several areas of concentrated study stand out: i) economic impacts of "greening" reforms to the EU CAP; ii) economic impacts of voluntary mechanisms which retire agricultural land; iii) economic impacts of measures to improve water use efficiency of irrigated agriculture in relatively water scare areas; and iv) innovation impacts of result-oriented voluntary instruments. The following sections in turn present findings in these areas.

[30] Even where more general requirements to improve environmental performance exist, agriculture may be exempted. For example, in the United States, runoff from agricultural fields is excluded from the need for federal permits under the Clean Water Act (OECD, 2016[163]).

[31] There is extensive analysis on the economic impacts of *agricultural policies* (e.g. production subsidies), and there is also some *ex ante* discussion of the potential economic impacts of agri-environmental instruments. However, outside of the contexts discussed in this paper, there is a relative paucity of studies that attempt to assess the economic impacts of voluntary agri-environmental measures in OECD countries; studies instead seem to focus on assessing performance in terms of environmental impacts or cost effectiveness (see Section 3), or on factors contributing to adoption or participation in voluntary measures. This may reflect an assumption that the impacts on productivity and profitability are minimal.

4.2.1. Economic impacts of voluntary and hybrid instruments in the EU context

The European Union's CAP has undergone considerable reform. Efforts to "green" the CAP have resulted in around 50% of support payments now being conditional on hybrid agri-environmental constraints ("cross-compliance" requirements), and another 10% of support is paid under voluntary agri-environmental schemes (OECD, 2017[103]). Reflecting the incremental nature of these greening efforts, there is a group of studies that assess the economic impacts of various iterations of the CAP, either as a whole (i.e. Pillar I and Pillar II measures) or for a specific part. In such studies, the marginal effect of *environmental* measures is not always explicitly identified (e.g. some studies look at the impacts of Pillar II measures without distinguishing between environmental and other measures). In addition, a range of different economic performance impacts is studied: for example input use, technical efficiency, productivity, and output.

Bokusheva, Kumbhakar and Lehmann (2012[104]) analysed the direct effects of general agricultural subsidies, with and without environmental cross-compliance (ECC) requirements, and also with ecological direct payments for organic farming (DP), on the productivity and output of Swiss cropping and dairy farms. They found that adding ECC requirements to direct subsidies affected dairy and cropping farms differently, increasing the output of dairy farms by around 3%, but decreasing cropping output by around 0.4%. However, the opposite result was found in relation to input productivity. Overall, the economic impact of the addition of ECC requirements was mixed. In relation to direct payments, the authors found "scarce empirical evidence" of any impact on farm production. The authors also found evidence of sub-optimal input use, with Swiss farmers disproportionately using labour and capital relative to land, fertiliser and other materials.

Gocht et al. (2016[31]) studied the economic (and environmental) impacts of CAP greening requirements for the EU-28, using a scenario modelling approach which compared CAP greening to a baseline without these requirements. They found that greening is expected to lead to a small decrease in agricultural production, and a corresponding small rise in prices. Overall, they project that average farm incomes will increase slightly (around 1%), as price effects outweigh production effects.

Mamardashvili and Schmid (2013[105]) studied the technical efficiency[32] of Swiss farmers over the period 2003 to 2009. They found (p. 308[105]) that "ecological direct payments had a positive impact on the technical efficiency of sample farms in all three regions" studied (plain, hill and mountainous), in contrast to their hypothesis that ecological direct payments are higher for farmers engaged in extensive farming activities (who have correspondingly lower technical efficiency). However, the authors note that the empirical literature has reported contrasting results with respect to the relationship between subsidies and farm technical efficiency; although they do not make clear whether these contrasting results relate to agricultural subsidies in general or more specifically to subsidies for improved environmental performance.

Pufahl and Weiss (2009[106]) examine the impacts of agri-environmental (AE) programmes and the "Least Favoured Area" (LFA) scheme on input use and output of German farms, for the period 2000-05. They find that participation in these schemes *increases* farm output (sales) compared to the control group, although this causal impact is weaker for the AE programmes than the LFA scheme. They find that these impacts can be attributed to expansions in area cultivated by participating farmers rather than productivity increases; productivity impacts in terms of sales per hectare were minimal and not significant. Schroeder, Gocht and Britz (2015[107]) similarly find that CAP Pillar II AE programmes support modest extensification of agriculture. To the extent that expansions in cultivated area bring land into production that was not

[32] The authors defined technical efficiency as "the ratio of the observed output to the maximum feasible output in an environment characterised by random shocks" (p. 302[105]).

previously cultivated (a question not explored by the authors), the LFA scheme, and to a lesser extent the AE scheme, could have unintended negative environmental effects.

Manevska-Tasevska, Rabinowicz and Surry (2013[108]) study the impacts of different elements of CAP subsidies on the technical efficiency[33] of Swedish farms for the period 1999-2008. Agri-environmental measures studied were the Pillar II environmental subsidies and set-aside premiums (a range of non-environmental measures was also considered). They found a significant and positive effect of the two agri-environmental measures on the technical efficiency of milk, cattle and pig farms in Sweden.

Salhofer and Streicher (2005, p. 3[109]) argue that the production impacts of agri-environmental policies depend on two things: the specific conditions relating to inputs or production methods and to what extent these constraints are actually binding on production. The authors assess production impacts (measured as effects on grain yields) of ten voluntary agri-environmental programmes in Austria, which in the study year (1997) accounted for 68% of agri-environmental expenditures in Austria. They found that different programmes had different effects: three had significant negative effects;[34] a programme incentivising crop rotation had a significant positive effect; and the rest had non-significant effects. The authors concluded that—since most programmes had no significant negative effects (and one even had a positive effect) on productivity—“serious selection bias” was present, creating substantial windfall effects.

Several studies also identify the potential for important unintended impacts of voluntary schemes. Lobley et al. (2014[110]) contend that practice-based agri-environmental schemes dis-incentivise innovation and can effectively “deskill” farmers via being (overly) prescriptive about farm management practices. Moreover, insofar as farmers provide environmental services out of intrinsic motivations such as stewardship ethics or a wanting to display “prowess” in environmentally sound management, agri-environmental schemes that are too prescriptive may actively prevent such service provision. This implies certain kinds of practice-based schemes could actually *negatively* affect pre-existing relationships between environmental and economic performance. In a similar vein, Rocchi et al. (2017[111]) studied farmers’ motivation frameworks for participating in an AES for managing land adjacent to a conservation area. They found that payment is not necessarily a key factor for some farmers deciding to participate, and that the potential for farmers to have a “conservation orientation” needs to be better taken into account in policy-design.

Vercammen (2011[112]) points out that when farmers are risk averse, conservation payments can induce *increased* use of fertilisers and pesticides via an “insurance effect”. This occurs because the environmental payments (which are decoupled from production) make farmers less risk averse and therefore less focussed on using input reduction as a risk mitigation strategy.

Overall, findings indicate that voluntary and hybrid (cross-compliance) agri-environmental measures in the EU context do not appear to generally have significant negative impacts on participating farms’ productivity. In some cases, production impacts are even positive—for example, participation has in some cases been linked with improved yields or expansion in cultivated area. These positive impacts may potentially be taken as evidence of a “win-win”. However, they may alternatively be evidence of selection bias, a lack of

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[33] The authors measured technical efficiency as follows: “[t]echnical efficiency is estimated relative to the best performing farms included the data sample for each specialisation. The value of the estimated technical efficiency coefficients ranges between 0 and 1, and denotes for farm efficiency between 0% - 100%.” (p. 5[108]).

[34] These programmes were: organic farming; a programme “equal to organic farming in regard to crop production only”; and a programme applying to extensive grain cultivation which included “a limitation to low-yield varieties; non-application of growth regulators or fungicides; crop specific limits of N-fertiliser application; non-application of sewage sludge; maintaining of the grassland area” (p. 9[109]).

additionality and the creation of "windfall gains" for participating farmers—i.e. that farmers are receiving payments for actions they would have undertaken in the absence of the policy, for example because they are intrinsically motivated to produce environmental services or because it is economically profitable to do so (Salhofer and Streicher, 2005[109]; Chabé-Ferret and Subervie, 2013[73]; Baylis et al., 2008[113]; Zezza et al., 2017[114]).

4.2.2. Economic impacts of voluntary land retirement instruments

Agri-environmental instruments which retire agricultural land from production directly alter land use patterns, and therefore can affect agricultural productivity by altering the relative supply (and therefore price) of productive inputs (Lubowski et al., 2006[115]).

Wu and Lin (2010[116]) find that the US Conservation Reserve Program (CRP) has a significant and positive effect on agricultural land values, ranging from 2% to 14% depending on location.

Sullivan et al. examine the broader economic impacts of the CRP for rural communities. In doing so, they identify both positive and negative potential economic impacts on farms: on the one hand, land retirement may increase the productivity of adjacent fields (e.g. by providing a buffer to wind erosion)—the authors cite the estimated "on-site benefits" as USD 122 million per year. On the other hand, similar to Wu and Lin (2010[116]), land retirement increases the value of remaining agricultural land and may make have negative impacts on farmers who rent land or on those who wish to expand their operations. Also, insofar as highly productive land is retired (due to the fact that highly productive land may also be highly susceptible to erosion and thus provide high environmental benefit when retired),[35] the authors find that the CRP can "leav[e] expanding operations and beginning farmers competing for less productive land at rental rates that are higher than would be the case in the program's absence" (2010[116]).

Feng et al. (2005[117]) show that both the economic and environmental impacts of voluntary land retirement programmes (such as the US CRP) can be very different when operating in the presence of "working lands" programmes (i.e. agri-environmental schemes incentivising adoption of conservation activity). Because working-land programmes compete with land retirement programmes, the impacts of land retirement programmes on the area and quality of land retired may differ compared to the case where the land retirement programme operates in isolation. The authors find that "the presence of both a large working land and land retirement program can result in more environmental benefits and income transfers[36] than a land retirement only program can achieve" (p. 1237[117]).

4.2.3. Economic impacts of public instruments for water for the environment

Agriculture accounts for 70% of freshwater use globally and is the major user of freshwater in many OECD countries (OECD, 2018[118]; Gruère, 2016[119]). Irrigated agriculture contributes significantly to water-related environmental issues, including aquifer depletion, loss of habitat and biodiversity due to insufficient environmental flows, and water quality degradation. Among the many policy instruments available to help encourage agriculture to reduce its impact on water resources, governments, particularly in water scarce areas, often make use of instruments to redistribute water allocated to agriculture towards the environment. Two broad instrument types are public purchase of water entitlements from consumptive users

[35] The authors note changes to eligibility rules—particularly in relation to ranking indices and the introduction of soil-specific rental rates—which increase the probability that highly productive land will be enrolled in the CRP.

[36] Defined as "the sum of producer surpluses retained by the farmers whose opportunity costs fall below the subsidies" (p. 1235[117]).

(predominantly irrigated agriculture) for environmental use (also known as "buyback")[37] and publicly funded investment in improving the efficiency irrigation infrastructure (both on-farm and off-farm).

Marchiori, Sayre and Simon (2012[120]) identify that government buyback of water rights from agriculture has been used in a range of water-scarce regions, including the Upper Guadiana basin in Portugal and Spain, the Eastern Snake River Plain Aquifer in Idaho, United States, and the Murray-Darling Basin, Australia, and have been considered in Cyprus,[38] Morocco, and Mexico.

The economic impacts of buyback has perhaps been most-studied in Australia, where buyback has been used most extensively in Australia's Murray-Darling Basin (MDB), and has caused significant debate due to concerns about cost-effectiveness on the one hand, and potential negative impacts on the irrigated agriculture sector and rural communities more broadly on the other. Using farm level survey data, Wheeler, Zuo and Bjornlund (2014[121]) find "weak to no significant evidence from the regression modelling to suggest that there is a delayed negative impact on net farm income from selling water entitlements [i.e. participating in buybacks], which supports the notion that the reduction in farm production has been offset by many irrigators using water sales proceeds to reduce debt (and hence interest payments), restructure and reinvest on farm" (p. 72[121]). However, the authors note the need for ongoing incremental adaptation by irrigators to meet future water scarcity in the context of increased climate variability, particularly by irrigators who have sold entitlements. Dixon, Rimmer and Wittwer (2011[122]) model the impacts of buyback in the southern MDB using TERM-H2O, a dynamic multiregional computable general equilibrium model containing water accounts. They similarly find that buyback is likely to have little impact on aggregate farm output, and that resulting increases in water rights prices would be offset by decreases in irrigable land prices. However, model results indicate farm resources would be substantially reallocated, both to dryland agriculture but also to high-value irrigated crops such as horticulture, and that impacts in individual regions differ.

Investing in upgrading irrigation infrastructure is also an oft-used policy, sometimes in conjunction with buyback but also separately.

Irrigated agriculture in southern Spain has undergone a period of modernisation under the National Irrigation Plan (Plan Nacional de Regadíos, PNR) in which a significant proportion of upgrade costs are borne by government, generally 50% (Barbero, 2006[123]).[39] Tarjuelo et al. (2015[124]) found that this modernisation "increased the productivity [of irrigated agriculture] per unit of land and water...However, this did not lead to benefits [e.g. improved profitability] for farmers in all cases because investment and

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[37] Similar "buyback" mechanisms are more commonly used in the fisheries sector.

[38] *Note by Turkey*: The information in this document with reference to "Cyprus" relates to the southern part of the Island. There is no single authority representing both Turkish and Greek Cypriot people on the Island. Turkey recognises the Turkish Republic of Northern Cyprus (TRNC). Until a lasting and equitable solution is found within the context of the United Nations, Turkey shall preserve its position concerning the "Cyprus issue".

Note by all the European Union Member States of the OECD and the European Union: The Republic of Cyprus is recognised by all members of the United Nations with the exception of Turkey. The information in this document relates to the area under the effective control of the Government of the Republic of Cyprus.

[39] The PNR is not solely an agri-environmental measure, as it has multiple objectives, as follows:

- saving water and rationalising water management in irrigation bodies
- contributing to consolidate the national agro-food system under the framework of the CAP and the market evolution
- improving social and economic status of farmers
- contributing to the territorial balance, maintaining population in rural zones
- controlling the inputs use, reducing diffuse pollution and water consumption, and
- incorporating other environmental aspects into the management of irrigation bodies.

energy costs have increased." Rodríguez-Díaz et al. (2011[125]) found that modernisation in Andalusian irrigation districts incentivised structural change and changes in productivity, shifting agricultural production away from cotton and towards permanent plantings such as citrus. The authors found that "[i]n general, farmers tend to move to more profitable crops, trying to offset the higher costs of the new system with an increase in farm income" (pp. 1002-1003[125]).

Roobavannan et al. (2017[126]) studied a particular catchment within the MDB (the Murrumbidgee catchment), and found that reduced water allocations to the agriculture sector (as a result of both buybacks and partial transfer to the environment of water "saved" during infrastructure upgrades) were associated with reduced aggregate agricultural output and employment. However, more broadly, unemployment declined and median household income increased in the study area. The authors provided the following reasons for these seemingly paradoxical results: firstly, out-migration of people from study area, and secondly the absorption of the labour force by faster-growing non-agricultural sectors.

Water markets are a multi-purpose policy instrument (i.e. not strictly an agri-environmental policy instrument) that allow for re-allocation of water between competing users (including potentially between consumptive and non-consumptive users). This can serve both environmental and economic objectives. In terms of economic impacts, Qureshi et al. (2009[127]) argues that water markets provide incentive for irrigators to adapt and improve their productivity. Evidence from areas where water markets are well-established, particularly the southern connected Murray-Darling Basin in Australia, shows that water markets play "a critical role in maintaining irrigation sector incomes during drought, with likely adaptation advantages under mild to moderate future climate change scenarios" (Wheeler et al., 2014[128]). Estimates of the marginal economic value to agriculture of water markets are substantial (Grafton et al., 2011[57]). However, Wheeler et al. (2014[128]) note that the adaptation capacity of agricultural irrigators depends not only on access to markets but on other factors, particularly crop type (with annual crops allowing significant more flexibility than perennial crops).

4.2.4. Innovation impacts of result-oriented agri-environmental mechanisms

Many authors argue that practice-based voluntary measures stymie innovation because they simply ask farmers to choose from a pre-selected menu of acceptable "best management practices" (BMPs) (Winsten et al., 2011[129]). An improved ability to stimulate innovation and adaptation of management practices to local conditions is likewise identified as one of the key reasons for preferring result-oriented mechanisms (Matzdorf and Lorenz, 2010[80]; Moxey and White, 2014[81]; Burton and Schwarz, 2013[49]; Zabel and Roe, 2009[130]). However, empirical evidence on the degree to which result-oriented mechanisms indeed spur innovation is scant, not least because result-oriented mechanisms are still in their infancy.

Winsten and Hunter (2011[131]) and Winrock International (2010[132]) document the results from the Performance-based Environmental Policies for Agriculture (PEPA) Initiative, which pilot-tests result-oriented mechanisms for reducing agricultural nonpoint source (NPS) pollution in Iowa and Vermont, United States. Winrock International found that the pilot successfully "stimulated new, innovative management practices" (p. 5[132]), and that results differed across states. In Iowa, farmers planted winter cover crops, eliminating the need to apply chemicals to prepare fields in the spring, as well as reducing the need for spring field traffic to apply chemicals; both of which reduced NPS pollution. In Vermont, farmers "experimented with manure injection as a way to reduce P loss on corn and hay fields". The authors found that "[p]articipants, especially grass-based dairy farmers, are excited about the potential for manure injection to reduce P runoff, while at the same time increasing yields" (p. 5[132]). Winsten and Hunter (2011[131]) documented how the pilot initiative induced changes in fertiliser application regimes, achieving a small (in terms of cost savings) "win-win" outcome.

Russi et al. (2016[133]) provide one of the few empirical examinations of a well-established (i.e. not in pilot phase) result-oriented voluntary mechanism: they studied the MEKA-B4 (a result-oriented agri-environment measure in place in Baden-Württemberg, Germany) over the period 2000 to 2014. This objective of this measure is to preserve species-rich grassland. The authors identified that because the objective concerns preservation, the actions needed are not necessarily innovative ones, but rather the "maintenance of traditional management strategies"; conversion of existing intensively-managed pasture is not necessarily the goal of this instrument, and moreover "would be difficult and take a long time". Nevertheless, the authors found that this mechanism "does promote some minor degree of innovation, because farmers are encouraged to fine-tune their management strategies to optimise the conservation of species-richness, which before was more of a by-product of extensive farming than a specific goal (for the fine-tuning of management strategies" (p. 73[133]). Thus, while the innovation potential of result-oriented policies "seems promising" (Zabel and Roe, 2009, p. 126[130]), it appears to be contingent on setting objectives that actually require or allow for an innovative approach.

Using results from an analysis of a Swedish result-oriented mechanism for carnivore conservation in livestock communities, Zabel and Holm-Müller (2008[134]) show that use of result-oriented voluntary mechanisms that are administered via collectives can spur innovation in terms of the payment mechanism design, allowing it to be adapted to local conditions.[40] This work suggests that when analysing the impact of policies on innovation, it is not only innovation on-farm that may be affected.

4.3. Dynamic impacts: Agri-environmental policy and structural change

Structural change in agriculture is a complex phenomenon involving multiple and interlinked driving factors, including agri-environmental policies. The following discussion concentrates in particular three dimensions of structural change: the entry or exit of farms, a change in farm size, and modification of land use.

Isik (2004[135]) studied the spatial impacts of Clean Water Act environmental regulation on the US dairy industry and found that state-level differences in regulatory stringency may have contributed to the relocation of dairy farms. However, Metcalfe (2001[91]) and Park et al. (2000[92]), performing similar analyses for the US hog industry and the US livestock sector respectively, found that while regulations were costly and were different across state lines, there was little to no evidence that differences in state regulations prompted farm migration.[41] These apparently conflicting results point to the importance of other factors—these authors identify sunk infrastructure, agglomeration externalities, established marketing and distribution channels—as being more important determinants of firm location.

Park et al. (2000[92]) also note the potential for endogeneity of environmental policy and the size of the sector; that is, the industry's structural characteristics may affect the level of regulatory stringency rather than (or as well as) vice versa.[42] Metcalfe's (2000) results also support this.

[40] In this case, group payments were made at the level of the village, and each village then decided collectively on how to distribute payments to individuals.

[41] Metcalfe (2000) did find evidence that environmental stringency affected production decisions for small farms (<1000 head), but did not comment on whether this affected small farm location decisions. No impact was found for large operations. Metcalfe notes this latter result (no impact) is consistent with a number of previous studies.

[42] This could occur via various mechanisms: for example, structural characteristics may be correlated with environmental damages which motivate imposition of regulation and determine regulatory stringency; structural characteristics may be correlated with the industry's ability to influence regulatory decision-making (e.g. via political processes).

Ahearn, Yee and Korb (2005[136]) studied the determinants of structural change in the US agriculture sector for the period 1982 to 1997. They found that conservation payments under the US CRP "increased the share of small farms and decreased the share of large farms, while decreasing exits from agriculture" (p. 1188[136]).

A number of studies investigate the structural impacts of cross-compliance measures and agri-environmental schemes under the CAP. Pufahl and Weiss (2009, p. 8[106]) found that, for German farmers, participation in an agri-environmental scheme resulted in an expansion of area under cultivation, and further that participation "significantly reduced the purchase of farm chemicals (fertiliser, pesticide)". In contrast, Bokusheva, Kumbhakar and Lehmann (2012, p. 97[104]) found that EU requirements to establish ecological compensation areas "reduced farms' productive acreage by 7%".

Sahrbacher, Hristov and Brady (2017[137]) studied the impacts of equalisation of Single Payment System (SPS) payments[43] and the subsequent introduction of "greening" requirements in the form of Ecological Focus Areas (EFA) for farms in Scania, Sweden and Saxony, Germany). They found evidence of induced structural change in both study regions due to the SPS equalisation: farm size increased substantially and the number of farms decreased in Scania where the policy change was larger; similar but smaller impacts also occurred in Saxony. However, the subsequent reform introducing EFA requirements had little structural impact in either region—the authors concluded that EFA measures did not affect farm, largely because the previous SPS equalisation reform already resulted in the reallocation of a large amount of land to fallow land, which is automatically eligible for meeting EFA requirements. This study points out some flaws in the EFA mechanism design: first, the authors find that "farms can continue to use their most productive land in production, while offsetting the potential costs from EFA restrictions through acquiring marginal land (which is made available through structural change)" (p. 128[137]); second, the greening measures induced an increase in leguminous crops and an expansion in livestock production, rather than enhancing biodiversity as intended. This study also shows the importance of the existing institutional context in determining the ultimate impact of an incremental policy change: had the preceding SPS equalisation reform not incentivised a large shift towards fallow land, the impact of EFA measures may have been quite different.

Brady, Ekman and Rabinowicz (2011[138]) also examined the impacts of the EU's SPS (with "good agricultural and environmental condition" (GAEC) requirements) on structural change and other outcomes. They found that structural change decelerated as payments became more decoupled, because "minimal land management" to keep land in GAEC provides a new income opportunity. This finding is supported by Nordin (2014[139]), who similarly found that grassland support payments in Sweden lowered the pace of structural change in grassland regions. Czyzewski, Bazyli; Smedzik-Ambrozy (2017, p. 14[140]) similarly find that "CAP subsidies in the 2007-2012 financial [period] led to the petrification of the productive structures in the EU agriculture, to some extent preventing them from evolving in the pro-environmental direction".

Olper et al. (2014[141]) study the impacts of CAP payments on out-farm migration over the period 1990 to 2009. They find overall that CAP subsidies reduce out-farm migration (although the effect is not very high), but that the effect of Pillar I payments is more effective for reducing out-farm migration than that of Pillar II payments. However, decomposing the analysis to look at specific types of Pillar II payments, the authors find that agri-environmental payments have a significant *positive* effect on out-farm migration, in contrast to LFA payments. This result is unexpected, in that economic theory suggests that the effect on out-migration should be negative (i.e. keep labour on-farm) insofar as agri-environmental payments incentivise more labour-intensive activities such as organic farming (Petrick and Zier, 2012[142]). One potential

[43] See Sahrbacher, Hristov and Brady (2017, p. 118[137]) for a description of the equalisation reforms.

explanation could be that Pillar II payments were made in respect of maintenance of grassland or fallow land, rather than conversion to relatively labour-intensive organic farming; however this possibility is not explored by Olper et al. (2014[141]).

Overall, synthesised findings from the literature analysing the structural impacts of agri-environmental policies show that these policies can tend to decelerate the pace of structural change by allowing land retirement, fallow or low-management land uses to become a (more) profitable land use option for farmers. However, other types of agri-environmental schemes can also incentivise *expansion* of cultivated area. Thus, impacts are likely to be stronger in terms of land use change than total farm numbers or entry and exit decisions. In terms of impacts on farm size, there are indications that structural impacts differ with size; impacts are most likely to be positive for small farms but the effects for medium sized and large farms are less clear.

5. Summary of empirical findings from the literature review

This section brings together the key empirical findings from the literature studied in Sections 3 and 4, in order to compare findings on environmental and economic impacts side-by-side, by instrument type. While this necessarily omits some detail, it provides a useful quick reference for policy-makers.

5.1. Regulatory instruments

- Common forms of environmental regulations in agriculture are nutrient regulations (e.g. fertiliser restrictions, nutrient loadings limits), maximum stocking density regulations and land clearing regulations. Typical examples include, in the European Union: the European Water Framework Directive, the Nitrate Directive, and the Birds and Habitat Directives. In the United States, the US EPA regulates production practices of Concentrated Animal Feeding Operation (CAFO) under provisions of the 1972 *Clean Water Act*.
- There is considerable variation and uncertainty about the costs and benefits of environmental regulations targeting agriculture (Hardelin and Lankoski, 2018[5]).

Sustainability impacts

- Input and technology standards have in some cases been found to be highly effective in achieving specified environmental goals, but they need to be set at a level to achieve the goal(s) and require effective enforcement mechanisms.
- However, input and technology standards can be inefficient in that they achieve goals at high cost due to preventing flexibility.
- Standards have been demonstrated to be a better policy option in contexts where environmental risks are higher (e.g. highly vulnerable soils (Martínez and Albiac, 2006[41])).

Economic impacts

- There is little evidence in the literature studied of significant negative economic impacts of environmental regulations on the agriculture sector in aggregate, with some studies even finding a positive impact. However, costs of complying with environmental regulations appear to matter more for smaller farms, and also depend on factors such as the availability of alternatives (e.g. in the context of a ban on specific inputs) and the ability to pass on costs to consumers. (Metcalfe, 2001[91]; Piot-Lepetit and Moing, 2007[93]; Garcia-German, Bardaji and Garrido, 2014[143]; Park et al., 2000[92])

- Inefficient use of inputs, particularly fertilisers, has been found to be a key source of agricultural pollution. In this context, environmental regulation can create a “win-win” outcome by simultaneously inducing improvements in input use efficiency and reducing pollution. (Piot-Lepetit and Moing, 2007[93])
- There is some evidence that increased stringency of environmental regulations improves farm technical efficiency (van der Vlist, Withagen and Folmer, 2007[94]).
- Dynamic impacts of environmental regulations which decrease crop yields (e.g. pesticide bans) have been found likely to be less negative than initial impacts as market feedback mechanisms mitigate direct crop revenue impacts (Schneider, Rasche and McCarl, 2018[144]).

5.2. Hybrid instruments

- Key examples of hybrid instruments are ‘cross compliance’ and ‘greening’ requirements under the EU CAP and conservation requirements under the US Farm Bill (which primarily relate to soil conservation on highly erodible land).

Sustainability impacts

- Cross compliance requirements under the EU CAP have generally been ineffective at substantially improving the environmental performance of agriculture (*ex post* assessments include European Court of Auditors (2016[24]), Pe’er et al. (2017[25]) and Coderoni and Esposti (2018[68]); *ex ante* assessments include Schmid, Sinabell and Hofreither (2007[26]) Galko and Jayet, (2011[27]), Brady et al. (2009[28]), Pelikan, Britz and Hertel (2015[29]), Cimino, Henke and Vann (2015[30]), Gocht et al. (2016[31]), Cortignani and Dono (2019[32]), Solazzo et al. (2016[33]) and Gocht et al. (2017[34])).
- Conservation compliance requirements for the US Farm Bill appear to have contributed to reduced erosion on highly erodible land, and may have forestalled conversion of wetlands for crop production (Claassen et al., 2004[36]).

Economic impacts

- Cross-compliance requirements under the EU CAP appear to impact different farm types differently, but in general studies on the economic impacts show that impacts appear to be neutral or small in magnitude (Bokusheva, Kumbhakar and Lehmann, 2012[104]; Gocht et al., 2016[31]; Cortignani and Dono, 2019[32]).

5.3. Voluntary agri-environmental schemes for working agricultural lands

- There is a wide variety of agri-environmental schemes (AES) across OECD countries.

Sustainability impacts

- Environmental effectiveness of AES across OECD countries varies widely (Batáry et al., 2011[145]; Kleijn and Sutherland, 2003[48]).
- Significant efforts have been devoted to evaluating the environmental effectiveness of AES. These studies show that AES using an “action-oriented” or “pay-for-practice” approach have had mixed success in improving environmental outcomes (Batáry et al., 2011[145]; Kleijn and Sutherland, 2003[48]; Kleijn et al., 2006[146]; Uthes and Matzdorf, 2013[147]; Arata and Sckokai, 2016[45]), and in general have been found to lack cost-effectiveness (OECD, 2005[148]; Moxey and White, 2014[81]).
- There is some early evidence that using “result-oriented” or “pay-for-performance” approaches deliver better environmental outcomes than action-oriented approaches; however, such schemes are not yet widely used and more evidence is needed (Matzdorf and Lorenz, 2010[80]; Burton and Schwarz, 2013[49]).

- For biodiversity objectives, targeted schemes which focus on the needs and spatial distributions of specific species of interest have generally been shown to be more effective than untargeted ones (Batáry et al., 2011[145]; Lankoski, 2016[46]; Wrbka et al., 2008[149]).
- Based on the reviewed literature, when applied to intensively farmed landscapes, AES appear to have the best environmental performance when applied landscapes of simple to intermediate complexity (Batáry et al., 2011[145]; Kampmann et al., 2012[47]; Wrbka et al., 2008[149]) and when tailored to the specific region in which they are being implemented (Batáry et al., 2011[145]).
- Conservation payments can have unintended negative environmental impacts in a context of risk: the payments can induce risk averse farmers to use *more* fertiliser and pesticides rather than less as they become less focussed on mitigating risk by minimising inputs (Vercammen, 2011[112]).
- AES under the EU CAP have in some cases caused extensification of agriculture (Schroeder, Gocht and Britz, 2015[107]; Pufahl and Weiss, 2009[106]); a similar effect is observed in the US where EQIP[44] payments may lead to an expansion in irrigated acreage (Wallander and Hand, 2011[150]). To the extent that extensification brings land into production that was not previously cultivated, AES could have unintended negative environmental impacts.

Economic impacts

- There is some evidence that providing positive incentives to adopt sustainable farming practices (e.g. ecological direct payments under the EU CAP) can increase farm technical efficiency, providing a "win-win" outcome (Picazo-Tadeo and Reig-Martínez, 2007[40]; Mamardashvili and Schmid, 2013[105]; Manevska-Tasevska, Rabinowicz and Surry, 2013[108]).
- The evidence on the impacts of participation in voluntary AES on farm profitability is mixed. In some cases, participation in AES is associated with increased yields and profitability, which could be viewed as a "win-win" outcome. However, they may alternatively point to issues such as selection bias, a lack of additionality and the creation of "windfall gains" for participating farmers—i.e. that farmers are receiving payments for actions they would have undertaken in the absence of the policy, for example because they are intrinsically motivated to produce environmental services or because it is economically profitable to do so. (Salhofer and Streicher, 2005[109]; Chabé-Ferret and Subervie, 2013[73]; Baylis et al., 2008[113])

5.4. Public investment in structural adjustment towards "greener" agricultural systems

- Two key examples of public investment in structural adjustment towards "greener" agricultural systems are policies that support farmers to adopt more sustainable technologies on-farm, and policies that support farmers to retire agricultural land from production.

Sustainability impacts

- Studies evaluating investment in water-saving technology (e.g. irrigation infrastructure efficiency upgrades) have generally shown this policy measure has generally increased the productivity of affected farmers in terms of water use (diversions) per unit of agricultural production. However, the overall effect on the environmental performance of

[44] Environmental Quality Incentives Program.

agriculture depends on factors such as changes in return flows, whether productivity increases stimulate increased irrigated area (i.e. expansion on the intensive margin), changes on the intensive margin, or change of crop type. (Gleason et al., 2011[62]; Riffell et al., 2010[63]; Waddle, Glorioso and Faulkner, 2013[64])

- Reviewed studies show that AES that pay farmers to retire agricultural land appear to have generally positive results for biodiversity on retired lands, and can have broader environmental benefits in terms of improved ecosystem functions, although such wider benefits are less studied. However, there is a need to consider such programmes holistically and account for the potential for intensification on remaining working land, as well as to ensure retired land is appropriately managed (i.e. avoid land abandonment).

Economic impacts

- Publicly-funded investment in improving the efficiency of agricultural irrigation infrastructure has been found to induce structural change, with farmers adjusting towards higher-value crops. While studies found infrastructure upgrades have increased productivity of irrigated agriculture, higher electricity costs may offset productivity gains (Tarjuelo et al., 2015[124]; Rodríguez-Díaz et al., 2011[125]).
- Land retirement programmes can have a variety of economic impacts on farmers, including: increasing the productivity of adjacent field (e.g. by providing a buffer to erosion), and increasing land values for agricultural working lands (which impacts farmers differently depending on whether they own or rent land). Reviewed studies show that it is also important to consider interactions between land retirement programmes and programmes targeting working lands (Feng et al., 2005[117]; Lubowski et al., 2006[115]; Sullivan et al., 2004[151]; Wu and Lin, 2010[116]).

5.5. Environmental taxes and charges

- Key examples are input taxes (e.g. fertiliser taxes), performance taxes (e.g. nutrient or greenhouse gas emissions taxes, and natural resource tariffs (e.g. water or water infrastructure tariffs).

Sustainability impacts

- Based on reviewed studies, input taxes (e.g. nitrogen taxes) appear to be an inefficient mechanism for (water) pollution control: in general, very high taxes are needed to induce reductions in fertiliser use to environmentally-sound levels, and they therefore have a higher impact on farm profits than nutrient input restrictions (Picazo-Tadeo and Reig-Martínez, 2007[40]).
- Emissions taxes appear to be the first-best instrument for addressing nutrient emissions from agriculture (Martínez and Albiac, 2006[41]), however they may be difficult to implement in practice due to difficulties in tracking non-point source emissions.
- Based on the reviewed studies, input taxes appear to be more efficient than water pricing mechanisms (taxes on irrigation water) for addressing water pollution issues (Martínez and Albiac, 2006[41]; Semaan et al., 2007[42]).
- The relative performance of input standards and input taxes in achieving environmental objectives may depend on soil type.
- Water taxes or tariffs are not often used to manage water scarcity for agriculture, but are used in some cases to recover costs of water irrigation infrastructure. Water pricing appears to be an inefficient mechanism for (water) pollution control (Martínez and Albiac, 2006[41]; Semaan et al., 2007[42]).

Economic impacts

- Reviewed studies find that, in general, very high taxes are needed to induce reductions in fertiliser use to environmentally-sound levels (e.g. Picazo-Tadeo and Reig-Martínez (2007[40])).

5.6. Tradeable allowances

- Examples include emissions trading schemes, water quality markets, tradeable offset schemes, and in-lieu-fee programmes.

Sustainability impacts

- Well-functioning water markets have been shown to be an effective mechanism to manage consumptive water use, but may need to be used in conjunction with reallocation mechanisms (for example, regulations on minimum environmental flows, "buyback" mechanism to purchase water for the environment) in order to address over-allocation issues. Potential for (unintended) negative environmental impacts (e.g. changes in return flows and dilution flows, activation of previously-unused water access rights) needs to be addressed (Connor and Kaczan, 2013[55]; Bjornlund, 2004[56]; Grafton et al., 2011[57]).
- There is limited quantitative information available on the outcomes of biodiversity offset markets (Bull et al., 2013[58]; Vatn et al., 2011[60]).

Economic impacts

- Water markets have been shown to provide incentive for agricultural and other water users to adapt and improve their productivity (water use efficiency) (Qureshi et al., 2009[127]).
- Water markets have been shown to deliver significant benefits for agriculture, particularly in dry years, and particularly in relation to highly-developed water markets such as those in the Australian southern connected Murray-Darling Basin. (Grafton et al., 2011[57]; Peterson et al., 2004[152]; Wheeler et al., 2014[128]).
- Limited evidence is available on the impacts of other market mechanisms (e.g. biodiversity offsets) on economic performance.

6. Limitations of the literature and directions for future work

While this review shows that researchers have already expended considerable effort in considering various specific aspects of the impacts of environmental policies relevant for agriculture, there are several ways in which future work could improve the evidence base, particularly in relation to assessing the *environmental impacts* of agri-environmental policies. It should be recognised that there are some important limitations of existing research into the cost-effectiveness of agri-environmental policies. For example, Balana, Vinten and Slee (2011[66]) find that results from most cost-effectiveness analyses of programmes implemented under the EU's Water Framework Directive are unlikely to be generalizable because the studies: i) are based on "representative" farm types and as such may fail to capture the inherent heterogeneity across farms, ii) concentrate on the effect of a single measure, and iii) do not take into account uncertainties in both costs and environmental effects estimates.

The scientific effort to assess the environmental impacts of agri-environmental policies requires much more work, and this is a limiting factor to being able to successfully evaluate the policy cost-effectiveness (Baylis et al., 2016[153]; Kleijn et al., 2006[146]). Baylis et al. (2016[153]) note a number of practical difficulties which seriously impede robust impact evaluation for agri-environmental policies: policies having multiple outcomes or multiple scales; spatial spill-overs (leakage); failure to account for confounding factors; the limits of practically implementing randomised control trials in an agricultural context, and the small scale of many agri-environmental initiatives. Hanley, Whitby and Simpson (1999[154]) also point out that too often attempts to assess policy effectiveness are based on measures of participation rather than ecological effectiveness. In many cases, there is a lack of suitable indicators to assess the environmental effectiveness[45] of agri-environmental policies, as well as a range of methodological issues such as time lags, disentangling the effects of multiple policy and market effects, etc. (Mauchline et al., 2012[155]; OECD, 2012[156]; Woon Nam et al., 2007[157]; Baylis et al., 2016[153]). Cisilino et al. (2018[51]) also point out that crucial gaps or mismatches between how information is recorded in different databases impedes holistic analysis. Coderoni and Esposti (2018[68]) note that where the original purpose of the database is different than researchers' subsequent purposes, missing values for variables of researcher interest may be a significant issue (in their case, the authors report significant missing information on variables needed to calculate farm-level GHG emissions in the EU FADN database).

Kleijn et al. (2006[146]) and Jeanneret et al. (2010[158]) find that some aspects of the policy design itself make evaluation of effectiveness difficult. For example, too often the vague specification of policy objectives frustrates any attempt to assess the extent to which those objectives are achieved (OECD, 2005[148]; Jeanneret et al., 2010[158]; Kleijn and Sutherland, 2003[48]; Kleijn et al., 2006[146]; Mauchline et al., 2012[155]; Baylis et al., 2016[153]). In relation to policies for the provision of ecosystem services in agriculture, Hardelin and Lankoski (2018, p. 8[5]) found that "structured evaluations of the impacts of existing policy mixes on ecosystem services are often lacking". Briske et al. (2017[159]) similarly find that environmental outcomes of agri-environmental policies are insufficiently documented. Thus, policy-makers and administrators have a role to play in designing policies which are able to be effectively evaluated, particularly in relation to setting measurable policy objectives and conducting methodical evaluations. More explicit design of policies to facilitate evaluation could improve the knowledge base within this domain.

Further, evaluations of agri-environmental policy impacts rarely assess both environmental and economic aspects. This makes it more difficult to compare the impacts of policies on agricultural economic performance (e.g. productivity or profitability) and agricultural sustainability and to assess policy cost-effectiveness (Ansell et al., 2016[160]).

[45] Environmental effectiveness is "the capacity of the instruments to achieve stated environmental goals or targets of practices" (OECD, 2010[4]).

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