# Guidelines for Cost-effective Agri-environmental Policy Measures

OECD

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# *Foreword*

The main authors of this study, which was prepared for the Joint Working Party on Agriculture and the Environment, were Jussi Lankoski and Andrea Cattaneo. The study draws on background papers prepared by consultants: Professor James Shortle (The Pennsylvania State University, United States) and Professor Richard Horan (Michigan State University, United States), who prepared a background paper dealing with policy design for environmental standards, environmental taxes and tradeable permits, and Dr Simon Mortimer and Dr John Finn, who prepared a background paper dealing with the Agri-environmental Footprint Index methodology for the evaluation of agri-environmental policies. The aim of the report is to provide policy makers with a set of tools for the design and implementation of cost-effective policy measures to address environmental issues in the agricultural sector.

Wilfrid Legg provided overall guidance.

The study was prepared for publication by Françoise Bénicourt and Theresa Poincet.

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# Executive Summary

The aim of the *Guidelines* study is to help policy makers with additional tools to design and implement cost-effective agri-environmental policies. It focuses on environmental standards, environmental taxes, agri-environmental payments and tradeable permit schemes to address agri-environmental concerns (externalities). It is important to note that the goal of this study is not to promote any specific policy instrument or instrument-mix in any OECD country but to improve understanding of how different types of policy instruments can be used, in what context, and which key design and implementation issues need to be considered for the success of a given instrument.

The study focuses on the design and implementation of environmental standards, environmental taxes, agri-environmental payments and tradeable permit schemes to address agri-environmental concerns. It fundamentally deals with two sets of issues. The first set addresses choices across types of policy instruments. For instance, when is an environmental tax better than a standard, or when is permit trading better than an environmental tax? The second set addresses the design of particular instruments.

On the basis of the policy analysis some important lessons for instrument choice and design can be drawn. From a general point of view, the most cost-effective measures are those: 1) designed to attain specific environmental performance goals; 2) targeted on those farmers best able to address environmental problems at the least cost; and 3) leaving farmers flexibility to choose how to meet the goals. However, given the complexity of the linkages between policies and environmental performance, the diversity of situations across and within countries with respect to farmers' compliance costs and agri-environmental conditions, the transaction costs of differentially targeted measures, and equity considerations, it is often difficult in practice to implement policy measures that fully meet these requirements.

Three key conclusions that emerge from the analysis of measures available to address agri-environmental concerns are:

1. *For standards, taxes and permits* and the informational issues that arise in their application, and design factors that influence their performance, there is no unique instrument type or design that can promise to achieve agri-environmental policy goals and to do so cost-effectively over all conditions. This conclusion derives from the physical complexity of agriculture's impacts on environmental systems, uncertainty about key economic and environmental relationships affecting environmental and economic outcomes, and the limited resources and capacities of environmental agencies. Political and equity considerations create additional complexity.

2. *For agri-environmental payment programmes,* including fixed-rate payments based on practices, differentiated payments/contracts and conservation/green auctions, the cost-effectiveness of agri-environmental payment programmes could be improved by using performance-based enrolment screens. This can be done through using proxies, like an environmental benefit index, wherever data availability allows this. However, cost-effectiveness gains achieved through performance-based measures have to be weighed against the potential increase in policy-related transaction costs.

3. *For policy instrument mixes,* these should, to the extent possible, combine instruments that complement and do not conflict with each other, in order to be cost-effective. Since no single policy instrument is likely to be unambiguously preferred over all available instruments under all conditions, the optimal strategy may involve the use of a mix of policy instruments. Instrument mixes addressing nonpoint source pollution from agriculture and the linking of income support payments and environmental performance (where practice-based measures are not feasible) are approaches that have been adopted in several OECD countries through environmental cross compliance.

# *Chapter 1*
# Introduction

Environmental standards, environmental taxes, agri-environmental payments and tradeable permit schemes are important tools in the policy makers' arsenal for managing agri-environmental issues. Applications vary internationally, and have evolved over time as lessons are learned about the merits of alternative approaches for different problems and as the problems themselves change. The scope of agri-environmental problems and issues has expanded over time including recognition that agriculture also contributes to providing environmental services. As a consequence, the types of policy instruments used to address them have expanded with varying degrees of success.

This study focuses on the design and implementation of environmental standards, environmental taxes, agri-environmental payments and tradeable permit schemes to address agri-environmental problems. It is important to note that this is not a complete or exhaustive list of available policy instruments for policy makers. In many OECD countries governments assist farmers through funding education and research and development as well as providing technical assistance and extension services at the farm level in order to increase voluntary adoption of environmentally friendly farming practices and technologies.

For example, educational programmes can encourage farmers to take pro-environmental actions leading to environmental improvements when: i) pro-environmental actions also increase profitability, ii) farmers have strong altruistic or stewardship incentives, and iii) there are also significant on-farm costs due to environmental damage (Ribaudo *et al.*, 1999). In fact, some educational programmes relating to conservation tillage, nutrient management, integrated pest management and irrigation water management have resulted in win-win solutions in which both profitability and environmental performance have improved when compared to conventional practices (Horan *et al.*, 2001). However, both potential win-win solutions and stewardship incentives are unlikely to satisfy society's overall demand for environmental quality from agriculture (*i.e.* when environmental externalities remain) and thus there is a need for more direct policy interventions that are the focus of this study. The study is fundamentally concerned with two sets of issues. One set has to do with choices across types of policy instruments. For instance, under which criteria does an environmental tax perform better than a standard, or permit trading perform better than an environmental tax? The second set of issues has to do with the design of particular instruments. Economic theory, supported by simulation analyses and *ex-post* assessments of environmental instruments, demonstrates that the details of implementation can matter greatly for both environmental and economic outcomes. This is particularly true given the unique features of agri-environmental problems. For example, the success of national cap-and-trade markets for air pollution has stimulated significant interest in using similar markets to address water pollution from point sources and agricultural and other nonpoint sources of water pollution. But the simple cap-and-trade model generally is not plausible in the case of agricultural nonpoint pollution due to the difficulties associated with measuring farm-level discharges, the random fluctuations in these discharges due to weather-related events, and the significant heterogeneity in how farm-level discharges from different locations are transported to water bodies. Instead, more complex markets are required, making design issues of the utmost importance.

The study has several objectives. One is to describe a menu of types of standards, taxes, payments and trading mechanisms. A second is to provide information to guide choices among these instruments. A third is to provide information to guide the design of particular types of instruments.

There is a substantial body of OECD work on which the study draws. This includes the proceedings of the Helsinki Seminar on the *Environmental Benefits from Agriculture* (OECD, 1997), which defines the concept of the "reference level"; the proceedings from the Workshop on *Evaluating Agri-environmental Policies* (OECD, 2005b); reports on: *Analysing Linkages between Agricultural Policies and their Environmental Effects: SAPIM Analysis* (OECD, 2010c); *Information Deficiencies in Agricultural Policy Design, Implementation and Monitoring* (OECD, 2007a); *The Implementation Costs of Agricultural Policies* (OECD, 2007b); *Effective Targeting of Agricultural Policies* (OECD, 2007c), and on *Financing Agricultural Policies with Particular Reference to Public Good Provision and Multifunctionality – Which Level of Government*? (OECD, 2005c). Although broader in topic area, they provide important reference material for this study. The *Synthesis Report on Agricultural Policy Design and Implementation* (OECD, 2008a) also provides useful contextual background as do the two synthesis reports of the work in the JWP on agriculture and the environment (OECD, 1998; 2001). Also of particular relevance are the recent OECD studies on *Instrument Mixes for Environmental Policy*

(OECD, 2007d), and the *Framework for Effective and Efficient Environmental Policies* (OECD, 2008b).

Useful information to guide instrument choice and design decisions is inherently contingent on clearly defined policy objectives. Thus, this study begins with a brief overview of the functions of agri-environmental policy instruments and criteria for policy evaluation (Chapter 2). Chapter 3 provides an overview of the core policy design parameters. Then a more specific analysis of agri-environmental policy mechanisms or instruments is presented. The various types of environmental standards, taxes, tradeable permit schemes, agri-environmental payments and policy-mixes that can be constructed are introduced and analysed in relation to the design parameters (Chapters 4 to 6). Chapter 7 provides a brief overview of agri-environmental policies in OECD countries. A discussion of the use of formal *ex-ante* and *ex-post* policy analysis and evaluation to assess the performance of alternative types of policies is provided in Chapter 8. The study concludes with an extensive summary and good policy practice principles for the design and implementation of cost-effective agri-environmental policies, in Chapter 9.

# *Chapter 2*
# Objectives of agri-environmental policy instruments and criteria for policy evaluation

## *Agri-environmental policy instruments*

The fundamental purpose of agri-environmental policy instruments is to achieve environmental policy objectives that would otherwise not be achieved given the absence or poor functioning of markets. Achieving those objectives requires either controlling environmental stress, such as polluting emissions, or inducing farmers to undertake pro-environmental activities to increase the flow of ecological services, such as management of agricultural practices and land to enhance desired wildlife habitat. In either case, achieving the desired end requires changes in producer decisions consistent with the achievement of the agri-environmental policy objectives.

A simple representation of agricultural externalities illustrates the role of producer decisions (Figure 2.1). Farms produce outputs using inputs that are private goods transacted in markets (market inputs, *e.g.* labour, fuel, machinery), and inputs that are public goods that are not transacted in markets (non-market inputs, *e.g.* climate, air quality).[1] Farms sell their final outputs in markets, but both types of inputs are not fully transformed into market goods. Fuel used to till land also produces air emissions. Farm animals are sources of odours, greenhouse gases, and solid and liquid wastes that, depending on how they are handled, can adversely or positively affect air and water and soil quality. Farms also produce a range of beneficial environmental effects that are also not sold on the market. Land can be managed to produce wildlife habitat, flood control, and other landscape amenities (*e.g.* scenery). Individual farm decisions may simultaneously contribute to multiple externalities. Certain tillage practices can simultaneously release carbon from the soil and increase sediment erosion and nutrient runoff. Over-grazing of livestock along (riparian) stream banks can lead to increased stream bank erosion, sedimentation of streams, nutrient discharges into streams, and reduced flood protection. Finally, individual farms contribute in the aggregate to determine overall environmental conditions. For instance, nutrient runoff travelling over land can affect the nutrient cycle on other land parcels and also combine with nutrient runoff from other farms and sources *en route* to water resources. Similar processes occur as the nutrients move through water resources, with the combined effect often exceeding the individual effects. And, the overall appearance of the landscape reflects the combination of appearance of individual farms ("spatial jointness").

While there are multiple factors explaining farmers' choices of what and how to produce, economic incentives have a large role in determining what farmers do individually and collectively. Indeed, agricultural production is highly responsive to markets for agricultural products and inputs (Shortle *et al.*, 1998) within the limits of given natural conditions, in so far as farmers benefit from increasing the value of market products relative to the value of market inputs. The theory of externalities explains that "missing markets" for environmental goods lead to individual and collective activities that are either environmentally harmful, or that fail to supply ecosystem services at optimal levels (Hanley *et al.*, 2007). In Figure 2.1, farmers have market incentives related to the returns from selling market outputs, and the costs of market inputs, but lack market incentives for managing non-market outputs. (It may also be argued that a loss of the quality of a large number of non-market outputs affects the production conditions for market [commodity] outputs so significantly that corrective action is unilaterally taken by the farmer, such as in the case of soil degradation.)

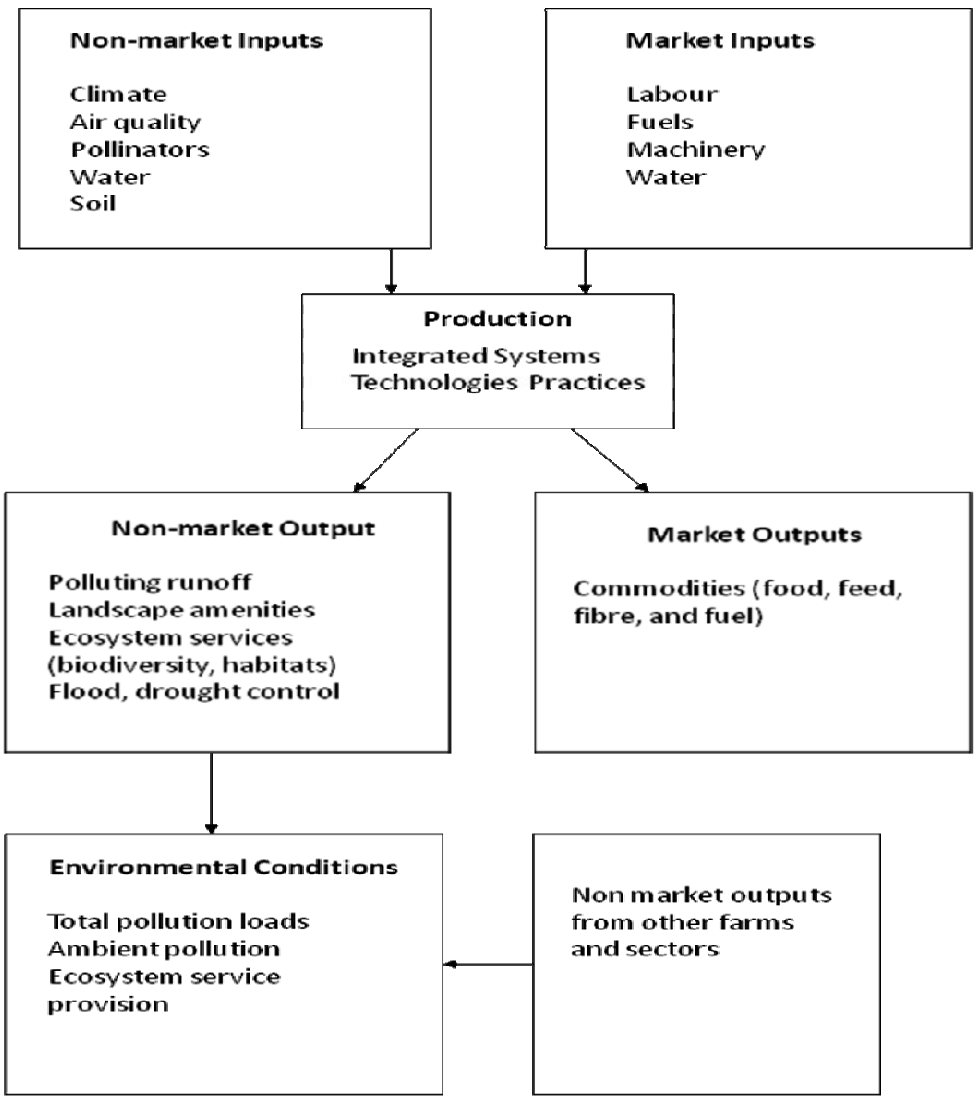
The over-arching objective of agri-environmental policies is therefore to correct for the incentive failures resulting from missing markets that lead to sub-optimal levels of environmental protection (or enhancement). This requires choosing instruments and making policy design decisions that result in the instrument, or mix of instruments, which induce individual and collective behaviour consistent with the achievement of environmental objectives beyond the level of corrective action induced by a decline in market output.

## *Criteria for agri-environmental instruments: policy performance metrics*

Measures of environmental and economic performance – so-called performance metrics – are needed to guide *ex-ante* instrument choice and design decisions and to measure *ex-post* instrument performance, and thus to discuss instrument design and choice in a systematic way. Environmental objectives should be set (and then later be achieved) with economic efficiency in mind, such that: i) the marginal benefits and marginal costs of achieving the environmental objectives should balance reasonably well; and

ii) whatever environmental goal is set, that goal should be achieved at least cost (OECD, 2008b). Where performance indicators cannot be obtained or are the result of other factors external to the producer's influence, proven practices should be used as a basis.

**Figure 2.1. Agricultural production and agri-environmental externalities**



### *Environmental effectiveness*

The first criterion for evaluating instruments is the capacity of the instruments to achieve stated environmental goals or targets of practices. While *environmental effectiveness* is an obvious criterion, it is not trivial. There are numerous examples of regulatory and other types of environmental policy instruments that have proven (or can be so demonstrated prior to implementation) to poorly achieve the environmental goals they are intended to achieve. Guarding against such a result begins with an explicit statement of the environmental goals and selection of measurable environmental indicators or environmental practices that empirically have been shown to lead to the desired environmental outcomes. However, the outcomes depend on many other external factors beyond the influence of the farmer.

A widely used framework for characterising environmental performance indicators is the pressure-state model (OECD, 2003). Environmental *pressure* indicators measure activities that induce environmental change. Pressure indicators may be the proximate cause of environmental change, such as nutrient runoff from farms entering water bodies. In Figure 2.1, such indicators would measure non-market outputs. Pressure indicators may also measure up-stream variables that determine the level of the proximate cause. In the case of nutrient runoff, up-stream variables would be applications of fertilisers and manure to fields, tillage practices that influence runoff, and conservation practices like buffer strips, the botanical characteristics of which being essential for the amount of nutrient runoff taken up. In Figure 2.1, such indicators would be market inputs, and technologies and practices. Environmental *state* indicators measure the condition of environmental systems that are affected by pressures. Examples of state indicators are the concentration of pollutants in environmental media, the health of ecosystems, or the ability of resources to support desired uses, such as fishing or swimming. In Figure 2.1, these indicators measure environmental conditions.

Environmental objectives or goals are typically expressed in terms of state indicators. Thus, for example, water quality goals may be expressed in terms of the capacity of the water resource to support designated uses. In contrast, indicators used for management are typically pressure indicators, since it is these that determine human-induced environmental change. Essentially, environmental goals express ends, while management indicators express means.

## *Cost-effectiveness*

A second criterion to guide instrument choice and design decisions relates to the resulting costs of achieving society's environmental objectives. Changes in farm practices to achieve environmental ends will generally involve reductions in farm income (cost incurred and income foregone), unless farmers are remunerated for their actions. The reason is quite simple. If pro-environment changes in farm practices were inherently profitable, then those farmers concerned about the environment or about their income or both, would adopt these pro-environmental practices, even though the environmental performance may not be evident in the short-term. In this case, there would be little need for agri-environmental policy. Thus the presumption, generally supported by the empirical literature, is that pro-environmental behaviours are economically costly (Ribaudo and Horan, 1999; Ribaudo *et al.*, 1999).[2] However, there are cases in which incomplete information and knowledge may hinder the adoption of win-win practices, such as no-till farming (may not always induce a win-win situation) and integrated pest management. In addition to farm-level costs, there may be costs or benefits to input suppliers and to consumers. These economic consequences become relevant if changes in farm practices to achieve environmental ends affect the demands for inputs (affecting input suppliers) or the supply or quality of agricultural goods (affecting consumers).

These economic considerations lead to the second, distinctly economic criterion for instrument choice and design: minimising the cost, prior to remuneration for profit losses if any, of achieving the environmental goal. This criterion is commonly measured by the *cost-effectiveness* of an instrument or instrument-mixes. The cost-efficient policy instrument is one that minimises compliance costs while achieving environmental target, thus maximising cost-effectiveness.

Cost-effectiveness can be defined with respect to reductions in environmental pressures, or in terms of improvements in environmental states. Specifically, farm-level cost-effectiveness means that farm-level outcomes, usually defined in terms of specific environmental pressure indicators, have been attained at least cost. Cost-effectiveness at the (physical) landscape level (or watershed or airshed level) means that landscape-level outcomes have been attained at least cost. It deals with the allocation of environmental management efforts across individual farms within a region. Landscape-level cost-efficiency implies farm-level cost-efficiency because an outcome can only be cost-effective at the landscape level if it is also cost-effective at the farm-level. However, the opposite is not true: farm-level cost-efficiency does not imply landscape-level cost-efficiency. There are two reasons for this. One is that there are differences in the impacts of individual farms on landscape-level environmental outcomes.

Some farms may have a large impact, while others may have little or none. An allocation of effort that requires as much from farms with little or no impact as those with a large impact, even though farm-level effort is cost-effective, would clearly impose unnecessary cost, and thus not result in allocative cost-effectiveness. A second reason is that farms vary in their costs of achieving environmental outcomes.[3] An allocation of effort that requires no more from a low cost supplier of environmental services than from a high cost supplier may again impose unnecessary cost in achieving desired landscape scale outcomes, even though farm-level costs are minimised.

Spatial variation in costs and impacts implies that the cost-effective achievement of landscape-scale environmental goals will generally entail differential levels of environmental effort across farms (*e.g.* Braden *et al.*, 1989; Fleming and Adams, 1997). For example, suppose that policy makers seek a 40% reduction in agricultural nutrient loads in a water course. Requiring 40% in the contributions of all farms would surely fail the allocative cost-effectiveness criterion because of differential impacts and control costs across the farms concerned.

### *Administrative costs*

A third criterion relates to public sector costs and capacities (policy-related transaction costs). Different instruments impose different demands on the management capacities of public agencies, and the costs to the public sector for design, implementation, monitoring and enforcement (Batie, 2005; Krutilla, 1999; OECD, 2007b). First-best instruments can pose design, implementation, monitoring and enforcement costs and requirements that are beyond the capacities or resources of environmental agencies. When this is the case, modifications to the first-best designs are required to construct feasible instruments. For instance, a first-best design for managing environmental risk from the application of a highly toxic pesticide might entail strict standards on the amount of application, timing, weather conditions, location relative to vulnerable resources, and other factors. However, because the actual use is costly to monitor, the best alternative (depending on the capacity of regulatory authorities) may be to simply ban the use of the pesticide. Such an instrument will not be first-best because it does not minimise compliance costs, but it will perform the best under the circumstances and would therefore be a *second-best* solution.

Political constraints may also restrict instrument design to second-best choices. This is because first-best policies will distribute public resources to meet cost-effectiveness criteria, whereas political constraints may result in resources being distributed to meet political or social (equity) criteria.

Targeting based on political criteria will reduce cost-effectiveness if it results in a different but sub-optimal allocation of resources, and it may limit desired environmental outcomes when public expenditures are essential but budgets are limited.

### *Ancillary benefits and costs*

A fourth criterion for instrument evaluation is ancillary (additional) benefits and costs. Ancillary impacts of an instrument may be environmental, economic or related to other objectives (such as food security). In the first case, an instrument that reduces nutrient loads will improve water quality, but it may also improve wildlife habitat if, for example, the technologies used to reduce nutrient loading include establishment of buffer strips or creation of wetlands. Another example is carbon sequestration in agricultural soils that may also provide co-benefits in terms of water quality and biodiversity. Ancillary benefits may also be economic. For example, an environmental tax may generate revenues that could be used to improve social welfare in some other area.[4] An environmental standard that has the same environmental outcome would not generate such revenues and would therefore not yield the ancillary economic benefits, but may incur lower transaction costs. However, it is possible that moving to more targeted instruments may entail losses of some ancillary benefits, also called "dissociation costs" (OECD, 2007b).

Environmental impacts that are ancillary with respect to one policy objective (*e.g.* water pollution control) may be the primary target of another (*e.g.* wildlife habitat conservation or enhancement). When this is the case, policy co-ordination to achieve the multiple environmental objectives is important, particularly if the pursuit of one objective conflicts with the pursuit of another. This is an important issue for agri-environmental problems as agriculture's broad spectrum of environmental impacts may span the domains of multiple regulatory agencies. For instance, regional or state water authorities may oversee water quality policy; state wildlife departments and local land-use authorities might oversee habitat-related issues; and state and national departments of agriculture might oversee soil conservation issues. Each agency could develop its own individual goals and then separately design instruments to address these. Policy co-ordination that simultaneously manages for multiple objectives can realise the gains from the potential synergies and improve overall efficiency. In contrast, unco-ordinated policies reduce efficiency, as they may fail to realise objectives at least possible cost (Weinberg and Kling, 1996). Unco-ordinated policies may also generate unintended environmental and economic consequences.

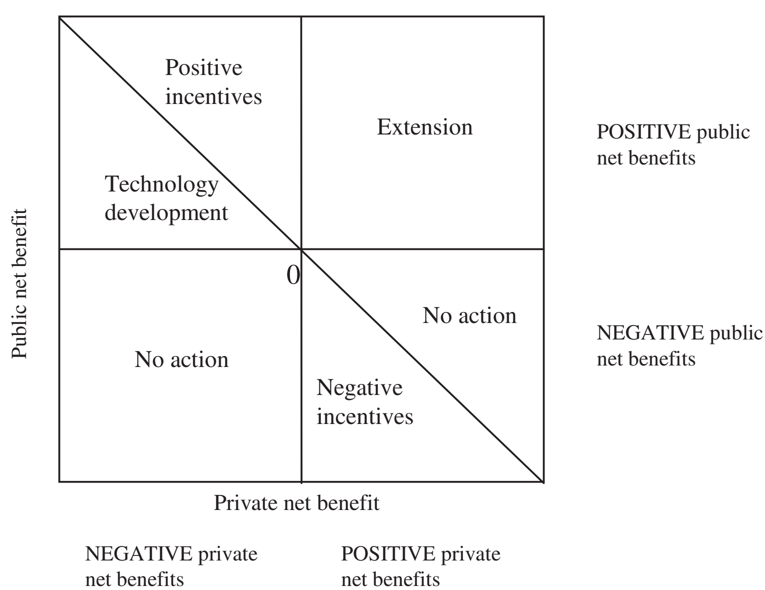
### *Equity*

A final criterion that plays an important role in evaluating policy instruments is the equity of the distribution of economic costs and benefits between and among different groups (producers, consumers, and taxpayers). It can be the case that more than one type of instrument will be capable of producing a cost-effective outcome, but each will yield different distributions of wealth and will therefore be viewed differently from an equity perspective. Thus policy makers will need to weigh up the trade-offs between equity, efficiency and other criteria in choosing among policy instruments.

### ***Public and private net benefits framework for policy instrument choice***

The choice of policy instruments for encouraging environmentally beneficial land-use change have been analysed by Pannell (2008). His analysis suggests that instrument choice should depend on the relative levels of private (or internal) and public (or external) net benefits (Figure 2.2).

**Figure 2.2. Efficient policy mechanisms based on a simple set of rules**




*Source:* Pannell (2008).

*Private net benefits* are defined as the benefits minus costs accruing to the private land manager as a result of proposed changes in land management, whereas *public net benefits* are benefits minus costs accruing to everyone other than the private land manager. In this way, the private net benefit dimension provides insight into the behaviour of the landholder, while the public net benefit dimension relates to the effects on everyone else that flow from the landholder's behaviour. The latter effects, commonly referred to as *externalities*, are used as justification by governments taking action to try to influence the behaviour of economic agents. Projects with positive *public* net benefits fall in the top half of the figure, while projects with positive *private* net benefits fall in the right half of the figure.

Pannell’s analysis (Figure 2.2) essentially outlines what policy mechanism, if any, policy makers should choose in order to maximise the net benefits of intervention:

- ***Positive incentives*** where public net benefits are highly positive and negative private net benefits are close to zero.
- ***Negative incentives*** where negative public net benefits clearly outweigh slightly positive private net benefits.
- ***Extension provision to farmers*** where public net benefits are highly positive and private net benefits are slightly positive.
- ***Technology development*** where negative private net benefits outweigh or are similar to public net benefits.
- ***No action*** where private net benefits outweigh negative public net benefits, or if public net benefits and private net *benefits* are both negative, and thus in both cases the land-use change should be accepted.

It is notable that the areas for positive incentives, negative incentives, and agricultural extension are only the sub-sets of the total. According to Pannell (2008) this framework reveals that the selection of cost-effective measures maybe more sensitive to private net benefits than to public net benefits. Moreover, policy measures, such as positive and negative incentives and extension, are more likely to generate high pay-offs if the private net benefits are close to zero. This is because land-use change can be prompted (prevented) with small positive (negative) incentives.

How can policy makers estimate private and public net benefits? According to Pannell (2008), with regard to public net benefits the framework should not require much extra effort when compared to what policy makers are already doing when choosing which policy instruments to adopt on the basis of environmental benefits. If further precision is needed, then a range of market and non-market valuation methods may be appropriate depending on types of benefits and costs (Pannell, 2008).

## *Performance metrics and uncertainty*

Each of the performance metrics described above is subject to several forms of uncertainty on the part of the regulatory agency, and each form of uncertainty is relevant to the analysis of payments, standards, taxes, and permit trading. One source of uncertainty about both costs and environmental impacts arises because public decision makers, when choosing instruments, are unable to predict with certainty the impacts of their choices on farmers' production and land-use practices, and the costs to farmers of changes in their practices. Economic models can be used to forecast policy-induced changes in production and land-use practices and compliance costs, but forecasts are always subject to uncertainty. There are two implications of this *ex-ante* uncertainty about compliance and compliance costs. One, as stated, is that the economic costs of prospective policies are uncertain. A second is that the environmental outcomes, as measured by pressure or state indicators that result from the application of instruments are uncertain, since those outcomes are driven by uncertain changes in production and land-use practices.

Uncertainty about environmental outcomes is affected by additional factors. One is the uncertainty about the levels of individual farmers' contributions to environmental externalities. For example, as has been noted, nutrient runoff contributions to water resources from individual farms cannot be measured because they are diffuse and complex. Models can be, and are, used to forecast the effects of changes in farm practices on environmental pressures, but such models are generally subject to substantial error. Models are also used to predict the effects of changes in farm pressure indicators on environmental state indicators. These models, too, are subject to substantial error. The challenge is to be able to assess the extent of these errors to ensure the results remain pertinent and their interpretation clearly understood. Finally, many agri-environmental processes, such as nonpoint source pollution, are driven by random weather and other events.

*Ex-ante* uncertainty about compliance, compliance costs, and changes in pressures and states are central to the agri-environmental policy problem. There are fundamental implications for choices between payments, standards, taxes and permits, and how these instruments are implemented. This point is elaborated in subsequent chapters.

There are also fundamental implications for the expression of environmental goals. Uncertainty about environmental outcomes implies risk. For environmental systems that are sensitive to variations in stress, optimal management of risk must consider more than just average levels of environmental pressure and state variables. Variability must also be managed. Scientific information, such as ecological dose-response relations, can inform the choices of margins of safety for ambient pollution levels or the appropriate metrics for other types of environmental goals. This is the approach implied, for example, by the US Total Maximum Daily Load approach to water quality management (National Research Council, 2001). Further, assessments of cost-effectiveness must be expanded to apply to the management of environmental risk (Shortle, 1990; Gren *et al.*, 2000; Kampas and White, 2003). Moreover, threshold effects may be important in the case of voluntary agri-environmental policies, since with an insufficient number of participants the result is low environmental effectiveness.

### *The role of property rights, environmental targets and environmental reference levels in policy choice*

An inescapable challenge of choosing between policy instruments is that some of the criteria that guide policy makers' decisions, such as fairness and equity, are dependent on the definition of reference levels and property rights. Therefore, it becomes apparent that defining how to address the environmental impacts of agriculture requires a case-by-case response in relation to the settings of environmental targets and the definition of environmental reference levels based on the identification of existing property rights determining who can demand remuneration and who is liable for charges.

The definitions of environmental targets and reference levels vary between countries. Environmental targets depend on society's preferences for environmental quality, while reference levels depend on the country's traditions and laws in defining property rights. The efficient setting of environmental targets has to balance the benefits of pursuing environmental objectives against the resulting welfare losses due to lower production or consumption of other goods and services. But, whereas the setting of environmental targets is based on efficiency considerations, the issue of identifying the relevant environmental reference levels (who should bear the costs of reallocating resources to meet environmental targets) is based on distribution (equity) considerations and property rights.

Figure 2.3 (OECD, 2001) illustrates four different cases with which farmers may be confronted in relation to such parameters (where X represents the level of environmental quality corresponding to environmental targets [$X^T$]; reference levels [$X^R$]; and current farming practices [$X^C$]). All cases (A to D) represent an identical environmental outcome and allocation of farm resources as the environmental target, $X^T$, is the same. What differs among these cases is the distribution of costs associated with achieving the defined environmental target (*i.e.* who pays or who is charged).

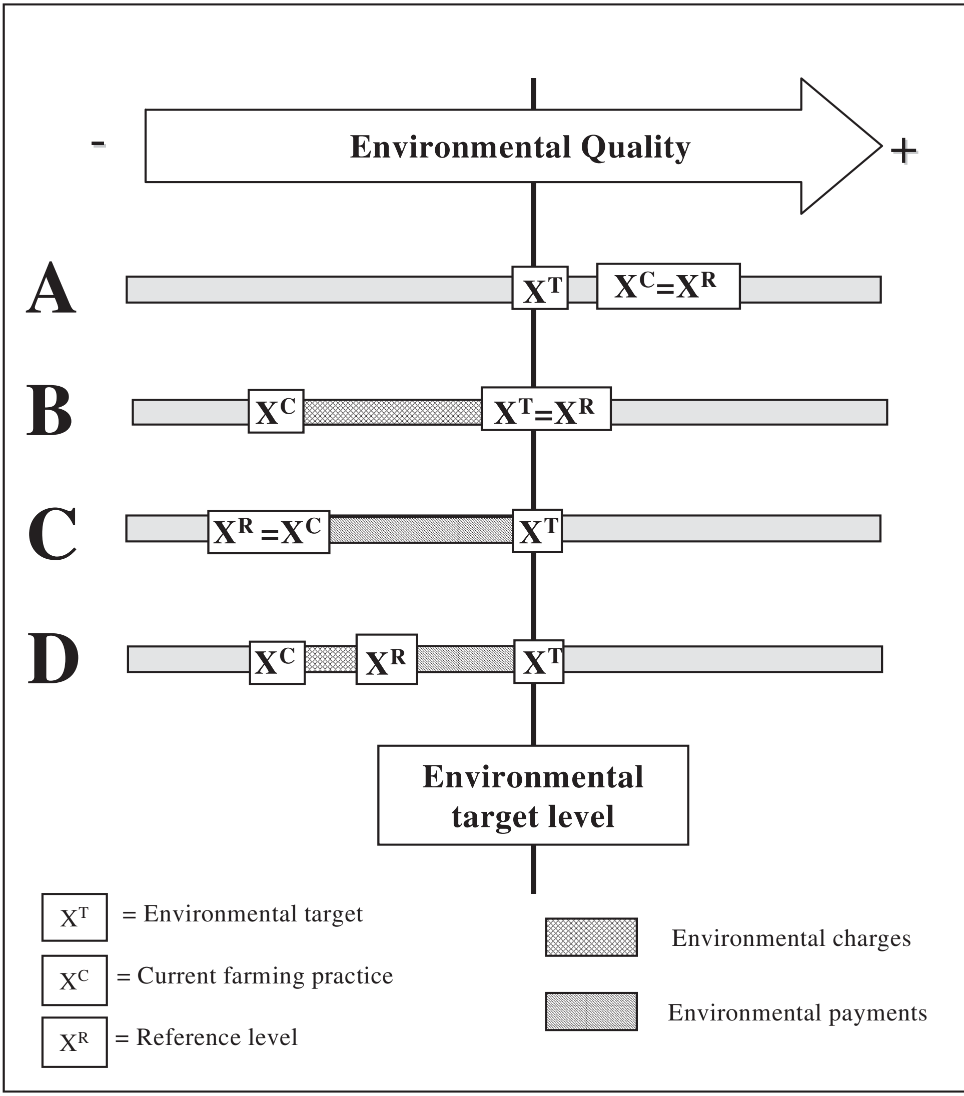
***Case A*** represents a situation where current farming practices provide a level of environmental quality corresponding to a reference level ($X^C$=$X^R$) above the environmental target ($X^T$). Thus, farmers are already using the farming practices required for achieving the socially desired environmental outcome. With $X^T$ and $X^R$ achieved at zero opportunity costs, *no policy action* is needed. In such a case, the reference level $X^T$ would normally be achieved through current farming practices $X^C$ ("good farming practices").

***Case B*** represents a situation where current farming practices ($X^C$) provide environmental performance below the reference level defined at the level of the environmental target ($X^T$=$X^R$). In this case, farmers are emitting pollution for which they do not have the property rights ($X^C$ <$X^R$), and they need to adopt farming practices required to achieve the desired environmental target level ($X^T$) *at their own expense.* If not, the government may charge a tax or penalty to induce compliance.

***Case C*** represents a situation where current farming practices achieve environmental performance corresponding to the chosen reference level ($X^C$=$X^R$) that is below the target level ($X^T$). As in this case property rights in land use are attributed to farming practices achieving an environmental reference level below the environmental target level, farmers *may need to be compensated* for changing from current farming practices ($X^C$) to practices required to achieve the environmental target ($X^T$).

***Case D*** represents a situation similar to Case C, where current farming practices ($X^C$) provide environmental performance below the environmental target level ($X^T$), but with the reference level above the environmental performance level of current farming practices ($X^C$) and below the environmental target ($X^T$). For improving their environmental performance, farmers need to adopt appropriate farming practices *at their own expense* up to the reference level ($X^R$) – if not, the government may charge a tax or penalty. Requirements for farmers to further improve their environmental performance beyond $X^R$ (for example, to reach the environmental target $X^T$) need to be remunerated.

**Figure 2.3. Environmental targets and reference levels**




*Source:* OECD (2001).

# Notes

1. Environmental change affects farms directly through changes (beneficial or otherwise) in public environmental inputs. In this case, agriculture is the recipient of an externality.

2. Some behaviours may simultaneously generate public and private benefits, but only to a point after which further investments generate private costs. At this point, further pro-environmental investments can often still enhance net social benefits.

3. Note that landscape level efficiency requires that marginal compliance costs should be equalised (equi-marginal principle), but the total or average compliance costs could vary.

4. Note that the so called tax-interaction-effect literature argues that in the case of environmental policy (as well as agricultural policy and trade policy) the other-market effects do not cancel out. In particular, the nature of environmental regulation, whether through regulations or taxes, systematically worsens the distortion in the labour market that arises from an existing income tax.

# *Chapter 3*
# Policy design parameters: an overview

In order to achieve an intended objective, a plan and means to reach it are required. The desired objective can be defined by choices of environmental goals along with the economic goal of cost-effectiveness. The plan then entails choosing and implementing policy instruments to achieve the desired outcome.[1] Some general issues involved in developing this plan are outlined in this chapter, with design requirements for specific instruments presented in greater detail in subsequent chapters.[2]

Instrument design is accomplished by making choices about various design parameters. Some specific choices may vary across instruments, but generally the available parameters can be summed up with three broad questions: 1) to whom and to what degree, among the set of possible contributors to environmental externalities, should the instrument be applied – that is, *who to target*?; 2) what is the optimal target variable or metrics, for defining and measuring individual farm-level compliance with environmental target – that is, *what to target* at the farm level?; 3) which incentive – that is, *what specific policy instruments* (*e.g.* payments, environmental standards, environmental taxes, and tradeable permit schemes) should be tied to the chosen compliance metrics to induce the changes in farm-level behaviour that are needed to produce the desired outcome?

## *Who to target?*

A fundamental question in the design of an environmental instrument is to whom it is applied.[3] One aspect of this question is the geographic region to which the instrument applies. The choice here depends on the geography of the environmental problem to be solved. Thus, for example, if the goal is to reduce nutrient pollution of a water course, then agri-environmental instruments should be applied within watersheds where agricultural activities contribute the nutrient run-off. Additional issues become involved. For example, different parts of the watershed may be under the control of different political jurisdictions, leading to an issue of co-ordinated or unco-ordinated instrument choices by different jurisdictions.

The second aspect is who within the geographic region should be targeted. Instruments generally, though not always, ought to be directed at those who are directly responsible for environmental harm or who are most capable of providing environmental enhancements. Within this set, concern for monitoring, enforcement, and environmental protection costs imply focusing management efforts among farms based on their relative capacity to provide environmental improvements – that is, differentially targeting individual producers with an eye towards improving landscape-level cost-efficiency. Continuing with the water course example, targeting lands on which farm-level discharges are delivered in higher proportions to the water course will be more effective and hence more cost-efficient than targeting lands from which few nutrients are delivered. Also, it will be easier and more cost-efficient to encourage abatement on farms having low incremental abatement costs. The correlation between compliance costs and environmental benefits (environmental sensitivity) has a large impact on the budget for voluntary policy instruments. In other words, if environmentally sensitive land is also low productive land (low compliance cost) then environmental goals can be achieved with a smaller budget.

Targeting producers on the basis of anticipated environmental impacts can be complicated by the uncertainty about how farm-level actions will translate into environmental improvements. *Ex-ante* predictions must be based on science-based modelling describing the external impacts of farms' practices. As has been noted, agri-environmental relations are generally complex and model uncertainty is often an issue. Yet, decisions must still be made. Decision science provides procedures for taking into account model-based uncertainty and the risk this implies in achieving the policy goals.[4] As a common rule, it is important to state clearly the limitations of the models and the uncertainties related to the results so that well-informed policy decisions can be made on that basis.

Targeting producers on the basis of environmental compliance costs can be complicated by the uncertainty on the part of the regulatory agency about farmers' costs. Farmers generally have better knowledge of their compliance costs than policy makers, although even for farmers it may still be difficult for them to assess their complete costs.[5] This *asymmetric information* creates a problem that is referred to as *adverse selection*, which limits the policy maker's ability to tailor policy instruments to address individual producers' circumstances. For instance, a producer having low environmental compliance costs would be unwilling to divulge this information to an agency that might then decide the farmer can easily achieve more stringent regulations. Some types of instruments can perform better than others in the presence of this uncertainty, so that asymmetric information about compliance costs can be a fundamental driver in the choice of one instrument over others.

It is important to note that answers to the "who to target" question need not be limited to individual farms. For instance, environmental impacts of pesticides are managed in large part by regulations limiting the pesticides that chemical manufacturers can market to farmers and other users. Regulation of a comparatively small number of chemical manufacturers, though it reduces allocative efficiency due to a lack of targeting, is easier politically and administratively than regulation of the many small farms that actually cause environmental harm. In some cases the reduction in allocative efficiency may be offset by reduced administrative costs.

### *What to target?*

A second fundamental question is what is the appropriate measure or basis of compliance at the farm level? Compliance measures are broadly differentiated as being based on environmental outcomes *(performance-based)* or based on farmers' input and technology choices *(input-based* – also referred to as *practice-based* in this study).[6] The distinction is essentially one of ends *versus* means, with performance-based measures emphasising ends and input-based measures emphasising means. The distinction is illustrated by the processes in Figure 2.1. Input-based instruments may directly regulate the levels or characteristics of market inputs used in farm production that affect the level of environmental externalities (*e.g.* pesticides, fertilisers, fuels), or prescribe the specific practices used to produce market outputs or to affect the flow of non-market outputs (*e.g.* specific technologies used, such as nutrient or pesticide best-management practices). Performance-based instruments focus on the flow of non market goods from the farm, such as nutrient runoff or sediment erosion.

First-generation environmental policies for air and water quality protection (*i.e.* those enacted in the late 1960s and 1970s) were largely input-based. In the case of agriculture, these policies included bans on some pesticides, and regulations governing uses and practices for others. Though input-based regulations still tend to be used in some instances (*e.g.* to address emissions from large confined animal feeding operations [CAFOs] in the United States), there is ample evidence that input-based policies overly limit the flexibility of farms to choose cost-effective options for reducing their own emissions.[7] The result is reduced efficiency and, in some cases, failure to achieve environmental objectives due to the focus on means rather than ends.[8] In consequence, there is increasing emphasis on performance-based instruments that focus on ends rather than means. The focus on ends gives farmers flexibility to choose the means, for which they will have incentives to do so at minimum cost.

The ideal performance-based compliance measure is a farm's contribution to the externality (Shortle and Horan, 2001). But the application of such performance-based instruments may be limited by uncertainty. The regulator is unable to observe or otherwise measure farmers' contributions to many environmental externalities due to the manner in which these externalities are produced. For instance, nonpoint source effluents of nutrients and pesticides are diffuse processes that occur over landscapes. These processes are also highly random, driven in large part by weather and other environmental conditions that fluctuate over time. Together, these two features prevent accurate observation of or inferences about individual contributions, particularly for the large number of potential contributors that are often characteristic of agri-environmental problems (*e.g.* even a small watershed may contain thousands of farms). This means alternative compliance bases must be considered.

Fortunately, removing *actual* measured performance from the set of possible compliance measures may not imply a loss of efficiency (though it may imply an increase in instrument complexity) (Griffen and Bromley, 1982; Shortle and Dunn, 1986). Economically and ecologically desirable candidates for compliance bases will be: 1) highly correlated with the policy maker's environmental goals, so as to align farmers' interests with those of the agency; 2) enforceable; and 3) targetable in time and space (Braden and Segerson, 1993). One option to measuring performance directly is to use proxies for direct environmental stress (*e.g.* estimates of field losses of nutrient surplus to surface or ground waters) or other site-specific environmental performance indicators that are constructed from observations of site-specific data. Examples of performance indicators for agriculture are estimates of annual average gross soil loss (for managing sediment pollution), estimates of nutrient surplus (for managing nutrient pollution), and estimates of carbon being sequestered under various practices.

An alternative performance measure that might be observable, and which has received considerable interest from economists, is the aggregate impacts of farm-level contributions to externalities – so-called ambient-based measures of environmental performance. For instance, policies could be based on the ambient concentrations of pollutants in environmental media, on the aggregate provision of landscape amenities, or on changes in regional wildlife populations. This is a special type of performance-based measure in that it is not based on farm-level performance. Rather, the rewards or penalties that individuals face are based on the combined impacts of multiple (possibly very many) farmers. Ambient-based performance measures are generally easier to measure than farm-level performance measures, but their use entails significantly different farm-level incentives for environmental stewardship. The debate on ambient-based instruments is on-going. While some early theoretical studies were promising (*e.g.* Segerson, 1988), later work emphasised potentially critical limitations of the approach. These limitations primarily involve difficulties in designing ambient-based instruments that can encourage individual producers to co-ordinate their efforts to achieve a cost-effective outcome (Cabe and Herriges, 1992; Horan *et al.*, 1998; 2002; Hansen, 1998). While experiments into some of these limitations are now on-going (*e.g.* Cochard *et al.*, 2005), the debate remains academic as ambient-based instruments have not been applied in practice.[9]

Defensible estimates of a farm's contribution to externalities must be derived from reliable, science-based models describing the external impacts of the farm's practices. A fundamental question in assessing the appropriateness of performance measures is, therefore, the availability, reliability, and cost of models for evaluating farm level environmental performance. The answer to this question will depend on the complexity of the environmental problem, the state of the science for the problem, and on the reliability that planners seek in environmental management (*e.g.* National Research Council, 2001). These issues are generally beyond the scope of this study.

The policy maker should consider the cost of developing indicators and the ease with which producers and regulators can make use of the information provided by the indicators. Different indicators will have different farm-level monitoring requirements, so the policy maker should be cognisant of what can be monitored without excessive cost.

Even if measurement of actual environmental impacts were possible in some instances, the high degree of natural variability of processes such as nonpoint source pollution, carbon sequestration, and flood prevention means that farmers will be unable to control these performance outcomes deterministically (without randomness). In this respect standards could be defined in terms of a probability of their attainment. For instance, a standard based on polluted runoff could be defined in terms of the mean or variance of polluted runoff, or it could be defined in terms of a probability: that runoff does not exceed a target level more than x% of the time, or that mean monthly runoff levels not exceed some pre-determined value. Monitoring would have to occur over a period of time, perhaps a number of months, to determine the sample distribution of the base. Only then could a producer be determined to be in compliance or not. The required time frame for monitoring may be significantly longer for some pollutants due to long time lags associated with the delivery of the pollutant to a water body. Some agricultural chemicals, such as phosphorus, can build up in the soil. Changes in management may not result in changes in water quality until the chemical stored in the soil is depleted.

Also, producers need to understand how their individual production decisions affect the indicator if that indicator is to be an effective compliance metric. Extremely complex indicators may lose their appeal if producers do not understand the relation between their actions and the indicators. However, excessive simplicity may also reduce effectiveness, as the ability of producers to improve cost-effectiveness is ultimately limited by the sophistication of the models being used to make predictions.

Another option for compliance bases that has received significant attention is inputs, practices, or land uses that are correlated with environmental externalities. In agricultural nonpoint pollution control, for example, these would include polluting inputs such as fertilisers and pesticides, farming practices that affect the movement of these chemicals into the environment, and land uses such as establishing buffer strips along field edges. Practices such as no-till and land-use changes (such as planting trees or establishing green set aside) can affect a farm's ability to sequester carbon. Land-use changes can also influence the provision of ecosystem services such as amenity values, flood protection, and wildlife habitat.

Though there are some limitations related to input-based compliance measures, they may in practice be the only option. It should also be noted that input-based approaches can be efficient if they are designed properly (Shortle and Dunn, 1986). Specifically, they must be well-targeted across environmentally-important inputs and across producers. This would set them apart from the class of inefficient, first-generation environmental policies that were poorly targeted.

### *Which incentives?*

Achieving desired environmental change inherently requires a move from the *status quo,* or business as usual. That change can be pursued through a variety of mechanisms, including moral suasion, regulatory standards, change in economic incentive structures created by the use of taxes, payments, or the creation of environmental markets, or a combination of the above. Here the focus is on payments, regulations/standards, taxes, and tradeable permit schemes. These alternative mechanisms for inducing change can be combined with alternative compliance measures to define a range of instruments for addressing agri-environmental problems (Annex A). Each combination provides a different type of incentive for farmers to alter their behaviour to promote environmental stewardship. The magnitude of the incentive depends on the magnitude of the instrument. For instance, a small environmental tax will generate fewer environmental improvements than a large tax. The regulatory agency can adjust the instrument magnitudes, both across individual farms and across compliance measures, to attain the efficiencies that are coincident with the chosen environmental goals.

The appropriate stimulus may also require the simultaneous use of multiple instruments. For instance, an important result related to performance-based instruments is that a separate instrument is generally required to address each policy goal (Tinbergen, 1952). Addressing nutrient runoff and stream bank erosion may therefore require separate instruments tied to proxies for each of these items. Additional instruments would be required in the case of input-based approaches. However, there are instances in which combinations of instruments (*e.g.* taxes and standards; a nitrogen tax and a pesticide ban) are appropriate (Baumol and Oates, 1988; Braden and Segerson, 1993; Horan *et al.*, 2004; OECD, 2007d).

The discussion now focuses on a more detailed examination of standards, taxes, and tradeable permits in Chapter 4, agri-environmental payments in Chapter 5, and policy-mixes in Chapter 6.

# Notes

1. Some environmental goals can only be achieved probabilistically (*e.g.*, on average), in which case attainment of the goals can only be assessed over time.
2. Issues associated with public participation and process are not addressed here, though these may be required elements of policy development by some agencies.
3. For a thorough treatment of targeting of agricultural policies, see OECD (2007c).
4. It should be noted that environmental modelling intended to support management of agricultural externalities is an active area of research and application, and that policy makers have indicated a willingness to use modelled measures of agri-environmental performance in a number of contexts.
5. Farmers face uncertainties before changing farm management practices. They often depend on information and external advice in assessing those costs.
6. A third class of compliance measures, output-based measures, would be based on market commodities (*e.g.* regulations to limit the production of pollution-intensive commodities, such as corn). These output-based measures are not considered here as they tend to be so poorly correlated with environmental performance that instruments based on these measures generally produce poor, and sometimes perverse, incentives for environmental management (Baumol and Oates, 1988; Braden and Segerson, 1993).
7. For further discussion on the cost-effectiveness of input taxes, see, for example, OECD (2007d).
8. Historically, input-based policies were also applied uniformly across producers. For instance, technology standards required producers to adopt technologies or practices from an approved list that was not tailored to address individual producers’ environmental or cost characteristics. This lack of targeting reduced allocative efficiency (Davies and Mazurek, 1998).
9. Ambient-based instruments are not discussed further. See Weersink *et al.* (1998) and Shortle and Horan (2001) for more detailed discussions on this issue.

# *Chapter 4*
# Tailoring environmental standards, environmental taxes and tradeable permits

Due to the focus in this chapter on instruments that effectively provide society with initial rights to environmental quality, the discussion of these instruments focuses on the control of negative externalities. The same instruments could in principle be constructed to also address the provision of positive externalities, but in practice other approaches such as payments or subsidies are used to do this.[1]

## *Environmental standards*

Environmental standards are mandates applied to the quality or quantity of marketed products (product standards), technologies or processes (process standards), or environmental performance (performance standards). In terms of Figure 2.1, product standards regulate marketed production inputs or outputs, process standards directly regulate choices of production and pollution control technologies, while performance standards directly regulate measures of non-market outputs (including indicators of environmental performance). Here process and input-based product standards are collectively referred as input standards. In terms of Figure 2.3, standards usually define the reference levels that farmers must reach.

### *Performance standards*

Environmental performance standards are a common method of regulating polluting emissions from non-agricultural sources. Discharge standards that limit effluents from industrial and municipal point sources of water pollution are used widely. Environmental performance standards limiting agricultural externalities are not routine, but they are certainly a plausible instrument in the agri-environmental toolkit.

Environmental performance standards can take a variety of forms, though they typically impose an upper limit on the externality or the selected indicator. Given the inherent variability of agri-environmental externalities, and the often large uncertainty associated with environmental prediction models, more consideration needs to be given to the construction of performance standards to regulate both the anticipated level and the reliability of environmental performance. At the farm level, this can be done by careful choice of the performance indicators to be regulated (*e.g.* limits on both the mean and variance of emissions)[2] and by including other provisions to enhance reliability (McSweeny and Shortle, 1990). These provisions may include restricting the set of practices that are allowed to meet the standard to those considered to have acceptable reliability. In this case, the standard essentially combines performance and input standard elements. Restrictions on the practices eligible to meet a performance standard may also be imposed if the environmental model used to estimate environmental performance accounts for only a limited set of practices.

An argument for performance standards by comparison to input standards is that they allow producers the flexibility to meet mandated environmental outcomes in any way they choose, thus allowing them to find ways to achieve the standards at minimum cost. Performance standards may therefore promote farm-scale cost-effectiveness, and also promote cost-saving technological innovations (Sterner, 2003). These are desirable attributes, though the implied gains will be diminished if the set of practices eligible to meet the performance standard is limited for reasons noted above. The costs of such limitations on practices may be diminished, if not eliminated, by allowing producers to develop a strategy of specific practices that it will use to satisfy the standard and then seek pre-approval from the regulatory agency prior to implementation. This way, the producer and the agency have a contract that specifies the conditions for being in compliance (EPA, 2007).

While environmental performance standards can perform well with respect to improving farm-level environmental performance and doing so cost-effectively (subject to caveats when restrictions are imposed on practices allowed to meet those standards), there are some issues associated with their use. The first is their capacity to achieve overall environmental goals, at least by themselves. The second issue is their ability to minimise overall costs of achieving the agency's environmental goals.

The concern about meeting overall environmental goals, at least without additional instruments, arises because performance standards conventionally limit only one of two variables that determine ambient environmental conditions (*e.g.* Sterner, 2003). Performance standards limit the emissions for firms, but do not limit the number of polluting firms (Annex C). Thus, entry of new firms, even though they comply with performance standards, may degrade environmental quality. The implication is that overall environmental conditions cannot be managed by performance standards alone. Efficiency is enhanced if entry is also regulated. Alternatively, tighter standards might be required for new sources, highly restrictive standards might be imposed to diminish the impact of entry, or standards might be periodically revised. Each of these alternative measures poses economic and administrative burdens.

In addition to the entry problem, the concern for overall cost-effectiveness emerges due to the adverse selection problem discussed previously. Though performance standards result in farms minimising their individual compliance costs, this does not imply that the total cost of achieving the resulting environmental outcome is minimised. Landscape-level efficiency is attained by designing standards that differentially target producers according to farm-level differences in both compliance costs and environmental impacts. This means that standards must be set differentially, even in the simplest of cases.

For example, landscape-level efficiency is characterised by an adjusted form of the well-known equi-marginal principle for allocating pollution loads across sources: that each source should incur equivalent marginal (incremental) compliance costs per marginal unit of environmental impact (Annex C). The implication is that performance standards that minimise total compliance cost must be differentiated across firms according to their compliance costs and environmental impacts. But compliance costs are *private information* that is unknown to regulatory authorities. Thus, regardless of what regulators know about the relative impacts of farms on environmental conditions, they will lack information needed to design cost-effective performance standards. Thus, adverse selection prevents the implementation of allocatively efficient standards. Moreover, imposing differentiated standards to producers is likely to raise equity issues and would imply that reference levels are determined at sub-national levels. However the cost effectiveness of standards could still be improved by defining more stringent standards in specific vulnerable zones.

The cost-effectiveness properties of performance standards would be increased if the standards are applied differentially based on producers' individual environmental impacts, and if: a) producers are fairly homogeneous with respect to costs, so that environmental impacts are the only differentiated feature of concern, or b) costs and impacts are negatively correlated. The latter case implies that targeting high-impact producers with more stringent standards is more likely to also target low-cost producers, reducing environmental compliance costs.

### *Input standards*

Input standards (product or process) place mandates or constraints directly on producers' choices. Here the production process, technology, the products that are used, or the manner in which they are used, are regulated. Product standards are important to agriculture in many contexts, some to protect the environment, some to protect food safety, some to protect "brands". In the environmental context, the most important example is pesticide licensing and label requirements regulating uses and methods of use. For agriculture, process standards might consist of regulations pertaining to the ways producers manage their crops, livestock, and their land. Options might include regulations on input use (*e.g.* levels, timing and forms of agricultural chemical application) or the use of specific practices and technologies (*e.g.* erosion and runoff controls, irrigation equipment, and collection and land application of animal waste). Process standards relating to the management of animal wastes are used for large confined animal operations to protect air and water quality.

Input standards do not provide producers with the flexibility or incentives to look for cost-effective solutions to environmental problems.[3] This essentially shifts responsibility for cost-effectiveness from producers to the regulatory agency, as this responsibility becomes embedded in the agency's design and implementation of the standards. The concerns raised about performance standards above exist again, and are amplified.

With respect to environmental performance, input standards, like performance standards, generally provide incomplete control of environmental outcomes because they focus on individual farms rather than on the aggregate environmental performance of the set of farms. Thus the entry of new farms (or new cultivated land) may lead to environmental degradation, even though those farms are in compliance with the existing standards. Again, overall environmental conditions cannot be managed by input standards alone. Controls are also required to limit the number of polluters. But the problem may be more pronounced in this case. An "optimal" design standard will regulate all products and processes that affect a farm's performance with respect to an environmental goal. But some activities may be more difficult and costly to measure and monitor than others, leading to a focus on management of the more observable controls. This, in turn, can lead to substitution to unregulated activities which could have undesirable impacts (Eiswerth, 1993; Stephenson, Kerns, and Shabman, 1996). For example, a standard on herbicides would reduce herbicide use, but may increase mechanical cultivation and soil erosion, which in turn could have undesirable impacts on water quality. Addressing such problems will add layers of regulatory complexity and increase compliance costs.

With respect to cost-effectiveness, the adverse selection problem discussed in relation to performance standards is compounded. Again, farm-specific policies are required for allocative efficiency, but regulators do not have the private cost information needed to efficiently allocate environmental performance across farms and, once again, equity issues are likely to arise. But the problem is compounded because input standards typically eliminate the flexibility individual farms need to make environmental improvements at minimum costs. Thus, costs will not be minimised at the farm level or across farms.

Moreover, the combination of farm-specific and input-specific standards could greatly increase administration burdens and costs. In general, there is a trade-off between administration costs and cost-effectiveness. A national set of input standards that are easy to observe, to administer, and to enforce can incur low administration costs. Gathering information to better target where controls are applied and developing a broader set of input standards that apply to diverse conditions can significantly increase administration costs while improving cost-effectiveness.

Input standards might make the most sense when only a few choices are highly correlated with the agency's environmental goals and when the risk of environmental harm is great.

For instance, commonly-used input standards include pesticide use restrictions and bans, the design of animal waste storage lagoons for large concentrated animal feeding operations, and use of nutrient management practices in areas where drinking water is threatened by polluted runoff. In particular, a number of pesticides have been banned because of concerns over impacts on human health and the environment. The implication could be that the environmental and health costs of continued use so outweigh the benefits that a total ban is the only option. The case for input standards, as opposed to performance standards, is also stronger when the set of available performance metrics lacks adequate reliability or is too costly to use.

Some OECD countries (Australia and New Zealand) mostly rely on regulatory requirements to address environmental issues in agriculture. Besides regulations, specific environmental issues are addressed mainly through environmental programmes targeting specific areas. Any financial support is provided in the form of technical assistance and extension, with some of that support going to investments in infrastructure and investment on farms. Canada also uses extension and community-based measures and, more recently, on payments for specific farming practices. (OECD, 2008c).

## *Environmental taxes*

The goal of an environmental tax is to alter the economic incentives facing farmers so as to align their economic interests with societal objectives. Essentially, the mechanism is intended to correct the incentive failures resulting from missing markets for environmental goods by replacing missing price incentives with administered taxes or charges. For instance, taxes on pollution can be used to charge for pollution. It is, however, important to keep in mind that taxes also have an effect on the distribution of income. This effect can be an important consideration for the choice of policy instrument.

### *Performance taxes*

The incentive-based analogue to a performance standard is a tax applied to the corresponding performance indicator. Environmental taxes applied to negative externalities have long been advocated by economists as an efficient remedy for environmental externalities (Baumol and Oates, 1988). Like performance standards, performance taxes leave farms free to choose least-cost compliance strategies, resulting in farm-level cost-effectiveness. Performance taxes also offer even greater incentives than performance standards for new cost-saving technologies (Hanley *et al.*, 2007) and can offer potentially better opportunities for allocative efficiency and for controlling the number of farms.

Like performance standards, the first choice in the design of performance taxes is the particular indicator that will be taxed. Here the issues are the same as they were for performance indicator standards: using performance taxes requires consideration of the availability, reliability, and cost measuring farm-level environmental performance.

Given the availability of an acceptable indicator, the next task is to design the tax structure. Fundamental challenges here are to set the taxes so that environmental targets are achieved cost-effectively. The rule for cost-effectiveness is again the equi-marginal rule presented above. Economists have demonstrated that a least-cost allocation can be achieved by a differential tax structure with the differentials based on farms' relative environmental impacts (Annex C). The reason is that the responsibility to evaluate trade-offs between costs and impacts remains with farmers, who view the environmental impact-based tax rate as a price signal to guide their own decisions. Unlike performance standards, the regulator does not have to perform this evaluation for farmers, and so information about individual firms' compliance costs is not needed to achieve cost-effectiveness in this case. This property is considered a major advantage of environmental taxes over environmental standards. A second benefit of the differentiated tax structure is that taxes can effectively limit the number of farms, provided the absolute tax rates are adjusted to ensure the agency's environmental goals are met.

Things become more complex if the environmental model is more complex and includes non-linear environmental processes and interdependent impacts across agency goals and across producers (see Annex C for elaboration). But these complexities may not be as limiting as they were in the case of performance standards. The key is to target the taxes based on estimated environmental impacts. The taxes will then encourage producers to weigh the estimated impacts against their own costs, promoting cost-effectiveness. The same cannot be said of performance standards.

There is one caveat to the efficiency gain of performance taxes. While achieving a cost-effective outcome is simplified relative to the use of performance standards, achieving the desired overall environmental target is not. This is because the precise response of farms to taxes is unknown: taxes are expected to reduce emissions, but just how much an emissions reduction will be produced by a given tax structure cannot be known *ex ante,* and must be learned. This suggests an adaptive approach in which taxes are adjusted over time to achieve desired results (Baumol and Oates, 1988).

### *Input taxes*

An input-based tax increases the cost of implementing a practice having adverse environmental impacts (or, alternatively, it can be structured to reduce the cost of implementing environmentally-friendly practices). Input-based taxes, since they are not based on performance, do not encourage farm-level cost-effectiveness unless all relevant processes are taxed at the correct rates. The effect is to shift all responsibility for cost-effectiveness from producers to the regulatory agency. The ultimate effectiveness and efficiency of process-based taxes therefore depend on the two design decisions the agency must make with regards to these instruments: 1) which processes to tax? and 2) at what levels to set the taxes?

The choice of processes to tax involves the same issues that arise in the design of input-based standards: efficiency is enhanced when the agency taxes producer activities that are most highly correlated with the agency's environmental goals. A first-best design would require separate taxes to be applied to each environmentally-relevant activity, including those that are not currently being used. The alternative of placing standards on only the most easily observed activities can lead to substitution to unregulated activities which could have undesirable impacts (Eiswerth, 1993; Stephenson, Kerns, and Shabman, 1996). Efficiency is reduced as the set of taxed practices is reduced because more intense control will be required for the targeted set of practices to compensate for the absence of incentives for control of the non-targeted set. The determination of which practices are likely to be the best prospects for targeting of instruments will depend on the nature of any resulting substitution effects, correlation with environmental quality, and enforcement and monitoring costs.

Tax levels are the second design choice to make. Allocative efficiency requires each practice be taxed at a different level for each producer, so that each tax reflects the array of environmental impacts stemming from each producer's use of the practice. Accordingly, optimal input-based tax rates would be both farm-specific and input-specific. Failing to tailor tax rates in this manner reduces efficiency.

As with input-based standards, it is unlikely that regulatory agencies would have the required information to set the tax rates at the correct levels (and important equity issues again arise), and so efficiency losses are likely. Moreover, the combination of farm-specific and input-specific taxes could easily increase administration costs. The differentiated tax structure could also lead to arbitrage opportunities, as producers having low tax rates for inputs such as fertilisers or pesticides could sell these inputs to nearby producers saddled with higher tax rates for these products. Input-based taxes make the most sense when only a few choices are highly correlated with environmental outcomes and when producers are relatively homogeneous (so that tax rates can be set more uniformly). The case for input taxes, as opposed to performance taxes, is also stronger when the set of available performance metrics lacks adequate reliability or is too costly to use.

Environmental taxes and charges are applied in some OECD countries on the sale of inputs identified as having a potentially adverse impact on the environment. Taxes and charges are currently levied on pesticides in Denmark, France, Italy, Norway and Sweden, while fertiliser levies are applied in Italy, Sweden and some states of the United States (OECD, 2008c).

## *Permit trading*

There is now general agreement among economists that markets for regulating environmental externalities can often achieve environmental targets at lower social cost than traditional design and performance standards and environmental taxes. Indeed, success stories for air emissions trading in the United States, and for water quantity and fisheries quota trading (OECD, 2002) have spurred interest in expanding the scope of markets for environmental management. The most visible developments internationally are those addressing greenhouse gases (carbon trading).

Receiving less attention, but an area of substantial discussion, is water quality trading, including programmes to address agricultural sources of water pollution. Water quality trading frameworks are being widely and actively pursued in the US. Elsewhere, water trading frameworks have been developed for nutrients and salinity in Australia, for nutrients in Canada, and to manage animal wastes, which affect water and other environmental resources, in The Netherlands. Trading is also being considered for a range of other ecosystem services affected by agriculture. For simplicity, and because most trading programmes involving agriculture focus on water quality, the discussion here focuses on water quality markets involving point and nonpoint sources as the primary example. Market-based programmes addressing other types of externalities from agriculture would face similar challenges.

The fundamental economic appeal of trading is that it offers a mechanism for achieving a cost-effective allocation of environmental effort across alternative sources without environmental regulators knowing the abatement costs of individual agents. Thus, trading offers the promise of solving the adverse selection problem while also achieving environmental goals. Yet, for reasons developed above related to the complexity of agri-environmental problems, there are significant challenges to the design of trading systems that can realise these gains for agriculture. The national scale cap-and-trade markets that have been so successful for controlling air pollutants in the United States – which to some extent look like the textbook models and that have prompted interest in water quality trading – are simply not plausible models for water quality management. Fundamental requirements of cap-and-trade markets are that emissions: 1) can be accurately metered for each regulated emitter; 2) are substantially under the control of the emitter; and 3) the spatial location of emissions is not relevant to the attainment of the environmental target (*e.g.* Sterner, 2003). These requirements are not characteristic of water quality management problems. On the contrary, there is uncertainty about sources and levels of emissions, about the response of emissions to abatement effort, and about water quality impacts of emissions from different sources.

Water quality trading between point and nonpoint sources has been advocated as a way of jointly managing point and previously unregulated nonpoint sources on a watershed basis, and supplementing farm income by having point sources purchase pollution reductions from nonpoint sources (EPA, 2003). These benefits can only be realised if trades occur. Trading activity has been significant in the Australian salinity market, Dutch manure markets, and a few US programmes. But outside of a few cases, trade volumes have been low. The reasons can likely be traced back to market design. Polluters will only seek to trade if the gains from trade are sufficiently large to cover the transaction costs of searching for trading partners and entering into agreements.

The following statements characterise the challenges facing the development of water quality markets:

- Water quality markets that can cost-effectively and fully address the water quality risks from both point and nonpoint sources of water pollution will be complex, and perhaps impossibly so.
- Second-best water quality markets that can provide workable market places, achieve water quality targets, and perform well with respect to control costs will require information on point and nonpoint source pollution control costs in market design. This is information that economists and planners generally consider irrelevant to the design of markets, since markets are intended to optimally solve allocation problems without such information. Thus, the fundamental appeal of trading relative to environmental taxes or standards, as described above, is certainly diminished compared to the idealised model. How much so will be inherently an empirical issue requiring the comparison of specific alternative designs.
- There is a need for greater use of economics in water quality market design, and a need for research to advance both the science and policy of trading.

### *Designing water quality markets: basic design issues*

At the most basic level, the planner has three integrated tasks to develop a market for pollution trading that is consistent with the achievement of an environmental target. The focus here is on the limiting case where the market is designed to protect a specific water body.

The first task is to define the environmental output (either good or bad) as the "commodity" that will be traded in the market. One element of the definition is the specification of the observable indicator of environmental performance (*e.g.* nitrogen runoff) to which the rights pertain. The indicator must be observable so that trading is enforceable, and under the control of the polluter if the polluter is to be held responsible for non-compliance. Metered emissions are the conventionally-defined "commodity" for point sources of water pollution. However, the unobservable and stochastic nature of nonpoint emissions precludes the use of actual emissions as the nonpoint source "commodity". Some other observable construct must therefore serve as the basis for defining the tradeable nonpoint "commodity". As with other agri-environmental instruments, the nonpoint "commodity" can be defined based on an alternative measure of performance or based on input use.

Another element of the "commodity" definition is the nature of the rights conferred. Essential for the market to allocate pollution reductions efficiently across sources is the inclusion of the right to exchange the "commodity". Also essential for cost-minimisation is that the specification of rights allows polluters flexibility in how they meet performance requirements. Thus, efficiency gains may be reduced by placing restrictions on the technologies that may be used to satisfy a discharge allowance or to produce credits.[4]

The second task is to define specific rules for trading the "commodity" between alternative sources. Trading rules are intended to ensure that water quality outcomes resulting from particular "commodity" trades are at least equal to those that would occur without the trade. Such rules are needed to address the fact that pollution reductions from different sources may have different impacts on environmental quality. These differences may result from differences in location relative to water bodies, differences in the reliability of promised reductions from alternative sources, and other factors.

The third task is to limit (cap) the aggregate supply of the "commodities" such that feasible market allocations of polluting emissions, given the trading rules, do not violate the environmental target(s) and the defined rules for initial allocations. The specification of caps on the supply of commodities is an obvious requirement for markets to achieve water quality targets. Caps determine the post-trade level of water quality. In conjunction with the trading rules, caps determine the scarcity levels for the tradeable "commodities". Scarcity is a required element of any market, as it generates value and hence the impetus to trade. Initial allocation rules are crucial for the acceptability of developing such a trading scheme.

But, fully addressing these tasks does not guarantee the emergence of a market that can fully exploit potential gains from trade while ensuring that the environmental target is met. As in other types of trading, specific choices matter to the economic and environmental outcomes, as do a host of other non-trivial matters, such as fostering the development of market structures within which trades can efficiently occur, and monitoring and enforcement (Cason *et al.*, 2003; Woodward *et al.*, 2002; Woodward and Kaiser, 2002). Addressing these basic tasks poses significant challenges for the design of water quality trading programmes that address nonpoint pollution.

### *Performance-based trading*

Like performance standards and taxes, the first choice in the design of performance-based markets is the particular indicator that will serve as the tradeable nonpoint "commodity". Here the issues are the same as for performance standards and taxes: using a performance-based "commodity" requires consideration of the availability, reliability, and cost of models for measuring farm-level environmental performance.

One approach to defining the nonpoint "commodity" is to use estimated emissions reductions. This is the method of choice in US water quality trading programmes. To illustrate the approach, Pennsylvania's nutrient credit trading programme is designed to reduce nitrogen and phosphorous loads from agricultural nonpoint sources into the Chesapeake Bay. The state's department of environmental protection has developed a spreadsheet that farmers (or third-party agents) can use to calculate nitrogen or phosphorus reduction credits from the implementation of agricultural best-management practices (BMPs) from an approved list. The credits are estimates of the steady state annual average reduction in the levels delivered to the Bay from a farm. The spreadsheet uses estimates of the nitrogen reduction efficiencies of BMPs to calculate the reduction of nitrogen loads at the farm, and applies two factors from the US Environmental Protection Agency's Chesapeake Bay model to estimate the proportion of nutrients that move from a farm to the Bay.

There are two important aspects of this approach to note in comparison to actually trading metered emissions. One is that there is enormous uncertainty about the actual water quality outcomes of individual trades based on modelled emissions. This exists because the prediction errors for water quality models are known to be quite large.[5] A second is that the flexibility in choice of abatement methods that is fundamental to the case for emissions trading is not as great when trading estimated emissions generated from a list of approved technologies. The reason such lists are used is because research on the effects of alternative management practices on pollution loads is generally limited to a small set of known technologies relative to the domain of possibilities.

### *Input-based trading*

An alternative approach is to define the nonpoint "commodity" directly in terms of observable "up-stream" inputs or practices that affect nonpoint pollution flows. There are numerous choices that combine to control the distribution of nonpoint pollution loads from a given location. For example, nitrogen pollution from a farm will depend on the amounts, timing, and form of fertiliser or animal manure applied, the crops that are cultivated, tillage practices, and the use of conservation practices that intercept runoff. Conceptually, markets can be designed that target inputs, such as fertiliser, structural practices, such as buffer strips, and even technologies, like tillage practices, that affect nonpoint loads (Shortle and Abler, 1997; Shortle and Horan, 2001; Lankoski *et al.*, 2008a). There are no trading programmes of this complexity, but trading programmes that target up-stream inputs or practices are of interest. The Dutch manure quota is an example. In that case, negative air and water quality externalities were directly tied to massive volumes of animal manure from intensive livestock production systems relative to the adsorptive capacity of the environment, making reductions in the volume essential. Trading systems have been proposed in which point source emissions could be traded for reductions in the use of fertilisers and/or reductions of cropland in fertiliser-intensive uses or the establishment of buffer strips (Hanley *et al.*, 1997; Lankoski *et al.*, 2008a).

### *Trading rules*

Once the tradeable commodities have been determined, the next decision is the rate at which the trades should occur. Metered point source emissions are not perfect substitutes for estimated nonpoint emissions or changes in nonpoint input use. Generally, trade ratios are used to define the rate at which credits from one source may be exchanged for credits from another source. The main purpose of a trading ratio is to ensure that trades lead to equal or better water quality outcomes than would have occurred without the trade. In this use, trading ratios are essentially intended to translate multiple attribute emissions, varying in type, location, observability, and stochasticity, into a homogenous tradeable "commodity". One such adjustment is for differences in the location of discharges in a watershed on the ambient impacts of the discharge. Ratios used for this purpose are typically delivery ratios. A second common adjustment is to account for imperfect substitution between point and nonpoint emissions stemming from the relative uncertainty associated with point and nonpoint emissions.

The design of trading ratios to address nonpoint risk in emissions-for-estimated emissions trading has been a focus of economic research on point/nonpoint trading (Malik *et al.*, 1993; Horan, 2001; Horan *et al.*, 2004; Horan and Shortle, 2005). There have been three essential insights. One is that optimal trading ratios depend on other design parameters, such as baseline requirements and caps, and are thus optimally selected simultaneously with those parameters. A second is that optimal trading ratios are optimally differentiated across sources to address differences in relative risk. The third is that optimal ratios for managing nonpoint risk may be less than one.

The need to differentiate trading ratios by source implies that a first-best trading market requires source-specific trading ratios to convert each source's emissions, or estimated emissions, into a homogeneous "commodity" (Shortle and Horan, 2001). Such a market would be too complex to implement. Simpler constructs are used in practice. The simplest cases involve all trades among like sources occurring on a one-to-one basis and a single trading ratio defining all trades between point and nonpoint sources. Slightly more complex markets work the same way within a particular region or zone, but then apply additional trading ratios to adjust for trades between zones. In all of these second-best forms of permit markets, the problem of adverse selection again arises. This is because the planner will need to predict how the market will respond to various trading ratios, and accurate prediction requires knowledge of source-specific costs. The need to utilise firms' private information to set the market parameters undermines one of the key advantages of a tradeable permits market.

### *Multi-attribute trading*

A third approach to nonpoint "commodity" definition that has been recently suggested (Shortle and Horan, 2005), but not fully developed, is to define the nonpoint "commodity" as a multi-attribute good. The motivation for this proposal is that emissions-for-estimated emissions trading, especially with the uniform trading ratios typical of trading programmes, limits the degrees of freedom need to optimally manage nonpoint risk. This is because there are no incentives when trading average (mean) emissions to control the variability of emission unless the two happen to occur in fixed proportions, which is not the case. Trade ratios for addressing risk between sources would be contingent not only on the types of sources, point or nonpoint, but also possibly on the specific technologies used for pollution control since these would affect the distribution of emissions.

This chapter has focused on explaining the types of standards, taxes, and permits that are available for agri-environmental externalities, informational issues that arise in their application, and design factors that influence their performance. A fundamental conclusion that emerges from this analysis is that there is no one single instrument type or design that can promise to achieve agri-environmental policy goals, and to do so cost-effectively. This conclusion derives from the physical complexity of agriculture's impacts on environmental systems, uncertainty about key economic and environmental relationships affecting environmental and economic outcomes, and limited resources and capacities of environmental agencies. Political and equity considerations create additional complexity.

# Notes

1. Many of the basic results would also apply to standards on minimum provisions of environmental improvements and to payments/subsidies for environmental improvements. However, payments/subsidies can have different impacts on industry structure (numbers of firms and the scale of production) (Baumol and Oates, 1988), and payments/subsidies also pull funds from other socially beneficial programmes, which can create additional social costs (Alston and Hurd, 1990).

2. However, the definition of standards in terms of means and averages deserves much further analysis as it raises important questions. First, they would be at best difficult to explain to farmers and to monitor, so that transaction costs will probably be high. Second, their legal enforcement (in case of regulations) may be problematic, as sanctions usually require proof of harm done.

3. Indeed, process standards are also known as command-and-control standards because producers are told exactly what actions they must take to be in compliance.

4. This is all obvious to the economist and it would seem that little more needs to be said here. Yet, is interesting to note that participation in US water quality markets is voluntary, and that in existing programmes, non-point polluters become subject to limits, in the form of the baseline requirements mentioned above, only if they choose to participate.

5. Uncertainty about non-point pollution trading outcomes stem from the inherent stochastic features of non-point pollution and from the model uncertainty. Defining the non-point commodity in terms of annual average steady state loads essentially averages out variability, leaving model uncertainty.

# *Chapter 5*
# Design issues for agri-environmental payment programmes

A majority of OECD countries offer monetary payments to farmers to encourage them, on a voluntary basis, to implement more environmentally friendly farming practices going beyond those required by regulations, or defined as good farming practices. Most of these agri-environmental programmes offer a single, fixed payment for compliance with a pre-determined set of environmental requirements, such as reduced tillage or limits on the intensity and timing of fertiliser, manure and pesticide applications. The obvious problem with this type of fixed-rate payment approach is that heterogeneity in either farmers' compliance costs or site-productivity of environmental goods supplied are not taken into account in policy design and implementation. Thus, offering a fixed-rate payment under heterogeneous conditions could reduce the cost-effectiveness of agri-environmental payment programme.

Designing and implementing cost-effective agri-environmental payment programmes is difficult because of asymmetric information between a farmer and a policy maker. Information asymmetries exist if farmers have hidden information (or characteristics), which may lead to *adverse selection* in determining which farmers sign up to the programme, or hidden action, which may give rise to *moral hazard* in the compliance of farmers in the implementation of the programme. There are two mechanisms that address adverse selection and could improve the cost-effectiveness of agri-environmental payments relative to fixed payment approach: i) bidding mechanisms and ii) self-selection mechanisms. Moral hazard can be addressed through variables, such as intensity of compliance monitoring, level of fines/sanctions, observable compliance criteria, and level of payment.

This chapter first discusses informational asymmetries that are manifested in adverse selection and moral hazard in the context of agri-environmental payment programmes. This is followed by a short review of key policy design parameters for conservation auctions. Then general policy design parameters for agri-environmental payment programmes, including budget, eligibility, enrolment screens, participation incentives and implementation and enforcement costs, are discussed. The chapter ends with an overview of policy practice with regard to agri-environmental payments in OECD countries.

## *Informational asymmetries and agri-environmental payments*

Designing and implementing cost-effective agri-environmental payment programmes is difficult because of asymmetric information between a farmer and a policy maker or implementing agency. Farmers have an informational advantage with regard to their pre-contract farming practices and individual compliance costs and they may have incentives not to reveal this information to policy makers. In this context adverse selection means that farmers whose actions would most benefit the environment will not self-select for the programme. For example, farmers who already cultivate with low input use intensity have greater incentives to join an agri-environmental payment programme stipulating reduced input use intensity than farmers with higher input use intensity, since changes to current farming practices, and thus compliance costs, will be smaller for the former group of farmers. As a result of this adverse selection, the additional environmental gain from the programme could be small and the compliance costs of participating farmers could be over-compensated. If each farmer type could be observed then they could be paid differentially according to their compliance costs and agri-environmental budgets could be saved (Latacz-Lohmann and Schilizzi, 2005). However, such programmes, tailored to the differential circumstances of farmers, would require a lot of information (and thus high administrative costs) and choosing to give a lower (or no) payment to farmers who have voluntarily adopted improved practices is likely to raise equity issues, as they would consider themselves penalised compared to others. It would also discourage farmers to voluntarily improve their practices in the absence of economic incentives and thus have perverse effects, especially when ethical convictions and societal opinions are important drivers for the improvement of farm management practices (see Weinberg and Claassen, 2006).

### *Adverse selection and self-selection mechanisms*

Principal-agent models (self-selection mechanisms) are typically used to address the adverse selection problem in agriculture (*e.g.* Wu and Babcock, 1996; Moxey *et al.*, 1999). In these models policy makers devise different contracts for different types of farmers and tailor the contract so that each farmer type prefers the contract intended for that type. This form of contract is difficult to design because farmers have incentives not to reveal their type to policy makers, or to misrepresent their type in order to get a good combination of environmental requirements and payment rates. However, the so-called self-selection constraint ensures that farmers reveal their true type through the choice of contract and thus it reduces the information asymmetry between policy makers and farmers. Self-selection constraints require that farmers with certain characteristics (*e.g.* highly productive land) prefer the contract meant for that type over all other options offered and are supplemented with individual rationality constraints (participation constraint), which guarantee that farmers are at least as well-off when participating in the programme as not participating. In other words, farmers must at least be compensated for their compliance costs (Latacz-Lohmann and Schilizzi, 2005).

In the context of agri-environmental payments these types of self-selection mechanisms have been analysed (*e.g.* Wu and Babcock [1996], Moxey *et al.* [1999], and Glebe [2008]). However, despite these theoretical developments and some empirical applications, there has been no actual implementation of incentive-compatible contracts in agri-environmental policy making.[1]

## *Moral hazard and enforcement mechanisms*

The basis of the moral hazard is imperfect information about farmers' actual compliance with environmental requirements, but imperfect information about farmers' compliance costs will influence incentives to be non-compliant (cheat), because farmers with high compliance costs are more likely to cheat since their pay-off from cheating is higher than that of other farmers, whereas the penalty from getting caught in being non-compliant is likely to be constant (Latacz-Lohmann and Schilizzi, 2005). Hence, in the case of imperfect monitoring, farmers have an incentive to renege on their contracts so that they receive compensation payment without incurring the full compliance costs implied by their contract (Latacz-Lohmann and Schilizzi, 2005).

Latacz-Lohmann (1998) has developed a model to analyse incentives for cheating. His analysis shows that policy makers can manipulate four contract variables in order to prevent farmers cheating: i) the intensity of compliance monitoring (the probability of detection); ii) the level of fine/sanction for detected contract violations; iii) the stringency of environmental requirements and thus resulting compliance costs for farmers; and iv) the level of agri-environmental payments. The propensity to cheat is highest when compliance costs are high relative to payment level. Latacz-Lohmann (1998) suggests that over-compensation (level of payment relative to compliance costs) can reduce incentives for cheating and thus the need for compliance monitoring. Moreover, compliance monitoring can be targeted to high-cost farmers.

Analysis by Lankoski *et al.* (2008b) confirms that the level of payment (and thus the penalty from cheating) relative to additional gains from cheating plays a crucial role. Their analysis is based on Finnish data and they show that because the agri-environmental payment level is high relative to income from production – and thus the additional gain from cheating with a fertiliser application limit – the optimal monitoring rate is very low. Optimal monitoring rates vary according to land productivity, but are less than 3% even for the highest land productivities that reflect highest country level average yields in the European Union.

The problem of moral hazard in the context of agri-environmental policy has also been analysed by Choe and Fraser, 1999; Ozanne *et al.*, 2001; Kampas and White, 2004; and Fraser, 2002. Choe and Fraser (1999) derive optimal monitoring strategies and incentive payments when farmers can exert either low or high compliance effort and monitoring is costly. Kampas and White (2004) examine the impacts of monitoring costs on the relative efficiency of alternative agri-environmental policy mechanisms. Fraser (2002) investigates the effects of penalties for non-compliance but does not consider monitoring costs. Ozanne *et al.* (2001) show that high degrees of farmer risk aversion will reduce the severity of the moral hazard problem. When farmers are risk averse, low penalties on groups targeted for frequent monitoring and high penalties on other groups should ensure that compliance at low cost or monitoring efforts can be targeted based on past performance (Fraser and Fraser, 2005). Naturally, enforcement costs can be reduced by new information technologies, such as Geographical Information Systems (GIS).

### *Auction mechanisms*

Auction theory provides an interesting way to extend the principal-agent approach by incorporating competition between farmers for winning a contract with the policy maker. Auctions have been recently applied to environmental conservation in agriculture (Latacz-Lohmann and Hamsvoort 1997, Stoneham *et al.*, 2003, Vukina *et al.*, 2006). Conservation auctions are those auctions in which farmers bid competitively for a limited number of environmental conservation contracts. When making a bid a farmer faces a trade-off between net pay-offs and acceptance probability so that a higher bid increases the net pay-off but reduces the probability of getting a bid accepted. Thus, competitive bidding can push farmers to reveal their self-estimated compliance costs and as a result it will reduce farmers' information rents and improve the cost-effectiveness of an agri-environmental programme. However, this improvement will depend on the accuracy of the compliance cost assessment by farmers and on the implementation of the auction mechanism. For example, farmers can still benefit from information asymmetries when providing information to assess the initial state of the environment and of their practices, and there will be efficiency losses if they fail to assess correctly their compliance costs.

Primary reasons to implement conservation auctions are to improve both allocative efficiency (bids with highest benefit-cost ratio are selected for the programme) and budgetary cost-effectiveness (maximise environmental benefits with a given fixed budget). Latacz-Lohmann and Schilizzi (2005) provide detailed discussion and guidance on how to design and implement conservation auctions and the following paragraphs draw on their research.

As regards policy design for conservation auctions the first choice to be made is between different *payment formats*. In the discriminatory format each bidder is paid according to his or her winning bid. In the uniform-price auction all those who are successful receive the same cut-off price, which is either the highest accepted or the lowest rejected bid. Different payment formats affect farmers' bidding behaviour. Under the discriminatory format the farmer's bid not only depends on the farmer's compliance cost, but also his/her best guess with regard to the highest acceptable bid. Thus, there is incentive for a farmer to bid above his/her compliance costs and thus secure an information rent. This incentive is higher for those farmers with low compliance costs than those with high compliance costs. Thus, under the discriminatory format the dominant strategy for a farmer is that of overbidding, and thus this type of payment format does not reveal farmers' self-estimated compliance costs. Under uniform pricing the farmer's dominant strategy is to bid his/her estimated compliance costs, because the bid only determines their chance of getting into the programme but not the payment level. Thus, this payment format reveals these compliance costs (Latacz-Lohmann and Schilizzi, 2005), although farmers will be over-compensated.

Another key auction design parameter is so-called *reserve price*, which is the upper limit of payment per unit of conservation benefit, which can be pre-announced or not. A reserve price increases bidding competition and thus reduces farmers' information rents, but also provides the signal of maximum willingness to pay for farmers' provision of conservation services or environmental goods. Pre-announcing a reserve price may create problems in repeated auctions (*e.g.* the US Conservation Reserve Program [CRP]), where farmers learn the level of the reserve price and offer their bids at the reserve price (Reichelderfer and Boggess, 1998; Latacz-Lohmann and Schilizzi, 2005).

Auctions can be implemented with a fixed budget or with a fixed target. Fixed target auctions set a target for a programme (*e.g.* hectares of wetlands conserved) and bids are accepted from farmers until the target is reached. This means that the budget of the programme is open until the auction is completed. Fixed target auctions can be used when policy makers must meet environmental objectives (Latacz-Lohmann and Schilizzi, 2005). However, the fixed budget is a common form in that bids are accepted on the basis of benefit to cost (bid) ratios until the pre-determined fixed budget is exhausted. Thus, fixed target auctions may be more appealing to environmental agencies, whereas fixed budget auctions may appeal to finance ministries.

Bidder learning is a real problem with repeated conservation auctions. Thus, policy makers should not publish information about highest or average acceptable bids or distribution of bids received in the previous bidding rounds. This problem can be reduced by changing the rules of auction in each bidding round in order to create uncertainty among bidders (Latacz-Lohmann and Schilizzi, 2005).

One of the key design issues in conservation auctions is the bid evaluation system (enrolment screen). Because farmers' environmental practices usually provide multiple benefits a multi-criteria bid scoring system could be adopted to aggregate the overall environmental benefit of bid. These types of multi-criteria bid scoring systems are, for example, the environmental benefit index (EBI) for the US CRP and the biodiversity quality index (BQ) used in the BushTender programme in Australia.

Although conservation auctions provide an innovative and promising agri-environmental policy approach, there are some important caveats that need to be considered before full-scale conservation auctions are implemented. The main disadvantages raised by various authors include the likely higher policy-related transaction costs (PRTCs) for both governments and farmers (OECD, 2007c). Higher transaction costs for farmers might reduce the number of applicants, which reduces the competition and may in turn reduce the cost-effectiveness of auction. However, Heimlich (2005) estimated the transaction costs of the CRP at 3% of expenditures in initial years and only 1% in succeeding years.

Also, strategic bidding behaviour, as well as collusion, reduces the cost-effectiveness of auctions. A general understanding is that auctions are not suitable for small scale, local environmental goods and services, since the smaller the number of potential bidders the lower is the bidding competition and the higher is the risk of collusion and strategic bidding (OECD, 2007c). Moreover, single round bidding is preferred to multiple rounds, since cost-effectiveness gains from one-shot auctions are eroded under dynamic settings as winning bidders learn the reserve/cut-off price (*e.g.* Stoneham *et al.* [2004]; Latacz-Lohmann and Schilizzi [2005]; Hailu and Schilizzi [2004]). Changing the weights of multi-criteria bid scoring between different rounds should, however, reduce such strategic bidding behaviour.

Cost-effectiveness gains from conservation auctions vary significantly. In addition to the CRP, conservation auctions have been used in Australia where two such auctions include the BushTender programme, which has been analysed by Stoneham *et al.* (2003), and the World Wildlife Fund auction (Auction for Landscape Recovery), which has been analysed by White and Burton (2005). Stoneham *et al.* (2003) and White and Burton (2005) find considerable cost-effectiveness gains from auctions relative to fixed uniform payments. Stoneham *et al.* report cost-effectiveness gains from auctions to be 700% – that is, conservation auctions provided seven times more biodiversity benefits in the first-round, compared to a fixed-price scheme with the same budget. However, transaction costs of conservation auction were estimated to be 50-60% of expenditure in the first round. White and Burton report the cost-effectiveness gains to be between 200 and 315% in the first round. Latacz-Lohmann and van der Hamsvoort (1997) found cost-effectiveness gains to be in the range from 16-29%. Connor *et al.* (2008) use data from the Catchment Care auction (in the Onkaparinga catchment in South Australia), which was a sealed bid, first-price, discriminating price auction and simulate various alternative auctions, differentiated payments and fixed payment policies. Their results show that with same budget a uniform fixed payment achieves 56% of the estimated environmental benefit obtained with auctions. They show that cost-effectiveness gains from auctions come through the EBI bid prioritisation rather than through the reduction of information rents.

Since empirical evidence regarding auction performance is still inconclusive, experimental laboratory auctions with stakeholders and reduced-scale field pilots need to be carried out first before implementing full-scale conservation auctions (Latacz-Lohmann and Schilizzi, 2005).

## *General policy design and implementation parameters*

The environmental and economic performance of agri-environmental payment programmes depend critically on several policy design parameters that affect which farmers will apply and which applications are accepted. Design of cost-effective agri-environmental payment programmes requires: i) the identification of those farmers, land parcels, and practices that are most likely to achieve programme objectives with the least cost; and ii) formulation of eligibility criteria, payment incentives, and enrolment screening so that the "right" farmers apply (Cattaneo *et al.*, 2005). Thus, policy makers have a number of design parameters available to attract the right participants; those farmers who can make the most valuable contribution to achieving programme objectives. These agri-environmental payment programme design parameters including budget, eligibility criteria, enrolment screens, payment incentives or payment type, and administrative costs and compliance monitoring are reviewed in detail below. It should be noted that agricultural extension, provision of information and communication are also important design parameters.

### *Programme budget*

When agri-environmental expenditure is limited by the budget (which is not always the case) then an alternative definition of cost-effectiveness – the so-called budgetary cost-effectiveness – can be adopted in which environmental benefits should be maximised within a given budget (see Cattaneo *et al.* 2005). Budgetary cost-effectiveness is not a precise mirror image of standard cost-effectiveness, since maximising environmental benefits subject to budget constraints requires that both farmers' compliance costs and transfer payments to farmers should be minimised. Thus, in contrast to the standard cost-effectiveness criterion the payments to farmers are an issue in budgetary cost-effectiveness, since payments reduce the available budget to further increasing environmental benefits (Cattaneo *et al.*, 2005). Competitive bidding on agri-environmental payments usually helps to stretch a limited budget to achieve more environmental benefits than is possible by using a fixed payment. The cost-effectiveness of bidding improves when bid prioritisation relies on measurement of environmental benefits.

### *Eligibility criteria*

Eligibility determines who can apply for enrolment and what practices they can use. Eligibility criteria can be used to focus programme implementation to those farmers, land parcels, and practices that are most likely produce environmental benefits in a cost-effective manner. Eligibility criteria can be narrow or broad and can be based on a wide range of factors, such as farm type (*e.g.* livestock or crop farms), land characteristics (*e.g.* slope and erosion risk), land use and land cover (*e.g.* cereals or grassland), practices (*e.g.* nutrient management and reduced tillage), and geographical location. Broad eligibility will yield a large pool of applicants which can then be narrowed by using different types of enrolment screens (Cattaneo *et al.*, 2005). The broad pool of applicants may be good from a cost-effectiveness viewpoint, because it will more likely attract those farmers who can make the most valuable contribution to achieving programme objectives. However, with a given budget a broad pool means that a higher number of applications will be rejected, which may affect farmers' incentives to apply in the future for agri-environmental programmes (Cattaneo *et al.*, 2005).

### *Enrolment screens and payment incentives*

If eligibility criteria are broad then the policy maker has to use an enrolment screen to select the right participants, for example on the basis of benefit-cost ratio of applications. These enrolment screens can be performance-based, cost-based or bid-based.

In performance-based screening farmers are paid according to measured environmental performance or benefits generated (*e.g.* by using proxies, such as the EBI). If they are paid according to their performance, then even the farmers who have already reached the given performance will be included, but they would be excluded if the payment is made only according to improvements made. In the latter case there should be more environmental benefits for the same budget, but transaction costs are likely to be higher (given the need to collect data on historical practices), some producers may abandon good practice to become eligible, and the programmes may be viewed as inequitable (Weinberg, 2006).

Cost-based screening refers to a fixed payment (EUR/USD/hectare) and it can be proportional to actual compliance cost (so called cost-share programmes) or be based on estimated compliance costs (*e.g.* average compliance cost related to nutrient management or the establishment of buffer strips). This type of screening can produce a cost-effective outcome only in the case where environmental benefits and compliance costs are negatively and highly correlated (Babcock *et al.* [1997]; Wu *et al.* [2001]; Cattaneo *et al.* [2005]).

Bid-based screening produces a cost-effective outcome when used jointly with a performance based screen. For example, in the US CRP, the combination of performance screening through the EBI and competitive bidding is used to select CRP participants. This benefit-cost targeting allows policy makers to rank and select participants on the basis of the benefit-cost ratio of their bids (where the EBI represents the benefit and farmer's bid represents cost).[2] If bidding is competitive it will push farmers to reveal their estimated compliance costs and as a result will reduce farmers' information rents and improve the cost-effectiveness of an agri-environmental programme. Thus, competitive bidding maximises environmental benefits per programme payment but, as it only covers compliance costs, it does little to support farm incomes (Cattaneo *et al.*, 2005).

Hence, enrolment screens and payment incentives work in combination to select the right participants and this is especially case when eligibility criteria are broad.

### *Administrative costs (or policy-related transaction costs) and compliance monitoring*

Although benefit-cost targeting through environmental performance screening combined with competitive bidding or with differentiated payments is likely to yield higher budgetary cost-effectiveness than fixed-rate payment approaches, the gains from targeting need to be weighed against the potential increase in the administrative costs of the programme, losses of ancillary benefits (dissociation costs) and equity considerations. Improving the precision of the policy instrument increases policy-related transaction costs as a percentage of payments (Vatn, 2002). However, as shown in OECD (2007b) although the PRTCs of targeted payments can be higher as a percentage of transfers than those of untargeted payments, total PRTCs are not necessarily higher and, in many cases, the total costs of achieving a desired policy outcome could be lower for well-targeted payments.

The policy maker has several variables available to ensure compliance, such as the intensity of compliance monitoring, the probability of detection, and the level of penalty for detected non-compliance. Moreover, when different types of practices are selected as compliance bases the policy maker needs to consider how easily these can be monitored and farmer’s compliance observed. For example, the establishment and management of buffer strips are easily observable, whereas timing and amount of per hectare of nutrient or pesticide application is extremely difficult and costly to monitor (Johansson, 2002).

### *Cost-effectiveness of performance-based versus practice-based programmes*

Cattaneo *et al.* (2005) used empirical simulation models to assess how alternative working land (that is, cultivated land) agri-environmental payment programme designs affect farmers’ profits, consumer welfare and environmental performance. They developed an aggregate environmental index (AEI), which is similar to the EBI used by the US Department of Agriculture to rank CRP contracts, for assessing environmental performance. Alternative programme designs analysed are: i) different types of practice-based policies, in which payments are fixed-rate incentive payments for producers who implement eligible environmental practices;

and ii) different types of performance-based policies that use either performance-based payments or bid-based payments in conjunction with performance-based screens. Their results show that for a given budget the performance-based payments and bid-based payments achieve much higher environmental performance than practice-based payments. By keeping the budget at USD 1 billion, the performance-based programme with bidding could improve environmental performance by more than 15% over current production patterns, whereas a performance-based payment without bidding could improve environmental performance by 12%, and a practice-based payment[3] by only 1%. When programme cost is measured per aggregate environmental point (given by the index) the performance-based programme with bidding achieves environmental improvements at an average cost of USD 6 per aggregate point, performance-based payment without bidding at an average cost of USD 8 and a practice-based payment at USD 17 (for new practice adoption) and USD 73 (when also on-going practices are rewarded in addition to new practice adoption).

Weinberg and Claassen (2006) use the same simulation framework and conclude that the same environmental benefits obtained with a USD 1 billion practice-based programme could be achieved for only a USD 200 million budget by implementing a performance-based programme. This result is explained by two important factors for cost-effective policy design: environmental heterogeneity and flexibility. Performance-based payments allocate payments to locations that provide the highest environmental performance gains and they provide the flexibility to producers to tailor their environmental management to their own resource setting. It should be noted that due to scarce data the administrative costs or PRTCs of practice-based and performance-based programmes were not taken into account in this analysis.

# Notes

1. For discussion on the actual implementability of these types of incentive-compatible contracts, see, for example, Lichtenberg (2002) and Latacz-Lohmann and Schilizzi (2005).

2. Note that in the case of CRP the overall EBI score combines both environmental benefits and cost of contract (bid). This same applies for “offer indices” used in the Environmental Quality Incentives Program (EQIP). However, for clarity of discussion related to benefits and costs in this chapter, these two are separated.

3. In this chapter, the notation used by Cattaneo *et al.* is adopted. However, practice-based programmes in this study are expressed as "input-based".

# *Chapter 6*
# Policy-mixes for the agri-environment: overview of design parameters

The OECD study on *Instrument Mixes for Environmental Policy* (2007d) provides a comprehensive treatment of the economic efficiency and environmental effectiveness of using an instrument-mix rather than a single policy instrument. The main arguments for using instrument-mixes are: i) many environmental issues are multifaceted so that not only the amount of emissions, but also where emissions take place and when they occur are relevant; ii) many instruments can mutually strengthen each other; and iii) sometimes instrument-mixes can also enhance enforcement and reduce policy related transaction costs. However, there are also reasons for restricting the number of instruments in the mix. For example, when several instruments are applied in the mix there could be danger that one instrument hampers flexibility to find low-cost solutions to a problem that another instrument could have offered if it had been implemented on its own. And there are cases where some of the instruments in a mix are redundant and only increase total PRTCs (OECD, 2007d).

This chapter firstly provides a short overview of instrument-mixes addressing agricultural nonpoint source pollution. This is followed by an overview of general policy design aspects for environmental cross-compliance. Both of these topics have been analysed extensively in the OECD.

## *Instrument-mixes addressing agricultural nonpoint source pollution*

Since no single policy instrument analysed so far is likely to be unambiguously preferred over all available instruments in all conditions, the optimal strategy may involve the use of a mix of policy instruments. For example, economic instruments could be used together with regulations or with information instruments, or with other economic instruments (Weersink *et al.*, 1998). According to Braden and Segerson (1993), the information problems relating to nonpoint source pollution control suggest that no single instrument is likely to achieve a cost-effective outcome. Therefore, it may be preferable to use a combination of instruments for controlling nonpoint source pollution (NPSP). Thus, information problems inherent in NPSP control provide a theoretical rationale for the combination of instruments when single policy instruments are inefficient.

Theoretical research on instrument-mixes is sparse; exceptions are Braden and Segerson (1993) and Shortle and Abler (1994). Braden and Segerson (1993) analyse the simultaneous use of multiple instruments as a means to compensate for imperfect information in the case of NPSP. They show that multiple instruments may be more efficient than single instruments in the case of imperfect information. However, the efficiency of combined instruments depends on the way the pollution-related inputs interact with each other in the production and pollution process. To be efficient, instruments have to complement, not contradict, each other.

Shortle and Abler (1994) analyse a mixed-instrument scheme consisting of taxes, subsidies and permits for the use of polluting inputs. Farms need to hold permits for the use of polluting inputs and these permits can be traded. A farmer pays (receives) a tax (subsidy) if he/she uses more (less) inputs than allowed by the permits. Shortle and Abler show that a mixed system economises on information costs compared to firm-specific non-linear input taxes when effluents are stochastic (weather dependent) and unobservable.

According to Segerson (1990), the best policy approach to control agricultural pollution may involve the use of several instruments based on both incentives and regulation. The choice of the specific instruments to be used requires the balancing of multiple objectives relating to criteria such as cost-effectiveness, environmental effectiveness and administrative practicability. A policy-mix that attempts to balance these concerns will be imperfect in terms of any single criterion. It should be evaluated, however, as a compromise solution to environmental problems that lack an easy solution.

OECD (2007d) provides an overview of policy objectives and policy instruments addressing nutrient and pesticide runoff on the basis of information received from OECD countries in 2004. Altogether 93 national policy objectives were singled out in the responses, of which 44 concern nutrient runoff, 35 pesticide runoff, and 14 address both issues. Altogether 346 policy instruments were identified in the responses of which 198 address nutrient runoff, 119 pesticide runoff, and 29 address both issues. As regards nutrient runoff and pesticide use, regulatory instruments are most common in member countries that responded. Eighty-two regulatory instruments were mentioned to address nutrient runoff and 57 pesticide use. Economic instruments were the most common way to address both issues.

Also, information instruments are common, since 39 information instruments addressed pesticide use and 43 addressed nutrient runoff. With regard to economic instruments various types of subsidies dominate. Overall, taxes and charges play a minor role and thus there has been little emphasis on making polluters pay when governments have addressed agricultural nonpoint source pollution.

OECD (2007d) analyses instrument-mixes for controlling nonpoint sources of water pollution in agriculture in four countries: Denmark, The Netherlands, the United Kingdom and the Chesapeake Bay area in the United States. For Denmark, separate studies were made on instrument-mixes addressing nitrogen runoff, phosphorus runoff and pesticide use. Policy-mixes addressing both nutrient and pesticide runoff were analysed in the case of the United Kingdom, while the case studies of The Netherlands and the US focused on nutrient runoff. As regards the case study countries, the instrument-mixes have brought about significant environmental improvements in Denmark and in The Netherlands by the reduction in nitrogen surpluses expressed per hectare according to the OECD nitrogen balance database (from 1985 to 2004). In these two countries phosphorus balances per hectare also declined from 1985 to 2004. Moreover, pesticide use declined more in Denmark (in which a partly tax-based system was used) than in the United Kingdom, where the system relied more on voluntary-based approach.

## *Environmental cross compliance*[1]

The rationale for environmental cross compliance (henceforth termed cross compliance) in OECD countries involves at least three related elements: income payments to farmers may appear more acceptable to society when they must meet environmental requirements; leveraging or linking income support payments can better ensure compliance with environmental requirements; and policy-related transactions costs can be reduced. While the term cross compliance indicates that a number of policy instruments are linked, there is no unique approach to cross compliance implemented in some OECD countries. However, at least two necessary conditions for any cross-compliance mechanism are: there is a system of income support payments in place that can be leveraged with respect to specific farmers meeting environmental requirements as it is not possible to link across-the-board market price support instruments (such as border measures) to meeting environmental requirements (except in so far as that would apply uniformly to all farmers); and there are explicit or implicit "reference levels", which define the respective responsibilities of farmers and society in providing environmental services and thus the allocation of the costs of such improvement between farmers and society (through policy instruments).

Even if cross-compliance approaches are effective in achieving policy goals there may be other more cost-effective ways to do so where the primary objective of the support payment is compliance with environmental standards. These could include, for example, environmental regulations and associated penalties and charges that apply to all farmers irrespective of whether they receive other support payments; agricultural income support payments that apply to all farmers or a targeted group of farmers; and agri-environmental payments targeted to those eligible farmers that provide environmental services that go beyond what society expects of them.

Cross-compliance requirements provide a link between one or more policy instruments such that farmers are required to fulfil specified conditions in order to be eligible to receive an agricultural support or payments. In all countries implementing cross compliance, a link is made between two or more policy measures: in the case of the European Union (Annex B) and Switzerland, non-compliance of (mandatory) environmental regulations by farmers leaves them liable to lose agricultural support payments; in other countries, such as the United States, where the primary objective is farm income support, eligibility of payments depends on farmers meeting various environmental performance or practice conditions. In the case of an agri-environmental payment the primary objective is to achieve a given level of environmental performance, to which eligibility for payments depends on farmers voluntarily meeting specified conditions.

Cross-compliance requirements – by linking the respect of environmental conditions or regulations to the granting of agricultural support payments – have the potential to contribute to improving environmental performance of agriculture compared to a situation where the same level and structure of payments are made without any conditions attached. However, the comparison between different cross-compliance approaches or between cross compliance and other approaches and policy-mixes to achieve farm income and environmental objectives is an empirical question and is dependent on the baseline chosen for making such comparisons. Such an evaluation has not yet been undertaken in the OECD.

A study on environmental cross compliance in agriculture (OECD, 2010a) examines policy options to provide income support and to improve environmental performance. These options can be viewed as forming a policy continuum along which environmental objectives become increasingly dominant at the expense of other objectives related to transferring income to farmers. A move along this continuum involves closer targeting of environmental outcomes, and thus environmental effectiveness, but may involve potential efficiency losses with regard to other objectives of support. Efficiency and cost-effectiveness of five stylised programmes were analysed, including both cross compliance and agri-environmental payments. The programmes include mandatory cross compliance, voluntary cross compliance, voluntary cross compliance with environmental targeting, auctions based on compliance costs, and auctions based on environmental benefits. Analysis of agriculture, trade and environment in the arable crop sector (OECD, 2005a) shows that the cost-effectiveness of cross compliance is high when measured relative to the incremental cost of cross compliance and thus environmental gains are secured at low additional costs. However, when cross-compliance requirements are set to achieve significant environmental improvements, then some producers would either suffer income loss (when remaining in the programme is compulsory) or leave the programme in the case of voluntary participation. Moreover, improving the environmental performance of cross compliance usually requires better targeting of producers and environmental objectives so that the income support objective may become subordinated.

Environmental cross-compliance measures have been implemented in several OECD countries including EU countries, Norway, the United States and Switzerland and, more recently, also in Korea.

Box 6.1 provides a checklist of criteria to weigh up the potential advantages and disadvantages of cross-compliance approaches.

**Box 6.1. Checklist of criteria to weigh up advantages or disadvantages of cross-compliance approaches**

***Policy coherence*:**

- Greater synergies between agricultural and environmental policies;
- Public acceptance of agricultural income support payments to farmers though meeting environmental requirements;
- Further reform of agricultural policies, when such reforms are dependent on meeting environmental standards.

***Farmer involvement:***

- Inclusion of producers who would otherwise not enrol on a voluntary basis;
- Uptake of voluntary agri-environmental programmes that involve stricter conformity requirements and better legal compliance;
- Perception by farmers of compensation for producing environmental benefits, depending on whether farmers are able to perceive a link between compliance and receipt of payments.

***Agri-environmental performance:***

- Application of the Polluter-Pays-Principle in agriculture;
- Awareness of farmers of the consequences of their actions on the environment, in particular if cross compliance is made legally binding;
- Leverage on farmers through the provision of payments (or the risk of their withdrawal) to conform with existing legislation and codes of practice, in situations where codes of practice form part of the cross-compliance conditions;
- The number of producers who are not eligible for agricultural support payments who implement environmentally beneficial practices;
- Ability to meet minimum environmental standards without any additional payment where the standards define the baseline for agri-environmental policy measures;
- Balance in environmental obligations in the case where the environmental obligations linked to cross compliance go further than the regulations, if some sectors receive agricultural support payments and others do not;
- Certainty of environmental outcomes if cross-compliance measures are more general and less targeted to the situation on each farm;
- Environmental performance if agricultural support payments are counter-cyclical, given that there is inverse relationship between economic and environmental incentives;
- Environmental performance if there are homogeneous requirements across all farmers, yet individual farmers have different compliance costs.

***Transaction costs:***

- Potential to economise in administrative and policy transaction costs compared to the separate administration of agricultural income support, environmental regulations and agri-environmental payments to ensure a given level of environmental quality;
- Monitoring costs where cross-compliance measures are targeted closely to the situation on each farm, although administrative and monitoring costs could be lower where there are sector-wide measures;
- Incentive for environmental improvement from financial penalties for non-compliance if compliance conditions are not part of statutory requirements;
- Administrative and monitoring costs if cross-compliance conditions take heterogeneous compliance costs into account.

*Source: Environmental Cross Compliance in Agriculture* (OECD, 2010a).

# Note

1. This chapter is based on *Environmental Cross Compliance in Agriculture* (OECD, 2010a).

# *Chapter 7*
# Agri-environmental policies in OECD countries[1]

The toolbox of agri-environmental policy instruments applied in OECD countries to achieve their various environmental objectives reflect several issues including: i) the overall policy approach to the sector; ii) the specific environmental issues and their perceived linkage to agricultural activities; iii) the nature of property rights related to the use of natural resources (land, water and vegetation); and iv) societal concerns related to environmental issues. In addition, "suasive" measures are intended to change perceptions and priorities within the farmer's decision framework by heightening the level of environmental awareness and responsibility.

***Environmental regulations (regulatory requirements)*** are at the core of policies addressing environmental issues in agriculture. All OECD countries pursue policy and/or regulatory measures to prevent the negative impact of agriculture on the environment. Most of these regulations are related to the use (storage, handling, plant and animal application) of agricultural inputs (pesticides, chemical fertilisers, manure) which have the potential to cause negative environmental effects (in terms of soil, water and air pollution). These regulatory requirements range from outright prohibitions, to input standards and resource-use requirements. Most of these regulations are applied across the farm sector. However, in areas with higher environmental values (natural reserves), drinking water catchment areas, environmentally sensitive areas, or those close to densely populated areas, further regulations may be applied. Over time, these regulatory requirements have generally been applied more broadly, and as awareness of the risks developed, they have become more stringent.

On the basis of the OECD web-based *Inventory of Policy Measures Addressing Environmental Issues in Agriculture,* the country chapters in the agri-environmental indicators report (OECD, 2008a), and the Database on instruments used for environmental policy, Table 7.1 summarises in broad terms the main types of policy instruments used in OECD countries (OECD, 2009).

## Table 7.1. Measures addressing environmental issues in agriculture in OECD countries

| Measure/ Country | AUS | CAN | EU | JPN | KOR | MEX | NZL | NOR | CHE | TUR | US |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Regulatory Requirements** | XXX | XX | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX |
| **Environmental cross-compliance** | NA | NA | XXX | X | X | NA | NA | XX | XXX | NA | XXX |
| **Payments based on farming practices** | X | X | XXX | X | X | X | X | XX | XXX | X | XX |
| **Payments based on land retirement** | NA | NA | X | NA | NA | X | NA | NA | X | NA | XXX |
| **Payment based on farm fixed assets** | X | X | X | X | X | X | X | X | X | X | X |
| **Environmental taxes/ charges** | NA | NA | X | NA | NA | NA | NA | X | NA | NA | X |
| **Tradeable rights/ permits** | X | NA | X | NA | NA | NA | NA | NA | NA | NA | X |
| **Technical assistance/ extension** | XX | XX | X | X | X | X | XX | X | X | X | XX |
| **Community-based measures** | X | X | NA | NA | NA | NA | X | NA | NA | NA | NA |

NA: Not applied or marginal; X: Low importance; XX: Medium importance; XXX: High importance.

*Note:* The importance of the policy instruments in this table is related to the mix in the specific country. It is not designed to compare the importance of specific measures across countries.

*Source:* OECD (2010b).

Some OECD countries (Australia, New Zealand) rely mostly on regulatory requirements to address environmental issues in agriculture. Besides the regulations, specific environmental issues are addressed mainly through environmental programmes targeting specific areas. In many cases farmers and landowners (grouped in local initiatives) are involved in these programmes, which may be supported by short-term financial assistance to facilitate group activities improving environmental sustainability and self-reliance of the agricultural sector. Financial support may also be provided in the form of *technical assistance and extension*, with some support going to investments in infrastructure and on-farm investments. Besides regulatory requirements, Canada also relies mainly on *extension and community-based measures* and more recently on rather limited payments for specific farming practices.

Other countries (mostly EU countries, Norway, Switzerland and the United States) have also developed – in addition to environmental regulations – a wide range of voluntary programmes providing incentives (payments) to farmers to adopt specific farming practices with positive environmental effects and/or providing public goods (such as landscape, biodiversity, etc.). Although these programmes offer a large variety of measures, most of the payments are related to the support of extensive forms of farming (mostly on grassland – extensive management of grassland, extensive pastures). Such programmes exist in all countries and represent the most important part of spending on agri-environmental programmes. In Japan and Korea, agri-environmental payments have only been introduced recently and they represent a very minor share in the total support to agriculture.

Programmes providing *payments for retirement of agricultural land from production* are also implemented in a range of countries (European countries and the United States). These programmes mainly provide payments for conversion of agricultural land to wetlands or forest. However, in most countries these programmes have a rather limited importance, with the exception of the United States, where payments for retirement of agricultural land (CRP) account for the largest share of US agri-environmental payments.

*Environmental taxes and charges* are applied in some countries on the sale of inputs identified as having a potentially adverse impact on the environment. Taxes and charges are currently levied on pesticides in Denmark, France, Italy, Norway and Sweden, while fertiliser levies are applied in Italy, Sweden and some states of the United States.

Other economic instruments, such as *tradeable permits and quotas*, are used in a limited number of countries. These include tradeable rights for the development of wetlands in the United States, tradeable water extraction rights (implemented on a state/regional basis in the United States), and improving market mechanisms to free up trade in water rights under Australia's *Water for the Future* reform programme. Tradeable rights based on environmental quotas, permits and restrictions do not yet appear to play a significant role in agri-environmental policy, despite the growing use of such measures for environmental policy in other sectors.

*Environmental cross compliance* – measures linking minimum environmental standards to agricultural support programmes – is used in the United States, Norway and Switzerland, and has been implemented more recently in Korea. Some EU member states (*e.g.* United Kingdom) have been implementing environmental cross compliance since the 1990s. From 2005, cross compliance (including environmental components) has become compulsory in the EU15**.** In the new EU member states (EU12), partial cross compliance applies already and full cross-compliance will be introduced between 2009 and 2013.

## Note

1. This chapter is based on *Stocktaking of Policy Measures addressing Agri-environmental Issues* (OECD, 2010b).

# *Chapter 8*
# *Ex-ante* and *ex-post* evaluation of agri-environmental policies

## *Alternative evaluation methods*

A wide variety of different methodologies can be used to evaluate agri-environmental policies. Both "*ex-ante*" and "*ex-post*" evaluations have been used in the policy development process (OECD, 2005a). This chapter focuses on three decision-making aids: cost-benefit analysis, cost-effectiveness analysis and multi-criteria analysis. Rational appraisal of agri-environmental policy requires a comparison of costs and benefits. Benefits may or may not be measured in monetary terms. Where they are not so measured, the relevant methodologies are cost-effectiveness analysis (CEA) and multi-criteria analysis (MCA)[1]. Where they are measured in monetary terms, the relevant evaluation procedure is cost-benefit analysis (CBA) (Pearce, 2005).

It should be noted that whereas cost-benefit analysis provides information on whether or not it is socially profitable to undertake any of the agri-environmental or conservation measures, CEA and MCA are basically limited to choosing between alternative policy measures, or ranking of policy measures, given that at least one policy measure is selected (for a broader discussion see, for example, Pearce, 2005 and OECD, 2006). Thus, both CEA and MCA can be effective – *e.g.* in terms of maximum environmental effectiveness for a given unit cost – but they may be "inefficient" in the case where none of the policy alternatives is socially profitable if implemented, that is the benefit-cost ratio of these measures is less than 1.

### *Cost-benefit analysis*

With regard to policy evaluation, social cost-benefit analysis is the closest to a social welfare analysis (Johansson, 1991). However, social cost-benefit analysis is a very information-intensive methodology raising considerable methodological and measurement challenges, since monetary estimates for non-market goods are needed. The basic idea behind cost-benefit analysis is to measure in monetary units how social welfare is affected by a particular programme or regulation, such as an agri-environmental or conservation programme. Cost-benefit analysis can be done either *ex ante* or *ex post*. The rationale for *ex-ante* analysis is that it will provide information on whether the proposed policy is socially profitable or not. *Ex-post* analysis assists the process of learning about what does and does not contribute to overall social well-being (Pearce, 2005). Practical cost-benefit analysis takes into account the following questions and issues (Pearce, 2005 and OECD, 2006):

i. **Policies.** What policies are available to address a given environmental target(s) and should policies be undertaken at all? The answer to the latter question is "yes" if the (*ex-ante*) present value of expected benefits exceeds expected costs of policy, and "no" if the costs exceed benefits.

ii. **Costs and benefits.** Whose costs and benefits count? The basic rule is that benefits and costs to all citizens in a country should be included and in some cases also those in other countries (*e.g.* for global warming).

iii. **Impacts.** Which impacts are included in cost-benefit analysis? Any gains and losses to anyone whose welfare (or well-being) is affected should be included in cost-benefit analysis and thus any impact of the policy that affects individuals' well-being is therefore a proper impact for inclusion.

iv. **Time horizon.** What is the time horizon over which costs and benefits are counted and what is the appropriate discount rate? Individuals prefer "now" to "later" and this time preference has to be included in cost-benefit analysis. The discounting of future benefits and costs expresses this time preference. Discounting (that is, the process of finding the present value of future costs and benefits) has been extensively debated in the context of cost-benefit analysis because distant future costs and benefits may appear insignificant when discounted. Usually, a constant (time invariant) discount rate is adopted, while many studies show that individuals may use time-declining discount rates.

v. **Relative price effects.** The income elasticity of willingness to pay implies that some of the benefits may attract a higher valuation over time relative to the general level of prices, *e.g.* because environmental goods are valued more at higher incomes.

vi. **Risk and uncertainty.** In the case of risk, the costs and benefits are not known with certainty but their probability distribution is known, whereas in the case of uncertainty the probability distribution is not known. As regards risk, decision makers' risk preferences (risk neutral or risk averse) will affect the decision rule (whether it is the expected value of costs and benefits or expected utility). In the case of uncertainty, a sensitivity analysis is required related to uncertain parameter values. Pay-off matrices showing the effect on a chosen parameter value of certain "states of nature" can be used.

vii. **Equity.** In addition to aggregate costs and benefits some form of distributional analysis (who gains and who loses) is called for and this issue can be addressed by attaching equity weights for money values of costs and benefits.

Hanley *et al.* (1999) summarise the results from 13 cost-benefit analysis studies of agri-environmental schemes (AES) in the United Kingdom. Most of the studies (ten) used contingent valuation for deriving monetary value for environmental benefits. For 12 studies the benefit-cost ratio was determined. In four of the schemes, the lower bound benefit estimate resulted in the benefit-cost ratio which was less than 1 so that these schemes failed the benefit-cost test. The higher benefit estimate for these schemes, however, resulted in benefit-cost ratios that ranged from 28 to 262. Other schemes passed the benefit-cost test also with lower bound benefit estimates. A valuable resource available to policy makers is the *Environmental Valuation Reference Inventory (EVRI)* (www.evri.ca), containing almost 2 000 valuation studies.

### *Cost-effectiveness analysis*

Cost-effectiveness analysis is basically a comparison of environmental effectiveness to its cost. This will give a cost-effectiveness ratio, which can then be used when comparing different policies. It should be noted that "environmental benefits" in this case are not measured in money units: the outcome is not an estimation of global social profitability. The trade-offs between different environmental impacts are not made in the modelling phase as in the cost-benefit analysis and are thus more transparent for policy makers in taking their decisions. Hence, cost-effectiveness analysis helps to rank policy measures but does not indicate whether it is socially profitable to implement any of the policy measures.

It should be noted that cost-effectiveness analysis may be difficult to apply in situations where different policy options have opposing impacts on different environmental issues. Consequently, there is a need for a framework that can handle multiple objectives while at the same time taking into account cost-effectiveness. Multi-criteria analysis attempts to provide insight on how to choose between policy options when multiple environmental dimensions are involved. The basic premise is that some assumption will have to be made about society's preferences for the different environmental issues, so as to aggregate from many dimensions into a one-dimensional measure that can be weighted relative to its cost.

### *Multi-criteria analysis*

Multi-criteria analysis is similar in many respects to cost-effectiveness analysis, but involves multiple indicators of effectiveness. MCA is a framework for ranking or scoring the overall performance of alternative decision options against multiple criteria which are typically measured in different units (Hajkowicz and Collins, 2007). MCA can be useful in promoting explicit consideration of the value judgments that are implicitly made in the application of single-objective approaches. Second, a wider range of policy alternatives is usually identified when a multi-objective methodology is employed, because analysts are less likely to be constrained into considering only those objectives that can be easily monetised. However, it is important to highlight that ranking policy options always requires making assumptions about decision-maker preferences over the objectives of a policy. The use of MCA is therefore only as good as the extent to which it accurately elicits policy makers' preferences. In this respect, the operational usefulness of MCA depends both on how readily objectives may be quantified, and also on how well the objectives are formulated so as to be meaningful and relevant to decision makers. For example, equity may mean different things to different people; it may be defined spatially, as for the distribution of benefits among regions; or it may relate to the distribution of impacts across income classes.

Hajkowicz and Collins (2007) describe the MCA process as follows: i) choose the decision options; ii) choose the evaluation criteria; iii) obtain performance measures for the evaluation matrix; iv) transform these into commensurate units (transforming criteria in different units onto commensurate scale, often 0 to 1, in order to combine them in the overall utility function); v) weight the criteria; vi) rank or score the options (the weights are combined with the performance measures to attain an overall performance rank or score for each option); vii) perform sensitivity analysis (*e.g.* with respect to weights and performance measures; and viii) make a decision.

Comparative studies of MCA apply more than one MCA technique to a single problem in order to compare ranking or scoring given by different techniques. These studies have shown that different MCA techniques produce similar results and that there is no clear methodological advantage to any of the techniques (Hajkowicz and Collins, 2007). Hajkowicz and Collins (2007) reviewed 113 published water management studies from 34 countries. MCA, when carried out by experienced practitioners carefully following MCA procedures, was found to provide transparency and accountability to decision procedures, help in conflict resolution between stakeholders, and clarify issues thanks to formal methods of decision theory to inform choice. Agricultural applications of MCA are reviewed in Hayashi (2000) and natural resource management applications are reviewed by Romero and Rehman (1987).

### *Ex-ante and ex-post assessment*

Lessons learned from applications are essential. An example is the US experience with water quality trading. These programmes have been subject to a number of *ex-post* assessments, with most programmes, and especially those involving agricultural nonpoint sources, showing little if any trading activity. This lack of trading activity is clearly an issue if the expected economic and environmental gains from trade are real, and if so, realisable. Design flaws have been identified as a key factor (*e.g.* Breetz *et al.* [2004]; Morgan and Wolverton [2005]; King [2005]; King and Kuch [2003]; Ribaudo *et al.* [1999]; Hoag and Hughes-Popp [1997]; Shabman *et al.* [2002]; Stephenson *et al.* [2005]). There is significant effort by some public agencies and nongovernmental organisations that are committed to the success of trading in the US and elsewhere to provide nuts-and-bolts guidance for developing effective trading programmes (*e.g.* EPA, 2007).

But prior experience is a limited guide to agri-environmental policy design. Agri-environmental programmes have to date largely emphasised voluntary compliance approaches in which farmers are encouraged to adopt pro-environmental farming practices, provided with technical assistance in adoption, and sometimes offered payments for adoption. The effectiveness of voluntary programmes without, or with modest, financial inducements has been limited (Horan *et al.*, 2001). Programmes with adequate funding have had noted environmental success, though there are concerns for cost-effectiveness. A leading example is the US CRP. Applications of standards, and especially taxes, are limited. Standards are widely used and have been effective in addressing pesticide risks in many nations, but significant issues are raised about cost-effectiveness.

Given the limited experience with standards, taxes, and permit trading, an essential guide to evaluating alternatives is to conduct formal *ex-ante* assessments (Ribaudo and Shortle, 2001). There is a growing body of economic literature conducting *ex-ante* assessments of agri-environmental policies.[2] That literature generally indicates that policy instruments must be optimised and evaluated in the specific social, economic-political, legal, and environmental context in which they are applied. Lessons learned in one context should be applied with caution to another. The literature indicates that multiple instruments often address the complexity and uncertainty of agricultural problems better than single instruments.

*Ex-ante* assessments ideally use integrated assessment procedures that couple or combine economic and environmental models to evaluate economic responses to instruments, the costs of changes in resource allocation, and the impacts of changes in resource allocation on environmental metrics. Because of the importance of uncertainty about economic and environmental relationships for the design and performance of policy instruments, explicit consideration of uncertainty is essential. An example is Borisova *et al.* (2005) who compared input-based permit trading to input-based taxes for reducing agricultural nonpoint pollution in the Susquehanna River Basin. The research uses a simulation model that explicitly captures uncertainty about the impacts of agricultural practices on pollution loads, the delivery of pollution loads to downstream receptors, the economic costs of reducing pollution loads, and the economic damage costs of water pollution. In the study the instruments were designed to maximise the expected net benefits from pollution control rather than to achieve an exogenous target at the least expected costs. They found optimised input tax instruments to have a small but statistically significant advantage over input-based permit trading.

### *Agri-environmental Footprint Index – measuring environmental performance in agri-environmental policy evaluations*

All EU member states are obliged to monitor and evaluate the environmental, agricultural and socio-economic impacts of their agri-environmental programmes (Article 16, Regulation [EC] No. 746/96). The evaluation process aims to determine the extent to which policy objectives are being fulfilled, and to identify any changes necessary to bridge the gap between policy aims and outcomes. However, there is little consensus on how to monitor and validate the benefits of agri-environmental schemes (AESs) successfully. Critically, there are no agreed methodologies for tracking the environmental consequences of changing agricultural practices, or the benefits of particular agri-environmental policy measures.

The AE-Footprint project developed a common methodology and tools to assess the environmental performance of AESs and it was funded in response to the EU Task 11 – Agri-Environment: Assessment of Agri-environmental Schemes with Rural Development as the second pillar of the Common Agricultural Policy. The main objective of this research was the conceptual and practical development of a harmonised assessment system with which to assess the environmental performance of Europe's AESs.

The *Agri-environmental Footprint Index (AFI)* is a farm-level index that aggregates the measurement of agri-environmental indicators. It can be used for a number of purposes:

- To measure the changing environmental impact of individual farms within a particular context (farming type, geographical region) over time;
- To produce a measure of environmental impact that can be aggregated across farms in a similar context; and
- To enable comparison of the environmental impact of farms which do/do not participate in agri-environmental schemes/measures (AES/AEM).

The basic idea is that a policy maker will commission evaluators to apply the AFI methodology to a particular type of agriculture, or to a given agri-environmental scheme or mechanism to measure its effectiveness. The evaluators will follow a prescribed AFI methodology involving consultation with both stakeholders and a technical panel, the overall outcome being a quantitative index measuring the environmental impact at the level of individual farms. A higher AFI score indicates higher, or improving, environmental quality and thus a reduced negative impact. Farm level impact scores can be aggregated at a regional level to track temporal change and/or to provide comparisons of the success of the chosen policy mechanism.

The methodology developed in the AE-Footprint project involves the construction of an Agri-environmental Footprint Index, allowing the combination of various indicators reflecting the environmental performance of a particular farm. The approach employs components of multi-criteria analysis techniques to provide a means of combining indicators corresponding to a variety of farm management activities and relating to a range of environmental objectives. Multi-criteria analysis methods are ideally suited to the measurement of multi-faceted situations, especially where the relative importance of each component is not precisely defined (Park *et al.*, 2004).

The methodology incorporates the participation of stakeholders and technical advisors in designing a customised form of the AFI relevant for each particular policy scheme. In the methodology, stakeholders validate the assessment criteria and provide a series of weights allowing combination of different components of environmental performance. Such input of specific, technical and local knowledge ensures that the evaluation is appropriate to the local agri-environmental context. For the purposes of this methodology, a stakeholder can be defined as someone who can affect or is affected by the agri-environment scheme or the local agri-environment.

The methodology for the AFI can be described as a stepwise procedure (Table 8.1). As with any evaluation, the first step involves defining the aims of the evaluation, and would typically comprise a statement of the overall goals of the policy to be evaluated, the scope of the evaluation, the relevant farming systems and regions, the sampling strategy, and the time frame.

**Table 8.1. Steps in the Agri-environmental Footprint Index methodology**

| Step | Evaluation Team | Stakeholder Group |
|---|---|---|
| 1 | Define application | |
| 2 | | Create Assessment Criteria Matrix |
| 3 | | Define Issue and Domain weights |
| 4 | | Create Indicator Matrix |
| 5 | Collect data | |
| 6 | | Define Transformation Functions |
| 7 | | Define Indicator weights |
| 8 | Calculate Index | |
| 9 | Sensitivity Analysis | |
| 10 | Reporting | |

*For more information, see www.footprint.rdg.ac.uk/en/home_en.html.*

*Source:* Mortimer and Finn (2008).

# Notes

1. Note that within a cost-benefit framework *threshold analysis* can be used. This involves monetising all costs and benefits that can be monetised and at that point comparing the total costs and total benefits. A judgement can be made as to whether the non-monetised benefits are likely to bridge gap between the benefits and costs. If monetised benefits are only slightly less than costs and it is clear that non-monetised would easily be greater than the gap then that would make policy worthwhile. The same considerations apply on the cost side.
2. Two sources for literature reviews are Horan and Shortle (2001), and Shortle and Horan (2001).

# *Chapter 9*
# Summary and good policy practices

## *Summary*

### *Background*

Improving the environmental performance of agriculture is a high priority in OECD countries. Specific policy measures designed to address environmental issues in the agricultural sector are relatively recent, but are becoming more widespread. These measures vary considerably across and even within countries, reflecting the severity of environmental stress, the potential for providing ecosystem services, and historical and cultural developments that influence policy priorities. Such measures do not operate in a vacuum: they are implemented alongside agricultural income support policies and economy-wide environmental policies, in a wider socio-economic and technological context. Moreover, possibilities to create markets or quasi-markets are constantly evolving – which are closely linked to property rights – and thus alter the need, focus and type of policy intervention.

There are three major characteristics of the agri-environment that influence the design and implementation of agri-environmental policy. First, many of the environmental effects of agricultural activities are externalities (positive and negative) or public goods for which markets are absent, and property rights are lacking. When farmers do not have incentives (or disincentives) to take into account, the implication is that natural resources will be over-exploited, such that there will be too much pollution, and too little provision of environmental public goods. This provides the basic rationale for policy intervention in this area.

Second, the environmental impacts of agriculture vary spatially – from site-specific or water catchment area, through to national and even international relevance (such as biodiversity and greenhouse gas emissions) – and temporally, given that some environmental impacts can take a considerable time to become evident. Some of the environmental impacts result from the actions of individual farmers, but some result from the actions of many farmers in a particular geographical area (such as provision of habitat or cultural landscapes). This is a significant challenge for policy – targeting specific farmers and activities at the appropriate level of governance, while minimising public and private policy-related transaction costs, and ensuring that the incentive structure attracts the farmers who can contribute most to improving environmental performance (avoiding adverse selection and moral hazard), while taking into account equity issues.

Third, and linked to the previous two characteristics, partly due to the relatively recent policy concern with agri-environmental issues and partly due to the inherent nature of the relationship between agriculture and the environment, there is a lack of comprehensive data and analysis to inform policy decision makers. Although significant progress has been made in tracking environmental performance and policies, and understanding the linkages between agriculture and the environment, the complexity of the policy-environment relationships, the absence of markets and monetary values for many environmental effects leads to serious limitations in evaluating the effectiveness and efficiency of agri-environmental policies. Nevertheless, this is to some extent alleviated through the sharing of cross-country experiences.

In the arsenal of policy instruments used in OECD countries for managing agri-environmental issues, environmental standards; environmental taxes; agri-environmental payments and tradeable permit schemes are important. Other approaches are also significant – research and development, information provision and education, training and advice and, more indirectly, moral suasion. Applications of these various tools vary across countries, and they have evolved over time as lessons are learned about the merits of alternative approaches for different problems and as the problems themselves change. The scope of agri-environmental problems and issues has expanded over time, including recognition that agriculture also contributes to providing environmental services. As a consequence, the types of policy instruments used to address them have expanded with varying degrees of success.

The aim of this *Guidelines* study is to help policy makers in the design and implementation of cost-effective agri-environmental policies. It focuses on environmental standards, environmental taxes, agri-environmental payments and tradeable permit schemes to address agri-environmental problems. It is important to note that the goal of this study is not to promote any specific policy instrument or instrument-mix, but to better understand how different types of policy instruments can be used, in what context, and which are key design and implementation issues for the success of a given instrument.

Moreover, the list of instruments analysed in this report is not a complete or exhaustive list of available policy instruments for policy makers. In particular, this study does not deal with those approaches whereby governments assist farmers through funding education and research and development as well as providing technical assistance and extension services at the farm level in order to increase voluntary adoption of environmentally friendly farming practices and technologies. That they are not analysed should in no way diminish their importance. For example, educational programmes can encourage farmers to take pro-environmental actions leading to environmental improvements when: i) pro-environmental actions also increase profitability; ii) farmers have strong altruistic or stewardship incentives; and iii) there are also significant on-farm costs due to environmental damage. In fact, some educational programmes relating to conservation tillage, nutrient management, integrated pest management and irrigation water management have resulted in win-win solutions, in which both profitability and environmental performance have improved when compared to conventional practices. However, both potential win-win solutions and stewardship incentives are unlikely to satisfy society's overall demand for environmental quality from agriculture and thus there is a need for more direct policy interventions, which are the focus of this study.

The study is essentially concerned with two sets of issues. The first set addresses choices among the range of policy instruments. For instance, under which criterion an environmental tax performs better than a standard, or permit trading performs better than an environmental tax? The second set of issues addresses the design of particular instruments. Economic theory, supported by simulation analyses and *ex-post* assessments of environmental instruments, demonstrates that the details of the design and implementation of policy instruments matter greatly in terms of both environmental and economic outcomes.

Providing useful information to guide policy instrument choice and design decisions is inherently contingent on having clearly defined policy objectives. Thus, this study begins with an overview of the functions of agri-environmental policy instruments and criteria for policy evaluation. This is followed by an overview of the core policy design parameters. Then a more specific analysis of agri-environmental policy mechanisms or instruments is presented. The various types of environmental standards, taxes, tradeable permit schemes, agri-environmental payments, and policy-mixes that can be constructed are introduced and analysed in relation to the design and implementation parameters. A discussion of the use of formal *ex-ante* and *ex-post* policy analysis and evaluation to assess the performance of alternative types of policies is then provided.

### *Policy instrument choice*

The fundamental purpose of agri-environmental policy instruments is to achieve environmental policy objectives that would not otherwise be achieved given the absence or poor functioning of markets for environmental goods and services. Achieving those objectives requires either controlling or managing environmental stress, such as polluting emissions, or inducing pro-environmental activities to increase the flow of ecological services, such as management of agricultural practices and land to enhance desired wildlife habitat. In either case, achieving the desired end requires changes in producer decisions consistent with the achievement of the agri-environmental policy objectives.

### *Criteria for policy choice*

Five criteria are relevant in guiding *ex-ante* instrument choice and design decisions and to measure *ex-post* instrument performance: *Environmental effectiveness* is the first criterion for evaluating policy instruments and refers to the capacity of the instruments to achieve stated environmental goals or targets.

*Economic efficiency* refers to balancing of costs and benefits of policy intervention, that is, marginal value of environmental improvement to be equal to the marginal costs of generating that improvement. Although economic efficiency criterion is of little use in practice due to lack of information related to social costs and benefits of environmental improvements it is important reminder to policy makers that net benefits of policy intervention should be positive. *Cost-effectiveness* refers to the costs of achieving society's environmental objectives. The cost-efficient policy instrument is one that minimises compliance costs while achieving environmental target, thus maximising cost-effectiveness. Cost-effectiveness can be defined with respect to reductions in environmental pressures, or in terms of improvements in environmental states. Spatial variation in costs and impacts implies that cost-effective achievement of environmental goals will generally entail differential levels of environmental effort across farms.

*Administrative costs* refer to public sector costs and capacities. Different policy instruments impose different demands on the management capacities of public agencies, and the costs to the public sector for design, implementation, monitoring and enforcement. There is usually a trade-off between targeting and tailoring of policy instruments and their policy-related transaction costs.

*Ancillary benefits and costs* may be environmental, economic or related to other objectives (such as food security). In the first case, an instrument that reduces nutrient loads will improve water quality, but it may also improve wildlife habitat if, for example, the management practices used to reduce nutrient loading include establishment of buffer strips or creation of wetlands. Another example is carbon sequestration in agricultural soils that may also provide co-benefits in terms of water quality and biodiversity. Moving to more targeted instruments may entail losses of some ancillary benefits losses.

*Equity* plays an important role in evaluating policy instruments with regard to the fairness of the distribution of economic costs and benefits between and among different groups (producers, consumers, and taxpayers). It can be the case that more than one type of instrument will be capable of producing a cost-effective outcome, but each will yield different distributions of wealth and will therefore be viewed differently from an equity perspective. Policy makers will need to weigh up the trade-offs between equity, efficiency, and other criteria in choosing among policy instruments.

### *Uncertainty and policy instrument choice*

Each of the policy performance criteria are subject to uncertainty on the part of the regulatory agency, and each form of uncertainty is relevant to the analysis of payments, standards, taxes, and permit trading. One source of uncertainty about both costs and environmental impacts arises because public decision makers, when choosing instruments, are unable to predict with certainty the impacts of their choices on farmer's production and land-use practices, and the costs to farmers of changes in their practices. Economic models can be used to forecast policy-induced changes in production and land-use practices and compliance costs, but forecasts are always subject to uncertainty. There are two implications of this *ex-ante* uncertainty about compliance and compliance costs. One is that the economic costs of prospective policies are uncertain. A second is that the environmental outcomes, as measured by pressure or state indicators that result from the application of instruments are uncertain since those outcomes are driven by the uncertain changes in production and land-use practices.

Uncertainty about environmental outcomes is affected by additional factors. One is the uncertainty about the levels of individual farmers' contributions to environmental externalities. For example, nutrient runoff contributions to water resources from individual farms cannot be measured because they are diffuse and complex. Models can be and are used to forecast the effects of changes in farm practices on environmental pressures, but such models are generally subject to substantial error. Models are also used to predict the effects of changes in farm pressure indicators on environmental state indicators. These models too are subject to substantial error. Finally, many agri-environmental processes, such as nonpoint source pollution, are driven by random weather and other events, largely outside of the control of farmers.

### *Environmental targets, reference levels and property rights*

A crucial requirement in choosing between policy instruments is that some of the criteria that guide policy makers' decisions, such as fairness and equity, are dependent on the definition of reference levels and property rights. Therefore, it becomes apparent that defining how to address the environmental impacts of agriculture requires a case by case response in relation to the settings of the environmental targets and definition of environmental reference levels based on the identification of existing property rights defining who can ask for remuneration and who is liable for charges.

The definitions of environmental targets and reference levels vary between countries. Environmental targets depend on society's preferences for environmental quality, while reference levels depend on the country's traditions and laws in defining property rights. The efficient setting of environmental targets has to balance the benefits of pursuing environmental objectives against the resulting welfare losses due to lower production or consumption of other goods and services. But, whereas the setting of environmental targets is based on efficiency considerations, the issue of identifying the relevant environmental reference levels (who should bear the costs of reallocating resources to meet environmental targets) is based on distribution (equity) considerations and property rights.

### *General instrument design and implementation parameters*

In order to achieve an intended objective, a plan and means to reach it are required. The desired objective can be defined by choices of environmental goals along with the economic goal of cost-effectiveness. Instruments generally, though not always, ought to be directed at those who are directly responsible for environmental harm or who are most capable of providing environmental enhancements. Instrument design is accomplished by making choices about various design parameters.

Some specific choices may vary across instruments, but generally the available parameters involve responses to three broad questions: 1) to whom and to what degree, among the set of possible contributors to environmental externalities, should the instrument be applied – that is, who to target?; 2) what is the optimal target variable for defining and measuring individual farm-level compliance with environmental target – that is, what to target at the farm level – environmental outcomes (*performance-based*) or farmers' input and technology choices (*input-based*)?; and 3) which incentive – that is, what specific policy instruments (such as payments, environmental standards, environmental taxes, and tradeable permit schemes) should be tied to the chosen compliance metrics to induce the changes in farm-level behaviours that are needed to produce the desired outcome?

Even if measurement of actual environmental impacts were possible in some instances, the high degree of natural variability of processes such as nonpoint source pollution, carbon sequestration, and flood prevention means that farmers will be unable to control these performance outcomes deterministically (without randomness).

Two important policy implementation parameters are the choice of the level of administration and the choice of the enforcement strategy. Information is crucial input to the design and implementation of policy and use of local information allows better targeting and tailoring of policy incentives. However, there is potential problem with possible strategic behaviour of lower levels of government in the case of too close identity of interest between local administration responsible for policy design and implementation and farmers. Enforcement of policy requires resources and involves costs for compliance monitoring and imposing penalties for detected violations. Policy instruments may differ greatly in their enforceability, and thus in their enforcement costs. Management requirements that are observable by eye (buffer strips, green set-asides, etc.) are easier to monitor and enforce than non-visible constraints, such as fertiliser and pesticide application rates.

### *Environmental standards*

Environmental standards are mandates applied to the quality or quantity of marketed products (product standards), technologies or processes (process standards), or environmental performance (performance standards). Product standards regulate marketed production inputs or outputs, process standards directly regulate choices of production and pollution control technologies, while performance standards directly regulate measures of non-market outputs (including indicators of environmental performance).

Here process and input-based product standards are collectively referred as input standards.

Environmental *performance standards* are a common method of regulating polluting emissions from non-agricultural point sources. Environmental performance standards can take a variety of forms, though they typically impose an upper limit on the externality or the selected indicator. An argument for performance standards by comparison to input standards is that they allow producers the flexibility to meet mandated environmental outcomes in any way they choose, thus allowing them to find ways to achieve the standards at minimum cost. Performance standards may therefore promote farm-scale cost-effectiveness, and also promote cost-saving technological innovations.

The cost-effectiveness properties of performance standards would be increased if the standards are applied differentially based on producers' individual environmental impacts but this would require extensive and expensive information. Imposing differentiated standards to producers is likely to raise equity issues and would imply that reference levels are determined at sub-national levels. However the cost effectiveness of the standards could still be increased by defining more stringent standards in specific vulnerable zones.

*Input standards* (product or process) place mandates or constraints directly on producers' choices. Here the production process, technology, the products that are used, or the manner in which they are used, are regulated. For agriculture, process standards might consist of regulations pertaining to the ways producers manage their crops, livestock, and their land. Options might include regulations on input use (*e.g.* levels, timing, and forms of agricultural chemical application) or the use of specific practices and technologies (*e.g.* erosion and runoff controls, irrigation equipment, and collection and land application of animal waste). Process standards relating to the management of animal wastes are used for large confined animal operations to protect air and water quality. Input standards do not, however, provide producers with the flexibility or incentives to look for cost-effective solutions to environmental problems.

### *Environmental taxes*

The goal of an environmental tax is to alter the economic incentive structures of farms so as to align their economic interests with societal objectives. Essentially, the mechanism is intended to correct the incentive failures resulting from missing markets for environmental goods by replacing missing price incentives with administered taxes or charges. Parallel to an incentive-based performance standard is a tax. Environmental taxes applied to negative externalities have long been advocated by economists as an efficient remedy for environmental externalities. As with performance standards, the first choice in the design of performance taxes is to determine what will be taxed. Here the issues are the same as they were for performance indicator standards: using performance taxes requires consideration of the availability, reliability, and cost of information measuring farm-level environmental performance.

An input-based tax increases the cost of implementing a practice having adverse environmental impacts (or, alternatively, it can be structured to reduce the cost of implementing environmentally-friendly practices). Input-based taxes, since they are not based on performance, do not encourage farm-level cost-effectiveness unless all relevant processes are taxed at the correct rates. The ultimate effectiveness and efficiency of process-based taxes depend on the two design decisions the agency must make with regards to these instruments namely, which processes to tax, and at what levels to set the taxes.

### *Tradeable permits*

Tradeable permits for regulating environmental externalities can often achieve environmental targets at lower social cost than traditional design and performance standards and environmental taxes. Indeed, success stories for air emissions trading in the US have spurred interest in expanding the scope of markets for environmental management. The most visible developments internationally are those addressed to greenhouse gases (carbon trading). Another growing area is water quality trading, including programmes to address agricultural sources of water pollution, for example, point/nonpoint trading. Trading offers a mechanism for achieving a cost-effective allocation of environmental effort across alternative sources without environmental regulators knowing the abatement costs of individual agents.

It is fair to say that water quality trading markets are much more complex than emission trading presented in standard economics textbooks because there is plenty of uncertainty about sources and levels of emissions, and about effectiveness of different abatement measures and water quality impacts of effluents originating from different sources. When developing water quality trading market the policy maker has to first define the tradeable commodity for nonpoint polluters (*e.g.* fertiliser use reduction or establishment of buffer strips and green set-asides). The trading ratio has to be determined that takes into account delivery of pollutants and imperfect substitution between point and nonpoint emissions (on the basis of relative uncertainty related to reduction of emissions from these two sources).

Finally, the aggregate supply of permits has to be limited (cap) so that water quality targets are met and a method for the initial allocation of rights has to be chosen.

### *Agri-environmental payments*

Many OECD countries offer monetary payments to farmers to encourage them, on a voluntary basis, to implement more environmentally friendly farming practices going beyond those required by regulations or defined as good farming practices. Most of these agri-environmental programmes offer a single, fixed payment for compliance with a pre-determined set of environmental requirements, such as reduced tillage or limits on the intensity and timing of fertiliser, manure and pesticide applications. The obvious problem with this type of fixed-rate payment approach is that heterogeneity in either farmers' compliance costs or site-productivity of environmental goods supplied are not taken into account in policy design and implementation. However, the targeting of these programmes is often improved by defining prioritised zones.

Designing and implementing cost-effective agri-environmental payment programmes is difficult because of asymmetric information between a farmer and a policy maker. Information asymmetries exist if farmers have hidden information (or characteristic), which may lead to *adverse selection*, or alternatively, hidden action, which may give rise to *moral hazard*. There are two mechanisms that address adverse selection and could improve the cost-effectiveness of agri-environmental payments relative to a fixed-payment approach: i) conservation auctions, *i.e.* bidding mechanisms and ii) self-selection mechanisms. Moral hazard can be addressed through, for example, intensity of compliance monitoring, level of fines/sanctions, observable compliance criteria, and level of payment.

Auction theory provides an interesting way to incorporate competition between farmers for winning a conservation contract with the policy maker. Auctions can improve both allocative efficiency (bids with highest benefit-cost ratio are selected to the programme) and budgetary cost-effectiveness (maximise environmental benefits with a given fixed budget). In conservation auctions, farmers bid competitively for a limited number of environmental conservation contracts. When making a bid a farmer faces a trade-off between net pay-offs and acceptance probability so that a higher bid increases the net pay-off but reduces the probability of getting a bid accepted. Thus, competitive bidding will push farmers to reveal their estimated compliance costs and as a result it will reduce (but not eliminate) farmers' information rents and improve the cost-effectiveness of an agri-environmental programme.

As regards policy design for conservation auctions the first choice to be made is between different *payment formats*. Another key auction design parameter is the so-called *reserve price*, which is the upper limit of payment per unit of conservation benefit and it can be pre-announced or not. The reserve price increases bidding competition and thus reduces farmers' information rents, but also provides the signal of maximum willingness to pay for farmers' conservation services or environmental goods.

One of the key design issues in conservation auctions is the bid evaluation system (enrolment screen). Because farmers' environmental practices usually provide multiple benefits a multi-criteria bid scoring system could be adopted to aggregate the overall environmental benefit of bid. These types of multi-criteria bid scoring systems are, for example, the environmental benefit index for the US CRP and the biodiversity quality index used in the BushTender programme in Australia. Cost-effectiveness gains from conservation auctions vary significantly, but empirical evidence regarding auction performance is still inconclusive.

The environmental and economic performance of agri-environmental payment programmes depend critically on several policy design parameters that will affect which farmers will apply and which applications are accepted. Design of cost-effective agri-environmental payment programmes require: i) the identification of those farmers, land parcels, and practices which are most likely to achieve programme objectives with the least cost and ii) formulation of eligibility criteria, payment incentives, and enrolment screening so that enough of the "right" farmers apply, especially where the environmental benefits relate to spatial aspects involving several farms in the area. Thus, policy makers have a number of design parameters available to attract the right participants; those farmers who can make the most valuable contribution to achieving programme objectives.

Although benefit-cost targeting through environmental performance screening combined with competitive bidding or with differentiated payments is likely to yield higher budgetary cost-effectiveness than fixed-rate payment approaches, the gains from targeting need to be weighed against potential increase in the administrative (transaction) costs, losses of ancillary benefits, and equity considerations.

### *Policy-mixes*

Inherent information problems related to the many agri-environmental issues (*e.g.* nonpoint source characteristics of pollution) may necessitate the use of a policy instrument-mix, for example an economic instrument together with regulatory or information instruments. Since all single policy instruments have their strengths and weaknesses it is important to combine instruments so that their complementary interactions are maximised and counterproductive interactions minimised. Implementing a mix of instruments rather than a single policy instrument has a number of advantages: i) many environmental issues are multifaceted so that not only the amount of emissions, but also where emissions take place and when they occur etc. are relevant; ii) many instruments can mutually strengthen each otherl; and iii) sometimes instrument-mixes can also enhance enforcement and reduce policy-related transaction costs. However, there are also reasons for restricting the number of instruments in the mix. For example, when several instruments are applied in the mix there could be a danger that one instrument hampers flexibility to find low-cost solutions to a problem that another instrument could have offered if it had been implemented on its own.

### *Ex-ante and ex-post evaluation*

A wide variety of different methodologies can be used to evaluate agri-environmental policies. Both "*ex-ante*" and "*ex-post*" evaluations have been used in the policy development process. With regard to policy evaluation social cost-benefit analysis is the closest to a social welfare analysis. However, social cost-benefit analysis is a very information-intensive methodology and results are subject to large uncertainties, since monetary estimates for non-market goods are needed. The basic aim is to measure in monetary units how social welfare is affected by a particular programme or regulation, such as agri-environmental or conservation programmes, so that well-informed policy decisions can be made.

Cost-effectiveness analysis is basically a comparison of environmental effectiveness to cost. This will give a cost-effectiveness ratio, which can then be used when comparing different policies. It should be noted that "environmental benefits" in this case are not measured in money units: the outcome is not an estimation of global social profitability. The trade-offs between different environmental impacts are not made in the modelling phase as is the case for cost-benefit analysis and are thus more transparent for policy makers in taking their decisions. Hence, while cost-effectiveness analysis helps to rank policy measures, it is less useful than cost-benefit analysis in indicating whether it is socially profitable to implement any of the policy measures.

Multi-criteria analysis (MCA) is similar in many respects to cost-effectiveness analysis but involves multiple indicators of effectiveness. It is a framework for ranking or scoring the overall performance of alternative decision options against multiple criteria which are typically measured in different units.

## *Good policy practices for the design and implementation of cost-effective agri-environmental policies*

### *Agri-environmental policy objectives*

- The fundamental objective of agri-environmental policy is to achieve environmental policy goals with the least overall cost to society, including farmers' compliance costs (consisting of both direct costs and opportunity costs) and policy related transaction costs, taking into account equity considerations.
- Environmental objectives should be set (and then later be achieved) with economic efficiency in mind: i) the marginal benefits and marginal costs of achieving environmental objectives should balance reasonably well; and ii) whatever goal is set, it should be achieved at least cost.
- Environmental objectives should be specified, if possible and feasible, in terms of environmental performance rather than recommended practices.
- Environmental objectives should be quantifiable and formulated in a way that as far as possible allows progress to be assessed quantitatively.
- *Ex-post* evaluation of agri-environmental policies should, when possible, use ecological and environmental indicators as measures of impact to complement participation-based measures. If the policy choice has been made on the basis of a quantitative model, the assessment of model results should be integrated in the *ex-post* analysis.

### *Broad policy design principles*

- There are many policy design parameters that policy makers need to take into account, but essentially they involve three broad questions: i) who or where to target?; ii) what to target?; and iii) which incentives ought to be used?
- For a given result, governments should focus on measures that minimise unintentional transfers, net losses, losses of ancillary benefits, and transaction costs (OECD, 2007b).
- To facilitate the adoption of measures and to improve their design, the co-operation of relevant stakeholders is advisable.

- Who or where to target?
  - Measures should target those areas where agriculture contributes to providing environmental services or generates environmental harm.
  - Within these areas targeting those farmers who farm the most environmentally sensitive fields or livestock enterprises or those who can deliver environmental goods with the least cost would increase the efficiency of the programme. However, it may be difficult as may raise equity issues and require much higher administrative costs.
  - Applying a transparent benefit-cost targeting approach can help to deliver cost-effective outcomes, but other quantitative methods such as CEA and MCA can also be useful and leave trade-offs more explicit for policy decisions.
- What to target?
  - Directly targeting emissions or runoff is difficult in agriculture due to nonpoint source characteristics and thus may require targeting proxies such as farm practices or inputs, technology and land use.
  - Performance-based targeted measures may use proxies, such as nutrient surplus, manure surplus or environmental indices.
  - What is targeted should correlate highly with environmental objectives and should be easily monitored and enforced, without incurring high transactions costs.
  - Some performance-based measures are problematic because they cannot be deterministically controlled by the farmers, such as nutrient surplus which is dependent on nutrients contained in yield relative to inputs and where yield is highly affected by weather conditions.
- Which incentives?
  - Incentives should avoid adverse selection and moral hazard, incur as low as possible transactions costs and voluntary programmes should attract a high participation rate from the targeted farmer group.
  - The choice of policy instrument not only affects the environmental effectiveness (the uncertainty of environmental

outcomes) and cost-effectiveness but also the distribution of costs and thus the societal acceptability of the policy instrument.

  – In many situations a combination of instruments may perform better than single policy instruments but this requires that the policy instruments should maximise their complementary interactions and minimise counterproductive effects.

- The cost-effective design of agri-environmental payment programmes is difficult because of asymmetric information between a farmer and a policy maker and this information asymmetry is manifested in hidden information about the type of farmer (*e.g.* low and high productivity farmer) or hidden action as regards the farmer's compliance with environmental requirements.
- Two policy mechanisms address to some extent hidden information and adverse selection: i) bidding mechanisms, that is, auctions; and ii) self-selection mechanisms (principal-agent type differentiated contracts). However, empirical evidence from these mechanisms is still inconclusive and further research is needed. Transaction costs may vary substantially depending on the country considered
- Hidden action and thus moral hazard can be addressed with more intense monitoring; appropriate level of fines for noncompliance; observable compliance criteria; and appropriate level of agri-environmental payments (that is, the higher the payment, the higher is the implicit penalty of detected non-compliance), including over-compensation.
- Targeting monitoring efforts, with higher sanctions on farmers for those instruments that are more lightly monitored, can contribute to reducing monitoring and enforcement costs for a given level of compliance.
- In auction systems, farmers bid competitively for a limited amount of conservation contracts so that higher bids increase net pay-offs but reduces the probability of their being accepted. Competitive bidding reduces the rents farmers gain from their privileged information thus increasing budgetary cost-effectiveness.
- In those conservation auctions that are based on environmental screens or indices two sources of cost-effectiveness gains arise: those arising from competitive bidding and thus from information rent reduction, and those arising from improved environmental targeting. The relative importance of these two sources of efficiency gains is an empirical issue.

- Discriminatory pricing payment formats in which farmers are paid according to their bids if selected into agri-environmental programmes reduce farmers' information rents but do not completely eliminate them, since farmers' optimal strategies would be to shade their bids over their real compliance costs.

- The main weakness is that bidders learn to bid and use that information from earlier rounds to overbid in current and future rounds and thus ultimately the cost-effectiveness gains of bidding over flat-rate payment approach are eroded.

- Potential cost-effectiveness gains from performance-based agri-environmental programmes may be large, arising mainly because performance-based measures address the heterogeneity of farm conditions and the supply of environmental services, and provide flexibility to farmers to select those practices that are least cost in the context of each farm.

### *Broad policy implementation principles*

- The choice of the level of administration is important for policy implementation. Spatially targeted agri-environmental policies require a stronger involvement of local government and usually lower levels of government may result in better targeting and tailoring of policy incentives; however, there is a potential problem that a too close relationship between the local administration responsible for policy design-implementation and farmers could lead to enforcement difficulties.

- The choice of target (regulation) area is important because different agri-environmental issues have different spatial dimensions and the geographical delimitation of agri-environmental policy should fit the spatial dimension of the environmental issue in question.

- Enforcement of policy requires resources and involves costs for compliance monitoring and imposing penalties for detected violations. Policy instruments may differ greatly in their enforceability, and thus in their enforcement costs. Management requirements that are easily observable – such as buffer strips and green set-aside – are easier to monitor and enforce than those that are non-visible constraints, such as fertiliser and pesticide application rates.

# *Annex A*
# Compliance bases for alternative agri-environmental policy instruments

| Compliance Measure | Mechanism | | |
|---|---|---|---|
| | Standards | Taxes | Trading |
| ***Inputs*** | | | |
| Potential compliance bases | Pesticide registration<br><br>Restrictions on fertiliser application rates<br><br>Mandatory use of practices for pollution control, carbon sequestration, provision of habitat or landscape amenities | Charges on fertilizer or pesticide purchases<br><br>Charges on manure applications<br><br>Cost-sharing or other subsidies for inputs or practices that reduce pollution<br><br>Crop land retirement subsidies | Input trading |
| ***Environmental Performance*** | | | |
| Potential compliance bases | Restrictions on modelled nutrient loadings<br><br>Regulations on nutrient applications in excess of crop needs | Charges on modelled nutrient loadings<br><br>Charges on nutrient applications in excess of crop needs<br><br>Charges on estimated net soil loss | Estimated emissions trading |

# *Annex B*
# Cross-compliance in the European Union

The EU approach to cross-compliance includes partial or full loss of payments if the farmer fails to comply with mandatory standards stemming from existing legislation and the maintenance of good agricultural and environmental conditions. Cross-compliance creates a link between several separate policies, amongst them income support and selected statutory standards or requirements. These relate to environment, animal and plant health, public health and animal welfare and identification and registration of animals and are enshrined in existing laws. By introducing reduction of payments due to non-compliance the effectiveness of enforcement of existing environmental laws could be expected to increase.

Primary legal enforcement of environmental legislation is done through European Union member states' sanctioning systems. Cross compliance is assisting in reinforcing the respect for the basic requirements and standards, avoiding support to farmers that do not abide by these rules.

The EU uses a system in which both statutory requirements and voluntary provision are complementary. Farmers receiving agri-environment payments for voluntary commitments must in any case respect the mandatory standards. In that sense, the European Union cross-compliance system already provides the baseline for calculation of payments for agri-environmental measures. EU member states and Regional Authorities define the cross-compliance standards on the basis of the EU framework adapting them to local conditions in order to deal with heterogeneity in local circumstances.

Cross-compliance neither directly pursues an income support objective nor is it the primary mechanism for enforcing environmental legislation. Rather, cross compliance is a tool linking payment schemes to the respect of a wide array of mandatory requirements and fostering adherence to them.

*Source:* OECD (2008a).

# *Annex C*
# Standards *versus* taxes

The concern about performance standards meeting overall environmental goals, at least without additional instruments, arises because performance standards conventionally limit only one of two variables that determine ambient environmental conditions (*e.g.* Sterner, 2003). To illustrate, in a linear water quality model, the ambient concentration of a pollutant in the environment ($a$) is a weighted sum of the polluting emissions from individual sources ($e_i$, $i$ = 1, 2, …, $m$), where the weight ($\beta_i$, also known as a pollutant delivery or transport coefficient) applicable to an individual source is the proportion of its emissions that affect the ambient concentration:

$$a = \sum_{i=1}^{m} \beta_i e_i$$

Performance standards limit the emissions for firms ($e_i$), but do not limit the number of polluting firms ($m$). Thus, entry of new firms (increasing $m$), even though they comply with performance standards, may degrade environmental quality. The implication is that overall environmental conditions cannot be managed by performance standards alone. Efficiency is enhanced if entry is also regulated.

In the linear water quality model landscape-level efficiency is characterised by an adjusted form of the well-known equi-marginal principle for allocating pollution loads across sources: that each source should operate so as to have equivalent marginal (incremental) compliance costs per marginal unit of environmental impact. Mathematically, this condition is expressed

$$\frac{MC_1}{\beta_2} = \frac{MC_2}{\beta_2} = \ldots = \frac{MC_m}{\beta_m}$$

where $MC_i$ is the incremental compliance cost of firm $i$. The implication is that performance standards that minimise total compliance cost must be differentiated across firms according to their compliance costs and environmental impacts. But compliance costs are *private information* unknown to regulatory authorities. Thus, regardless of what regulators know about the relative impacts of farms on environmental conditions, they will lack information needed to design cost-effective performance standards. Thus adverse selection prevents the implementation of allocatively efficient standards.

In the case of performance taxes and a linear environmental model, the rule for cost-effectiveness is again the equi-marginal rule presented above. Economists have demonstrated that this rule will be satisfied by a differentiated tax structure with the following property:

$$\frac{t_i}{t_j} = \frac{\beta_i}{\beta_j}$$

where $t_i$ is the tax imposed per unit of emissions, or estimated emissions, by firm *i* and $t_j$ is the tax imposed per unit of emissions, or estimated emissions, by firm *j*, for any set of firms *i* and *j*. The implication of this finding, for a linear model, is that a least-cost allocation can be achieved by a differential tax structure with the differentials based on farms' relative environmental impacts. The reason is that the responsibility to evaluate trade-offs between costs and impacts remains with farmers, who view the environmental impact-based tax rate as a price signal to guide their own decisions. Unlike performance standards, the regulator does not have to perform this evaluation for farmers, and so information about individual firms' compliance costs is not needed to achieve cost-effectiveness in this case. This property is considered a major advantage of environmental taxes over environmental standards. A second benefit of the differentiated tax structure in the linear case is that taxes can effectively limit the number of farms (the variable *m* defined earlier), provided the absolute tax rates are adjusted to ensure the agency's environmental goals are met.

Things become more complex if the environmental model is more complex and includes nonlinear environmental processes and interdependent impacts across agency goals and across producers. But these complexities may not be as limiting as they were in the case of performance standards. To illustrate, suppose that the ambient pollution follows the nonlinear process

$$a = \sum_{i=1}^{m} \beta_i(e_i)e_i$$

where $\beta_i$ is now a function of $e_i$ so that each farms' emissions levels influence their environmental impacts (*e.g.* $\beta_i$ might reflect some edge-of-field uptake of nutrients which decreases as saturation occurs). Cost-effective relative tax rates in this case will depend on $e_i$, yet accurate predictions of $e_i$ can only be obtained using cost information. The asymmetric information problem therefore emerges once again, but it is potentially much less of an issue than in the case of performance-based standards. The key is to target the taxes based on estimated environmental impacts. The taxes will then encourage producers to weigh the estimated impacts against their own costs, promoting cost-effectiveness. The same cannot be said of performance standards.

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# Guidelines for Cost-effective Agri-environmental Policy Measures

Improving the environmental performance of agriculture is a high priority in OECD and many non-OECD countries. This will be of increasing concern in the future given the pressure to feed a growing world population with scarce land and water resources. Policy has an important role to play where markets for many of the environmental outcomes from agriculture are absent or poorly functioning.

This study focuses on the design and implementation of environmental standards and regulations, taxes, payments and tradable permit schemes to address agri-environmental issues. It deals with the choice of policy instruments and the design of specific instruments, with the aim of identifying those that are most cost-effective in very different situations across OECD countries.

Key conclusions from the study are that: there is no unique instrument that promises to achieve all agri-environmental policy goals; the cost effectiveness of payments systems could be improved by using performance-based measures; and policy mixes need to combine policy instruments that complement and not conflict with each other.



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