Linkages between Agricultural Policies and Environmental Effects

USING THE OECD STYLISED AGRI-ENVIRONMENTAL POLICY IMPACT MODEL

OECD Green Growth Strategy

OECD

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Foreword

This study was mandated by the Joint Working Party on Agriculture and the Environment to examine the links, using a quantitative approach, between various stylised agricultural policies and environmental outcomes. The study, which draws some general observations for policies, is based on analysis and data on four countries with different policy and agri-environmental characteristics: Finland (environmental regulations, payments and taxes in a crop farm); Japan (nutrient management in a rice/crop farm); Switzerland (nutrient management in a mixed dairy/crop farm); and the United States (conservation auctions in a corn/soy farm).

The authors are (in alphabetical order): Andrea Cattaneo, Hsin Huang, Jussi Lankoski and Hiroki Sasaki. The Secretariat wishes to thank a number of individuals who made substantive contributions to the four country case studies: Finland – Markku Ollikainen, who prepared a background paper dealing with manure policies and dairy production; Japan – Maiko Murayama, Riwako Makita, Osamu Minakawa, Yutaka Shibuya and Yasuhiko Kurashige; Switzerland – Ali Ferjani, Christian Gazzarin, Peter Kunz, Harald Menzi, and Albert Zimmermann; and the United States – Marcel Aillery.

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Executive Summary

Improving the environmental performance of agriculture is a high policy priority in OECD countries. But determining the environmental impact of agricultural policies is complicated because specific policy measures do not take place in isolation, but within a broad and evolving socio-economic and technological context. Quantitative analysis using models is not designed to exactly replicate the real world but can provide guidance on the expected environmental outcomes, which could be particularly useful in assessing the relative impacts of different policies. This can assist policy makers to better understand the linkages between policy instruments and environmental impacts, and the trade-offs or synergies involved, and therefore aid policy makers in the design and implementation of cost-effective policies.

The key policy question is to identify the change in farmers' actions that are due to specific policy interventions, and then to determine the extent to which those actions affect environmental quality. While the conceptual relationships are relatively well-established, quantitative modelling is complicated for at least four reasons:

- Biophysical processes are complex and the relationship between a given practice and its environmental outcomes is not always clear.
- Many of the environmental effects are site-specific, reflecting heterogeneous agricultural and environmental conditions, and thus some impacts cannot be extrapolated to the aggregate level through generalised policy-response coefficients.
- There are in practice a mix of policy instruments applied and multiple environmental impacts which make modelling particularly difficult.
- Many of the environmental impacts are not measured (or measurable) in monetary terms. The same agricultural production practices may produce very different bundles of commodity outputs and environmental externalities in different areas.

The conceptual and quantitative linkages between agricultural policies and environmental impacts have been analysed using the Stylised Agri-environmental Policy Impact Model (SAPIM). Developed by the OECD Secretariat, the SAPIM framework has been applied to Finland, Japan, Switzerland and the United States. SAPIM uses a combination of economic and biophysical models of representative farms (or production units) in the case studies in the countries concerned.

The SAPIM approach is pragmatic – a farmer's decision-making is analysed at the field parcel level, because this level of detail is necessary to capture the complex economic and biophysical interactions that are site-specific. SAPIM is specifically designed to capture the environmental effects of different agricultural policies through their impacts at the *intensive* margin (input-use intensity and production practices), the *extensive* margin (land-use allocation between different agricultural activities) and the *entry-exit* margin (land entering or leaving agriculture) under heterogeneous conditions.

A number of standard policy instruments are explicitly modelled: nitrogen taxes, nitrogen application standards, buffer strips, area payments and conservation auctions.

The ***Finnish study*** investigated how environmental regulations, environmental taxes and voluntary agri-environmental payments perform in the case of crop production with varying land productivity that implies different input-use intensities and adoption costs with regard to agri-environmental measures. The effects of alternative policy instruments on nutrient runoff and biodiversity were taken into account through their impact on input-use and land-allocation choices. Conservation auctions – in which farmers bid for a limited amount of conservation contracts – were also analysed.

The ***Swiss study*** examined a mixed dairy/crop farm, focusing on ammonia emissions, greenhouse gases (GHGs) and nitrogen and phosphorus surpluses. Many of the standard policy instruments on chemical fertilizer also have an impact on the amount of manure applied on crops and therefore the amount of excess manure that is then exported outside of the farm. Because nitrogen can be applied either as chemical fertilizer or as manure, the nitrogen surplus needs to be addressed by policies that influence both sources of nitrogen input.

The ***United States study*** focused on the economic and environmental performance of conservation auctions compared to the more conventional agri-environmental policy measures. Three alternative land-use types were analysed in this application – land retirement for environmental purposes (riparian buffers) and two alternative tillage methods to produce cultivated crops (no-till and conventional tillage). No-till and conventional tillage represent important cropping management choices under the working lands agri-environmental programmes. In this application the sources of heterogeneity include both differential land productivity and environmental sensitivity of the land, involving differing propensity for erosion and thus nutrient and sediment runoff.

In addition to the standard policy instruments, conservation auctions were analysed. The application of a uniform pricing auction reveals farmers' adoption costs and thus their information rent is reduced and budgetary cost-effectiveness is increased. On the other hand, a discriminatory payment gives farmers an incentive to place their bids above their adoption costs: low adoption cost farmers have a greater incentive to do so than high adoption cost farmers.

The ***Japanese study*** investigated the optimal land-use allocation and nitrogen application under a representative Japanese farm that consists of rice paddies, upland fields and land abandonment. This case study integrated paddy rice production with an upland field crop (wheat) in the same analytical framework. In general, paddy fields can provide either positive or negative environmental effects, depending on farm management practices. Consequently, the incentives provided to farmers that encourage environmentally friendly paddy rice production practices have a significant impact on the environmental effects.

In each of the four case studies, the importance of the specific policy environment was emphasised. In particular, the "policy package" is crucial as it defines the context and therefore the assumptions that must be applied in order to have a realistic representation of the impact of policies. Each of the case studies highlights different production systems, environmental issues and policy contexts. The common thread underlying all of the case studies is the impact of various policies under heterogeneous conditions. Specifically, all of the case studies have an important crop production component, in which the impact of fertilizer application is assessed in terms of crop yield and nutrient runoff. Social benefit analysis is adopted only in the Finnish and Japanese case studies, requiring monetary valuation of environmental effects (although detailed methodological discussion on monetary valuation is not conducted in this context).

In each case the analysis modelled alternative scenarios of policy options to determine the production choices and environmental outcomes that would be optimum from the perspective of producers and society (only in the Finnish and Japanese case studies). The results highlight the well-established observation that when positive or negative environmental externalities are not factored into farmers' decisions then the production choices and environmental outcomes will reflect the weighing-up of private costs and revenues by farmers. Policy intervention can potentially raise social welfare through bringing those externalities into the equation.

The analysis thus highlights the trade-offs involved – among production choices, policy instruments, economic and environmental outcomes. The value of the SAPIM approach is that a flexible framework has been developed that has the potential to be used by the policy and research communities to analyse their specific interests.

The SAPIM approach, like any other modelling approach, is subject to limitations with respect to the data, the model parameters, the economic and biophysical relationships represented. In particular, the site-specificity of agri-environmental relationships means that results cannot be readily generalised or attributed to more aggregate levels. A key source of uncertainty is arguably related to the valuation estimates of social benefits in the case studies. Nevertheless, the quantitative results in this study arising from the various scenarios modelled can be viewed and interpreted as illustrative.

The general policy lessons that can be drawn from the analysis are as follows:

- The heterogeneity of agricultural and environmental conditions makes it difficult to generalise a particular policy response to beyond where it was modelled.
- Un-regulated polluting activities should be included in policy design.
- It is important to take into consideration the existing policy environment when evaluating new policies.
- Environmental co-benefits and trade-offs should be recognised.

There has often been a lack of robust and quantitative analysis of the linkages between policy drivers and environmental outcomes in the agricultural sector. Decisions have been taken that have relied heavily on "trial and error" approaches to establish "which policies work". The approach described here is intended to redress the balance so that observed changes – for example, in nutrient runoff, or greenhouse gas emissions, or biodiversity associated with farming – can be better explained as to their cause and, in particular, their link to policy. The SAPIM approach has the potential to provide policy makers with a valuable tool to help them in designing and implementing effective and efficient policies.

Chapter 1
Introduction

Agriculture has been subject to considerable public intervention over the past half-century, perhaps more than any other economic sector (Robinson, 1989; Gardner, 1990). The provision of public support in the form of guaranteed output prices, input subsidies, deficiency payments, crop area payments, or disaster relief has encouraged and facilitated investment by farmers in production capacity expansion. While this has made it possible to achieve, *inter alia*, national production objectives, it has also been accompanied by more intensive soil tillage, increased reliance on agrochemicals, and expansion on marginal cropland. Given its associated effects upon the quality of soil, water and wildlife habitat, various authors have implicated agricultural policy as a contributing cause of environmental degradation (Libby, 1985; Pierce, 1993; OECD, 1989; Lewandrowski *et al*., 1997). Agricultural policies may also have positive effects on environment – for example, agriculture-related semi-natural habitats and open landscape, flood and drought control.

However, determining the environmental impact of agricultural policies is complicated because the actions responding to a specific policy do not take place in isolation, but within a broader and evolving socio-economic context. The first step in measuring the environmental impact of agricultural policies is linking a change in farmers' behaviour to the policy being evaluated. Because many other factors influence farmers' choices, it is critical to determine the extent to which a given policy incentive stimulated some farmers to do something that they would not otherwise have done. A second step requires assessment of how the portion of observed behaviour that can be linked back to a policy incentive then affects environmental quality – given that other factors also affect the environment (Smith and Weinberg, 2004). Even without the broader policy context, the complexity involved in assessing the environmental impacts of agriculture is illustrated by Van der Werf and Petit (2002), who review 12 indicator-based methods to evaluate environmental impacts at the farm level.

The more removed a policy instrument is from an observed environmental outcome the more challenging it will be to assess the specific policy's contribution to the outcome. For example, the role of a conservation tillage incentive payment in an observed reduction in soil erosion is likely to be easier to assess than the role of agricultural trade liberalisation. An added difficulty in disentangling the role of policies in environmental change is that many of the changes in farming that have led to environmental impacts may be linked to technological developments driven by competition in agricultural markets. The role of policy in influencing these trends is not always apparent.

Due to the difficulties in gathering empirical data and establishing policy counterfactuals, it is often the case that an *ex-ante* assessment of policies is performed using a combination of economic and biophysical models. Environmental process models can help overcome the non-point source and site-specificity complications of agricultural and agri-environmental programme evaluation by substituting predictions from models for direct observations of effects. For example, site-specific changes in (in-field) soil erosion due to particular erosion control practices can be estimated using the Universal Soil Loss Equation (USLE) and the Wind Erosion Equation. Both models provide reasonably accurate results and require only minimal data describing climate, topography, soil, and cropping information at the field level. In contrast, models of nutrient and pesticide runoff are far more complex, simulating multiple environmental effects from the transport and fate of multiple pollutants into environmental sinks.

Even with elaborate process models, the results are unlikely to match real world observations because farming practices are not the only factors that affect environmental quality. Weather variability and non-agricultural pollutants may have a considerable impact on the variation encountered in physical measurements. Thus, models can provide guidance on the expected environmental outcome, but do not substitute for an *ex-post* analysis of an impact of a specific policy.

In order to help governments in the design and implementation of good policy practice, the OECD has undertaken quantitative analysis to try to disentangle the effects of different policy measures and policy mixes on the agriculture-environment interactions. The OECD's Stylised Agri-environmental Policy Impact Model has been developed to analyse the linkages between agricultural and agri-environmental policies and their environmental effects.

The purpose of this document is to provide a general description of the SAPIM framework and four applications of SAPIM based on data from Finland, Japan, Switzerland and the United States. The structure of the report is as follows. The next chapter briefly discusses the main choices confronting the design of the SAPIM framework and provides the general conceptual framework for internalising agri-environmental externalities in a heterogeneous landscape which is the basic starting point for SAPIM analysis. Chapter 3 provides a brief literature review of the environmental effects of agricultural policies, organised according to environmental issues. Four SAPIM applications or case studies are presented in Chapters 4 to 7. Chapter 8 provides sensitivity analysis for the four case studies in order to test the robustness of results. Chapter 9 provides comparative analysis of the environmental and economic performance of alternative agricultural and agri-environmental policy instruments under heterogeneous conditions and different policy contexts. Finally, Chapter 10 concludes with some policy recommendations.

Chapter 2
General description of the SAPIM framework

This chapter begins with a brief overview of different approaches to quantifying the impact of agri-environmental policies. This is followed by a description of the general framework of SAPIM. However, each SAPIM application presented in Chapters 4 to 7 adopts an application-specific analytical and theoretical framework.

Agriculture-environment interactions: Model choices

Analytical frameworks for agri-environmental policy analysis can be usefully categorised into two dimensions of aggregation: sector and geographical. Sector disaggregation proceeds, usually within the context of a full industrial classification, from the agriculture sector as a whole, through crop and livestock activities at the next level, to individual crop and livestock enterprises at the most basic unit of production. Geographic disaggregation proceeds down from the global scale, to national, sub-national regions (which are more or less homogeneous with regard to soils, climate, and types of agricultural production), to the farm and ultimately field and sub-field level (in a precision agricultural context).

While it is technically possible to apply any analytical framework at any combination of sector and geographic disaggregation, in practice the main purpose of the model and the associated availability of model parameters and data are the main determinants of what combination is used. National and global analysis will by nature require data at the national level, or at least easily aggregated to the national level. However, analysis at this level of generality precludes accurate representation of relationships that vary spatially.

The SAPIM approach is a pragmatic one. It models a representative farmer's decision-making at the field parcel level, because this level of detail is necessary for policy analysis to capture the complex economic and biophysical interactions that are site-specific and generally characteristic of agricultural production. As will be seen in the case study country examples, this flexible approach permits detailed modelling of the interaction between agriculture, the environment, and government policy. However, a main disadvantage of the SAPIM approach is that to have an overall picture at a national level, a wide range of different representative farm models would have to be constructed to capture the full range of variation within the sector. Therefore, SAPIM results cannot be extrapolated directly to more spatially aggregate levels.

Another useful distinction to bear in mind is that between positive and normative approaches to analysis. Positive analyses attempt to develop statistically reliable relationships between observed outcomes and observed inputs. They may be guided by economic theory, but are open to results that contradict theory. Normative analysis identifies the optimal choices between a set of potential inputs and outputs assuming that economic theory, and only economic theory, is operating to guide the producers' choices. Positive approaches are most useful when the relationships are uncertain and the motivations of the agents are more obscure and less predictable – the intent is to actually discover the relationships. Normative approaches are most useful when the primary motivations are well-understood and uniformly believed to be predicted by basic economic theory. It is easy to imagine both sets of circumstances applying to different analyses relating to agriculture and environment. The SAPIM approach is normative, given that the main goal of the exercise it to evaluate the economic efficiency of alternative government policies.

General framework for agri-environmental externalities in a heterogeneous landscape

Both agricultural productivity and the supply of environmental externalities – positive externalities/public goods (such as biodiversity) and negative externalities (such as water pollution) – show significant heterogeneity due to spatial variation in the natural resource base and natural conditions. Consequently, the same agricultural production practices may produce very different bundles of commodity outputs and environmental externalities in different areas. Hence, the nature and degree of jointness between commodity outputs and environmental externalities vary spatially.

The SAPIM framework adopts an integrated approach: an economic model of decision-making on representative farms is combined with a stylised site-specific biophysical model that quantifies the impact of different policy instruments on production practices and on the multiple environmental effects. Due to the site-specific nature of many agri-environmental issues, analysis at a disaggregated level is necessary in order to capture the underlying heterogeneity of agricultural productivity and environmental sensitivity across different parcels of land. SAPIM is specifically designed to capture the environmental effects of different agricultural policies through their impacts at the *intensive* margin (input-use intensity), the *extensive* margin (land-use allocation) and the *entry-exit* margin under these heterogeneous conditions at the field parcel level.

In the SAPIM framework the environmental process functions, such as nutrient and herbicide runoff or greenhouse gas emissions, are integrated into economic optimisation models, which maximise an objective function, for example to maximise social benefits or private profits subject to resource and technical endowments, and policy incentives. Incorporation of social valuation estimates for environmental effects – when reliable valuation estimates[1] are available – provides a benchmark for policy analysis. SAPIM allows the analysis of many different types of policy instruments including area payments, input use taxes and regulations, payments for environmentally friendly production practices and technologies, green auctions and tradable permits. The results of the SAPIM modelling exercises thus have the potential to show the various environmental outcomes, farm income impacts and government budgetary expenditures as a result of different policy measures being applied in heterogeneous farm conditions, which can then be summarised in terms of outcomes for private and social benefits. The key concept is to consider the environmental effects as joint products of production, and the policies as part of the variable set which modifies returns to agricultural production.

A similar approach was used by Hochman and Zilberman (1978) in their model for analysing pollution/production trade-offs. Their model integrated physical and economic models at a disaggregate level to capture the heterogeneity of site characteristics, and then statistically aggregated the micro-units into the level needed for policy analysis. The basic model was later used in several studies examining agriculture-environment relationships (for a brief overview of different studies using this approach, see Lichtenberg, 2002). In the continuum of these studies Lichtenberg (1989; 2002) provides an excellent basis for examining farmers' input use and land allocation choices under heterogeneous land quality.

The basic conceptual framework for internalising agri-environmental externalities in a heterogeneous landscape is presented in Figure 2.1. SAPIM analysis is rooted in this type of conceptual frame. In this regard consider agricultural production under heterogeneous land productivity in a region where farms are located along rivers that drain the area. The land is divided into parcels each of which are of the same size and homogeneous in land productivity. Land productivity (θ) differs across parcels[2]. For ease of presentation suppose that they are only two crops grown in these regions, $j = 1,2$.[3] Both crops are produced under constant returns to scale technologies. Without loss of generality it is assumed that the crop 1 is better suited to lower productivity land. Output of each crop per unit of land area is a function of land productivity and fertilizer and herbicide application rates (fertilizer and herbicide per unit of land area). Production increases with respect to fertilizer and herbicide application and land quality, but with diminishing returns. By assumption, some part of the land can be left uncultivated and allocated to green set-aside or there may be exit of land from the agricultural sector to other sectors, such as forestry. The private and social profitability of green set-aside or forestry (π^0 and SW^0) are assumed to be independent of land productivity.

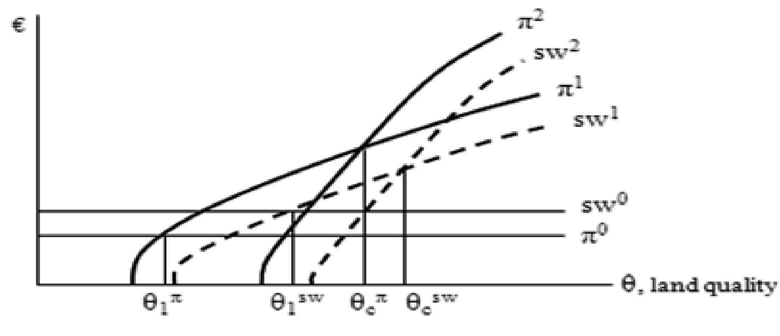
Crop production has several environmental effects:

- It affects surface water quality *via* nutrient and herbicide runoff. It is assumed that runoff for each parcel of land depends on the crop, j, and the amount of fertilizer and herbicide applied to the parcel. Both nutrient and herbicide runoff increase at an increasing rate with higher input application rates.
- It has an impact on biodiversity *via* the wildlife habitat provided by cropped areas and green set-aside. It is assumed that each type of land use provides a positive externality through its contribution to biodiversity by providing wildlife habitat. The contributions of different land uses need not be the same; hence it is assumed that biodiversity benefits generated by the region increase with the (aggregate) areas of each type of land use, but at a diminishing rate.
- Finally, cultivated crops and green set-aside may also generate both positive and negative climate externalities through sequestering carbon (green set-aside), providing offset benefits (bioenergy crops), and through generating GHG emissions related to input uses and tillage practices.

Farmers maximise their profits by applying fertilizer and herbicide inputs to each parcel at a rate such that the marginal revenue equals their respective unit prices (marginal cost). Because land productivity varies, the privately optimal rate of application for these inputs also varies over parcels and crops. Because of non-internalised environmental externalities related to water quality, biodiversity and climate, the privately optimal input application rates differ from the socially optimal ones. Accounting for nutrient and herbicide runoff damages and damage related to GHG emissions in the social optimum will therefore decrease application rates for these inputs relative to the private optimum.

Figure 2.1 illustrates the privately and socially optimal land allocation between green set-aside (π^0 and SW^0), crop 1 (π^1 and SW^1) and crop 2 (π^2 and SW^2). By assumption, negative externalities related to nutrient and herbicide runoff as well as GHG emissions are greater than the positive externalities related to the provision of wildlife habitat, and thus for both crops the social returns are lower than private profits. In the case of green set-aside, benefits related to the provision of wildlife habitat and carbon sequestration are greater than nutrient runoff damage (even though green set-aside is not fertilized, there is usually some nutrient runoff from these areas) and thus the social returns are higher than private profits. Since land productivity is homogenous within each parcel, there will be a corner solution so that the whole parcel is allocated to the land use with the highest profit or social return. This conceptual model has two critical land qualities. An entry-exit margin of crop cultivation is defined by the equality of private or social rents from cultivated crop and the corresponding rents obtained from green set-aside θ_1^{π} and θ_1^{sw}, respectively. Switching land quality allocates the cultivated land between two crops and is based on either profits θ_c^{π} or social rents θ_c^{sw}. As shown by Figure 2.1, the privately optimal land allocation between crop 1 and crop 2, θ_c^{π}, differs from the socially optimal one, θ_c^{sw}. In the social optimum more land is allocated to crop 1 which is cultivated on lower productivities of land with lower fertilizer and herbicide applications, and thus with lower nutrient and herbicide runoff damages and with lower GHG emissions related to production, transportation and use of these inputs. Moreover, relative to the private optimum, the social optimum allocates more land in environmentally friendly green set-aside.

Figure 2.1. Private and social optimal land allocation under heterogeneous land quality



Source: Lichtenberg, 2002.

The policy scenarios to be analysed in each case study are compared to these private and social benchmarks, and are meant to be representative of the instruments available to policy makers. The policy is: imposing regulations to provide indirect incentives or, if feasible, internalising positive and negative externalities, targeting them directly with payments and taxes. A policy instrument that is effective in incorporating the externalities will resemble as closely as possible the social optimum. Only if an agri-environmental policy allows differentiated payment rates to take into consideration land heterogeneity, the social optimum could be attained as the first-best policy. Due to land quality heterogeneity, to be first-best the policies should be differentiated by parcel because marginal adoption cost, benefits and damage of cultivation will differ by parcel.

It is quite difficult to determine the first-best solution, not only because of land quality heterogeneity, but also because of measuring marginal externality. As it stands, none of the policies in SAPIM are first-best solutions. The levels of policy measures (*e.g.* nitrogen tax rate, the amount of subsidy) were chosen in each case so as to replicate the selected environmental damage or compensate for the additional cost related to environmentally friendly farming.

These empirical practices are similar in concept to the seminal work of Baumol and Oates (1975), in which setting an aggregate target level for environmental externality and then finding the cost-minimising policy to attain this target reflects a more realistic decision framework for environmental policy.

Notes

1. Valuation estimates are subject to a great deal of uncertainty – the judgement of what is reliable is not considered in this report.

2. Besides productivity of land, the environmental sensitiveness of land could also be a source of heterogeneity. For example, the slopes of the parcels towards a watercourse could differ. In that case, heterogeneity would affect the parameters of the runoff function rather than the production function.

3. Land allocation choice here is between two crops, but it could also be between two different methods producing same crop, for example, conventional tillage *vs* no-till or reduced till.

Chapter 3 Environmental effects of agricultural policies: Literature review

A brief literature review is a useful starting point to putting into context the agri-environmental analysis later presented in this document. Thus, the purpose of this literature review is not to provide an exhaustive review of previous literature but rather to illustrate various methodologies used to analyse the environmental effects of agricultural policies and main results obtained in these studies. The review is organised according to the following environmental issues: soil erosion, greenhouse gas emissions, pesticides, nutrient-related water quality, livestock manure-related emissions and policy instruments controlling these emissions, and biodiversity. This is followed by a short section on the role of broad-ranging policies and multiple environmental effects.

Soil erosion

Environmental damage from soil erosion related to agriculture has long been recognised as an important externality. Lundekvam *et al.* (2003) discuss the link between increases in soil erosion and policies introduced in the 1950s and 1960s in Norway to support agricultural prices and subsidise cultivation of new land through land levelling. They estimate that land levelling increased erodibility of soils by 3-13 times. Norwegian policy has evolved towards an emphasis on environment and such policies were removed in 1985, and incentive payments for "no autumn tillage" were introduced in 1991. An erosion model adapted for Norwegian conditions (ERONOR) was used, showing the impact on soil erosion of land levelling and the subsequent introduction of no autumn tilling. The authors also emphasise how weather patterns are very important in determining soil loss.

Another study that makes use of a simulation model of runoff and erosion is the analysis of Souchere *et al*. (2003) on the impact of the European Union's Common Agricultural Policy (CAP) on soil erosion in Upper Normandy (France). They report the loss of 200 000 hectares (ha) of permanent grassland from 1970 to 2000 and use the STREAM model for the Bourville catchment to demonstrate how the observed conversion of grasslands to winter and spring crops is expected to cause an 85% increase in soil loss. The authors also report the importance of grassland location in limiting runoff. In this respect, Yang *et al.* (2004) and Khanna *et al.* (2003) also report on the importance of geographical targeting when introducing buffers under the United States Conservation Reserve Enhancement Program (CREP). The authors of these two papers compare current CREP enrolment in watersheds in Illinois (United States) to simulated least-cost solutions incorporating a hydrological representation of sediment transport.

Wu *et al.* (2004) develop a micro-level economic model incorporating environmental production functions to analyse the effect of payments for conservation tillage and crop rotations in the upper-Mississippi river basin. The authors report results for a range of payments. For example, a USD 20/acre payment would reduce wind erosion by 21% and water erosion by 10%.

The studies presented above use simulation models to assess the likely impact of future or past policies. The policies analysed are either immediately linkable to the process models being used, or if not, the agricultural policies are simply assumed to be driving current agricultural trends without trying to disentangle policy drivers from other causes of change in agriculture. Few *ex-post* analytical studies of policy impacts have been carried out. Baldock *et al.* (2002) literature review similarly reports how links are postulated for environmental impacts, which have increased over a period of years during which the policy has operated but also many other factors have been at work. Despite the intuitive appeal of these studies, the lack of a counterfactual limits the conclusions that can be drawn in terms of policies' specific contributions to environmental change.

Goodwin and Smith (2003) is one of the few exceptions where a true *ex-post* analysis of policy impact on soil erosion is carried out. Quantitative estimates of the effects of the Conservation Reserve Program (CRP) and the federal crop insurance programme on soil loss are obtained using cross-sectional county-level data encompassing agricultural production and soil erosion both before and after the introduction of the CRP and the expansion of crop insurance programmes subsequent to 1983. Observed patterns of soil erosion were obtained from USDA's National Resource Inventory (NRI). Producers are viewed as producing two types of output: crops and land allocated to CRP, each involving specific revenues and costs. The study confirms the Conservation Reserve Program significantly reduced erosion, whereas federal crop insurance and disaster relief programmes appear to have had little impact on erosion. Income supports tied to production had considerable adverse effects. The authors report that approximately half of the reduction in soil erosion attributable to CRP enrolment was offset by increased erosion induced by increases in income-supporting federal programmes. Specifically, a one-point increase in the percentage of total acres enrolled in CRP would reduce erosion by an average 0.28 tonnes per acre. Conversely, a one percentage increase in the proportion of revenues generated by direct farm programme payments would increase soil erosion by 0.125 tonnes per acre. Reinforcing the role of CRP in reducing soil erosion, Kellogg and Wallace (1995) also report on the impact of CRP on reducing annual soil erosion by 395 million tonnes in the period from 1982 to 1992. As an alternative to CRP expenditures, Plantinga (1996) in an application to Wisconsin reports soil erosion reduction benefits stemming from afforestation of marginal agricultural land once milk price support is reduced.

The CRP has been extensively studied also in terms of the environmental impact of expiring contracts, *i.e.* of land returning into production. The results of this research, typically *ex-post* at the plot level indicate that the environmental impacts of returning CRP land to crop production would be very location-specific (Unger, 1999; Dao *et al.*, 2002; Gilley and Doran, 1997; Huang *et al.*, 2002).

Another interesting analysis that decomposes observed changes in soil erosion into the impact of a specific policy and other trends is provided by Claassen *et al.* (2004). In their study the authors attempt to determine what proportion of overall cropland erosion reduction in the United States between 1982 and 1997 was actually due to conservation compliance (CC). The CC programme requires the application of approved conservation systems on Highly Erodible Land (HEL) cropland as a condition of eligibility for most farm commodity and conservation programmes.

To accomplish the task Claassen *et al.* use environmental indicators from the National Resources Inventory (NRI) and data on distribution of farms by commodity specialisation and programme payments from the Agricultural Resources Management Survey (ARMS). According to the NRI annual cropland erosion fell from 3.07 billion tonnes in 1982 to 1.90 billion tonnes in 1997, or about 40%. The proportion attributable to conservation compliance was ascertained by examining the extent to which: i) erosion reduction occurred on HEL land, ii) land use change unrelated to conservation compliance resulted in reductions in erosion, iii) erosion was reduced on farms with HEL land receiving payments (subject to CC) relative to farms with HEL not receiving payments, and iv) erosion was reduced beyond the level required for compliance. The conclusion of the study is that 295 million tonnes reduction in annual erosion (25% of total reduction) could be directly attributed to conservation compliance. Reduction in erosion is reported to be larger on farms that receive Federal farm programme payments than on farms not receiving payments, particularly on farms with wind-erodible soils.

Based on a model cash crop farm in south-west Ontario, Stonehouse and Bohl (1993) show that a one-time subsidy covering 20% of the outlay costs would induce a farmer to convert from conventional tillage to no-till. However, the study suggests that conversion to permanent cover crops such as alfalfa would require excessively high subsidies. Finally, with respect to the use of taxes, Aw-Hassan and Stoecker (1994) determined that if the off-site damages from conventional practices were taxed as high as USD 2.25 per tonne of soil loss, the area of high-yielding/high-erosion land under conservation tillage would increase significantly, while lower-yielding land would be converted to pasture. However, in a similar study, Stonehouse and Bohl (1993) show that the meaningful levels of soil erosion prevention *via* taxation are difficult to achieve and result in significant reductions in net returns.

Greenhouse gas emissions[1]

Recent studies have estimated the potential farm sector impacts of strategies to increase the quantity of carbon sequestered in agricultural soils and biomass. The general approach has been to construct a hypothetical situation in which farmers are paid to change land uses and/or production practices to store additional carbon in soils and biomass.

Antle *et al*. look separately at shifting cropland to grasses and reducing summer fallow. Three findings in Antle *et al*. are of note. First, eliminating fallow periods appears to be a cost-effective method of carbon sequestration while expanding grasslands does not — at least in eastern Montana. Over a 20-year simulation, sequestering about 7 million metric tonnes (mmt) of carbon by expanding grasslands requires an annual rental payment of USD 51.77 per acre and costs the government USD 3.15 billion (not discounted, which is more than 14 times the cost it would take by increasing continuous cropping). Second, a comprehensive GHG-mitigation strategy can include activities that sequester relatively small amounts of carbon but at a very low cost. At the highest payment level, switching from crop-fallow rotations to continuous cropping sequesters less than 19 mmt of carbon over the 20-year simulation. While this amount represents only a fraction of the sequestration potential of afforestation, two-thirds, or about 12 mmt, could be captured for an annual payment of USD 8 per acre. Finally, for a given activity, the cost of sequestering carbon can vary significantly within a region due to site-specific biophysical and economic characteristics, indicating policy makers should note the potential for cost savings if sequestration incentives are designed with flexibility to take advantage of heterogeneity among locations in each region.

There are studies that present more comprehensive assessments in which carbon sequestration is part of agriculture's larger potential to mitigate GHG emissions (Lewandrowski *et al.*, 2004; McCarl and Schneider, 2001; McCarl *et al.,* 2003).

Lewandrowski *et al*. (2004) use the U.S. Agricultural Sector Model (USMP), a spatial and market equilibrium model built to assess a wide range of economic, environmental, and policy issues of interest to US agriculture. Model-based findings reflect the provision of financial incentives to landowners for sequestering carbon through changes in land use (converting cropland to forest or grassland) and cropland management practices (adopting conservation tillage or alternative crop rotations). The main findings are that i) at a price of USD 10 per metric tonne (mt) for permanently sequestered carbon, the ERS model estimates that from 0.4 to 10 mmt of carbon could be sequestered annually from adoption of the land-use changes or management practices analysed; and at USD 125 per tonne, from 72 to 160 mmt could be sequestered, enough to offset 4 to 8% of gross US emissions of greenhouse gases in 2001; ii) the different sequestration activities studied become economically feasible at different carbon prices with farmers adopting cropland management (primarily conservation tillage) at the lowest carbon price, USD 10 per mt permanently sequestered carbon, and would convert land to forest as the price rose to USD 25 and beyond. Afforestation is preferred to conversion of cropland to grassland up through a USD 125 carbon price. Lewandrowski *et al.* also finds that the estimated economic potential to sequester carbon is lower than previously estimated technical possibilities: the *economic potential* by factoring into farmers' adoption decisions farmers could sequester up to an additional 28 mmt by adopting conservation tillage on additional lands at the top carbon price they studied, USD 125 per tonne, as opposed to soil scientists' estimate to sequester as much as 107 mmt additional carbon through conservation tillage.

In McCarl and Schneider (2001) and McCarl *et al.* (2003), farmers can adjust land uses, crop choices, and management practices in ways that increase carbon sequestration, decrease GHG emissions (*i.e.* CO_2, CH_4, and N_2O), or increase production of biofuel crops. The ASMGHG model – a market and spatial equilibrium mathematical programming framework – depicts production and consumption in 63 US regions for 22 traditional crop commodities, 3 biofuel crops, 29 livestock commodities, and more than 60 processed agricultural products. The study uses the biophysical Environmental Policy Integrated Climate (EPIC) model to calculate changes in carbon sequestration associated with Crop management activities. The authors simulate a subsidy when farmers switch to activities that reduce GHG emissions and a charge when farmers change to activities that increase emissions. For carbon valued at USD 9.60, USD 48.10, USD 96.20, and USD 480.80 per mt, net carbon sequestration is, 51.80, 146.40, 238.50, and 395.50 mmt, respectively, while net GHG mitigation is equivalent to 53.90, 154.10, 255.70, and 425.90 mmt of carbon, respectively. In terms of the balance between different greenhouse gas reduction options, at low carbon prices soil carbon sequestration, afforestation, and CH_4/N_2O emissions reduction dominate GHG-mitigation activities. At high carbon prices, the dominant mitigation activities are afforestation and biofuel production. Regardless of the value assigned to carbon, CH_4 and N_2O emissions reduction activities make a relatively small contribution to GHG mitigation.

Taken collectively, available studies suggest that the farm sector's economic potential to sequester additional carbon is significantly less than amounts deemed technically possible in soil science-based assessments. The most noteworthy divergence between soil science and economic assessments concerns conversions of cropland to permanent grasses. From a national perspective, Eve *et al.* (2000) estimate the technical potential of converting cropland to grassland at between 26 and 54 mmt of carbon per year. In contrast, economic assessments by Antle *et al.* and McCarl and Schneider find that sequestering carbon *via* this land-use change would not be competitive with other carbon-sequestering activities.

The published studies indicate that the most cost-effective mix of carbon-sequestering activities will depend on the level of carbon payment offered – or equivalently, the target quantity of total sequestration to be achieved by the programme. Across studies, changes in production practices – such as expanding no-till and shifting to carbon sequestering rotations – dominate farm sector responses at very low payment levels. Afforestation becomes the dominant sequestration activity at a carbon payment between USD 20 and USD 100 per mt, depending on the specific features of the context modelled.

Pesticides

The challenge concerning tracing the environmental impact of agricultural policies due to pesticides lies in the fact that pesticide risks cannot be adequately assessed by simply quantifying the amount of pesticide used because of the number of different chemicals with different potencies, transport characteristics, and non-target effects. Several studies exist, for example, concerning the definition of pesticide risk indicators (Levitan, 2000]; Falconer, 2002; Van der Werf, 1996). Stoate *et al.* (2001) provide an overview of empirical studies on the environmental impact of pesticides in Europe, but these are not linked to specific policies.

Dubgaard (2003) provides insight on the difficulties in assessing pesticide-related policies by analysing the achievements of the Danish Pesticide Programme. Denmark has implemented a pesticide policy that has led to the removal of about half the substances previously permitted. The attained risk reduction depends on the pesticide impact indicator used; however, most of the computed indicator values point to a reduction in the load of toxicity per hectare. The author concludes that the re-evaluation and monitoring programmes have contributed to an overall reduction in hazards from pesticides in Denmark. The author also suggests moving from an ad valorem tax on pesticides to one that is more targeted according to the environmental effect of each pesticide. In terms of the effectiveness of re-evaluating the registration of the most toxic pesticides, Villarejo and Moore (1995) come to similar conclusions of the Danish study for the case of the ban on Ethyl Parathion in California – that it was a relatively efficient tool.

The only study we found that analyses the impact of policies on observed pesticide use was the paper by Serra *et al.* (2005). The purpose of the article was to assess the impact of the 1992 CAP reforms on the use of crop protection inputs using farm-level data taken from the Eurostat Farm accounting Data Network for the period 1994-99. During this period the 1992 MacSharry reforms went into effect. The estimated elasticities suggest that an increase in both prices and area payments generate a statistically significant increase in the use of crop protection inputs. Price effects are reported to be more elastic than area payment effects. Hence, one could conclude that the reductions in price supports in favour of area payments that occurred with the 1992 CAP reforms stimulated a decline in the use of crop protection products. Serra *et al.* go on to simulate the effects of policy shocks on the use of crop protection products. In a simulation exercise, the authors assess the effects of shocking the model in accordance with the changes introduced by *Agenda 2000* and the 2003 Mid-Term Review of the CAP. As a result of the implementation of *Agenda 2000*, results suggest a reduction in the despite use of crop protection products slightly more than 3%, leaving pest damage almost unaltered. The move from the 1999 framework to the MTR scenario and the elimination of area payments in favour of decoupled payments involved more substantial changes. Their model forecasts an 11% reduction in the use of crop protection inputs as a result of this policy shift.

Remaining in the domain of simulation models, Psychoudakis *et al.* (2002) use an empirical multi-objective approach to illustrate potential reduction in agrochemical use in farming region in northern Greece. The income loss of reducing the use of four categories of agrochemicals is assessed. The solution suggests a subset of 12 cropping patterns with maximum income for the corresponding levels of agrochemical use. The solution shows that a substantial reduction of the use of agrochemicals can be achieved by changing the pattern of cropping alone. Psychoudakis *et al.* conclude that the aid provided for by EU support schemes to compensate farmers for income loss due to extensification results in a substantial reduction in the use of fungicides and insecticides.

Water quality (nutrient pollution)

Nutrients, chiefly nitrogen, phosphorus, and potassium, are important inputs in agricultural production systems. Of the three main nutrients, nitrogen and phosphorus may cause water quality problems in surface water and groundwater. In the case of agricultural nonpoint source pollution standard solutions for point source pollution, such as effluent standards and effluent taxes, cannot be applied directly, since pollution flows from nonpoint sources cannot be monitored with reasonable accuracy or at reasonable cost (Shortle and Dunn, 1986). The economic literature on the design of nonpoint source pollution control originated with Griffin and Bromley (1982). Subsequent influential contributions were made by Shortle and Dunn (1986) and Segerson (1988).

Empirical literature has analysed both the impact of resource base heterogeneity, such as differential soil productivity, on the effectiveness of uniform *vs* differentiated policy instruments, and the implications of random or stochastic factors on efficient policy design. Helfand and House (1995) analyse non-point source pollution control under heterogeneous conditions in the case of nitrate pollution from lettuce production in the Salinas Valley. They find the costs of uniform input taxes to be relatively small compared to differentiated input taxes, which represent the cost-efficient solution. Fleming and Adams (1997) have analysed alternative tax policies for controlling groundwater nitrates from irrigated agriculture. They also found that the gains from spatially differentiated taxes were quite modest. Shortle *et al.* (1998) comment that these results are unusual, since many other empirical studies have shown that information-intensive and highly targeted instruments in most cases clearly outperform uniform instruments when transaction costs are not taken into account. For example, Carpentier *et al.* (1998) analysed compliance and transaction costs of reducing nitrogen runoff from dairy farms by 40%. They show that perfectly targeted standard reduces compliance and transaction costs by almost 75% compared with the uniform standard.

Mapp *et al.* (1994) analyse regional water quality impacts of limiting nitrogen use by broad *vs* targeted policies in five regions within the Central High Plains. Broad-based policies analysed include: i) limitations on the total quantity of nitrogen applied (total restriction) and ii) limitations on per-acre nitrogen applications (per-acre restriction). Targeted policies analysed include: iii) limits on the quantity of nitrogen applied on soils prone to leaching (soil targeted restriction) and iv) specific irrigation systems (system-targeted restriction). Their results show that targeted policies provide greater reduction in environmental damage for each dollar reduction in net farm income, that is, targeted policies are more cost-effective than broad-based policies. Among the targeted policies nitrogen restrictions differentiated on production systems outperform nitrogen restrictions on soil types.

Lacroix *et al.* (2005) analyse the cost-effectiveness of farm management practices for reducing nitrate pollution in the north-east of France under uncertainty and climate variability. They use a biophysical model to assess the probabilistic cost-effectiveness of the farm management practices supported by the European Union for reducing nitrate pollution. Six nutrient management scenarios are analysed and each scenario is characterised by a set of farm practices defined for controlling nitrate pollution. Four of them simulate farm practices that have been suggested by the European Union to reduce agricultural nonpoint source pollution. Analysed scenarios include *e.g.* limiting fertilizer application on the land in vulnerable zones, integrated/precision fertilisation (aims to optimise yields and reduce the mineral nitrogen in the soil by measuring soil mineral nitrogen reserve), set-aside cross compliance, 20% reduction of nitrogen application and use of catch crops etc. Lacroix *et al.* found that none of the policy scenarios tested satisfies the nitrate concentration constraint every year. In the long run the optimal policy scenario is the one that combines integrated fertilisation with planting of catch crops immediately after the harvest. Their results highlight the effectiveness of catch crops to act as buffer against climatic, cropping and soil conditions because catch crops reduce the variability of nitrogen concentrations and thus reduce the risk of exceeding the water quality or environmental constraint.

Vatn *et al.* (1997) developed an interdisciplinary modelling approach named ECECMOD to analyse the regulation of nonpoint source pollution from agriculture. They analyse the impacts of following policy scenarios on losses of nitrogen, phosphorus and soil: i) 100% tax on nitrogen in mineral fertilizers, ii) 50% arable land requirement on catch crops/grass cover, and iii) a per-hectare subsidy for spring tillage. The tax induces both reduced fertilizer levels, more clover in the leys and better utilisation of nitrogen in manure. However it does not have any effect on soil or phosphorus losses. Requirement for catch cropping reduces all categories of losses and losses of nitrates are reduced twice as much as in the tax regime. Subsidising spring tillage has a stronger effect on soil losses than the catch crop regime, but it has insignificant effect on nitrate leaching. Tax on nitrogen is the least costly measure per hectare and per kilogram reduced N leached. Catch crops are more costly, but they have positive effects on erosion and phosphorus losses as well. If the focus is exclusively on erosion then spring tillage is the least costly measure.

Abrahams (2004) develops an empirical simulation model of corn production in the United States and its impacts on nitrate pollution, and examines taxes and standards on purchased nitrogen fertilizer and taxes and standards on excess nitrogen (that is, nitrogen surplus). These environmental policy instruments are examined with and without corn price support (analogous to the Deficiency Payment Program) and land retirement policies (analogous to the Acreage Reduction Program). The results show that economically efficient nitrate policy choices are sensitive to agricultural income support programmes: in the presence of income support programmes, the preferred instrument is a fertilizer tax, whereas without the support programmes, the preferred instrument is an excess nitrogen tax.

Lankoski *et al.* (2006) analyse the social profitability of conventional mouldboard plough tillage *vs* no-till by using Finnish data. Social welfare assessment of the alternative tillage practices takes into account the potential cost savings of no-till technology, such as lower fuel consumption and labour input, as well as potential yield losses in comparison to conventional tillage. Following environmental effects are taken into account in social welfare assessment: surface runoff of nitrogen, dissolved phosphorus, particulate phosphorus, and herbicide runoff including MCPA runoff and glyphosate runoff. Their results show that only in one case (barley cultivation) out of three it is both privately and socially profitable to adopt no-till technology. In other cases (wheat and oats cultivation) costs savings due to no-till adoption are not enough to compensate yield losses it entails so that it is profitable for a farmer to continue conventional tillage. From the social profitability viewpoint, there are important environmental trade-offs related to no-till technology adoption: nitrogen runoff, soil erosion, and particulate phosphorus runoff decrease; however, dissolved phosphorus runoff increases remarkably (these runoffs are over three times those of conventional tillage). Also total runoff of herbicides may increase because of increased use of herbicides to control perennial weeds under no-till. From the social profitability viewpoint, no-till adoption is again profitable only for barley cultivation.

Livestock manure related emissions and policies

Evidence for some European countries indicates that around 95% of ammonia (NH_3) emissions into the air result from agriculture, with about 60% from animal manure (mainly cattle) and the remainder arise mainly from the use of inorganic nitrogen fertilizers (OECD, 2001; 2007b).

Yap *et al.* (2004) examine the adoption costs of shifting from nitrogen-based manure policies to phosphorus-based policies in a hog-grain farm. In addition to the conventional manure management alternatives and crop land use pattern adjustments, they consider a possibility of changing the pig diets to reduce the amount of phosphorus in manure. Using a simulation model they find that relative to the nitrogen-based policies, phosphorus-based policies did not change the number of heads. However, pigs were fed an alternative diet using enzyme phytase, and phosphorus excretion was reduced. Also, the cropping pattern of the farm shifted slightly from corn and beans cultivation towards wheat production.

Fleming *et al.* (1998) examine the net benefits of swine manure management for alternative production systems. They allow for two alternative storage systems (anaerobic lagoon and slurry basin) and analyse two alternative nutrient target levels (based either on nitrogen or phosphorus standards). Assuming that the number of heads is given, they only focus on the net benefits from manure over delivery costs. In a related model, Feinerman *et al.* (2004) examine the costs of three alternative fertilization strategies (using a chemical fertilizer, manure and both) when either a nitrogen or phosphorus standard is imposed. They calculate the costs associated with each strategy and define the price thresholds to describe shifts in the management regimes.

Kaplan *et al.* (2004) and Smith *et al.* (2006) provide empirical analysis of regional manure policies. These papers demonstrate that the effects of manure policies depend crucially on the transportation costs of manure and substitution possibilities between manure and chemical fertilizers. Manure policies typically lead to reduced production, higher prices and decreased nutrient runoff. Also, it turns out that the willingness of crop producers to adopt manure-spreading on their crop lands affects the costs of manure management.

Ribaudo *et al.* (2003) apply the model by Fleming *et al.* (1998) to investigate three alternative manure management practices (waste management, waste utilization, manure transfer) to meet the manure nutrient standards at a regional level. They also determine the crop land area required to meet the standard on the basis of the willingness of the farmers to accept manure on their crop lands. Both papers find that imposing a phosphorus standard implies much higher adoption costs than a nitrogen standard.

Biodiversity and wildlife habitats

There are some cost-effectiveness analyses of biodiversity management in the literature. These studies have estimated biodiversity management costs as a cost to the public exchequer (see, for example, Moran *et al.,* 1996; Babcock *et al.,* 1997; Macmillan *et al.,* 1998) and have used broad measures of biodiversity such as habitat area. Wynn (2002) however takes into account farmers' adoption cost when using data from ten Environmentally Sensitive Areas (ESAs) in Scotland. Biodiversity is measured at the plot and farm level. Biodiversity and adoption costs are combined in cost-effectiveness ratios. Statistically significant biodiversity and adoption cost differentials were found between farm types (cattle and sheep, mixed, specialist beef and specialist sheep). Neither high biodiversity nor low adoption costs were necessarily associated with high cost-effectiveness, which emphasises the importance of both factors.

Kleijn *et al.* (2003) have analysed the effectiveness of European agri-environment schemes in conserving and promoting biodiversity. They reviewed 62 studies originating from five EU countries and Switzerland (5), 76% of which were from the Netherlands and the United Kingdom. Other studies were from Germany (6), Ireland (3) and Portugal (1). According to Kleijn *et al.* in the majority of the reviewed studies the research design was inadequate to assess reliably the effectiveness the schemes. Overall, 54% of the examined species (groups) demonstrated increases and 6% decreases in species richness or abundance compared with controls; 17% showed increases for some species and decreases for other species, while 23% showed no change in response to agri-environment schemes.

Wätzold and Drechsler (2005) develop a simple ecological-economic model for examining the cost-effectiveness of spatially uniform *vs* differentiated compensation payments for biodiversity enhancing land-use measures. Their results show that the cost-effectiveness may be very low for uniform payments.

Broad-ranging agricultural policies and multiple environmental effects

There are situations where major shifts in agricultural policy have little or no effect on the environment. One such case is reported by Wier *et al.* (2002) in relation to the EU's *Agenda 2000* reform of the agricultural sector. The authors simulate the impact on Denmark's environment of reductions of 15% in cereal prices, 20% in beef prices, and 15% in milk prices, combined with increases in per-hectare premiums for cereals, and headage premiums for cattle. The results indicate that the *Agenda 2000* reform has significant economic costs for Denmark but almost no effects on the environment.

Similarly, Cooper *et al*. (2004) report simulation results concerning the environmental impact of the Free Trade Area of the Americas agreement, and find it to be minimal.

Farm-level modelling approaches under heterogeneous conditions

Table 3.1 summarises the studies conducted at farm or field level under heterogeneous agricultural productivity and/or environmental sensitivity. These studies have focused on non-point source pollution of nutrients, pesticides and sediment. The analysed instruments range from regulations and taxes to subsidies for abating inputs.

Table 3.1. Summary of the literature review focused on farm level modelling under heterogeneous conditions, with a focus on non-point source pollution

			Physical model		Economic model			Policy features	
Study	Study area	Environmental Factor	Scale	Emissions impact heterogeneity[1]	Control cost heterogeneity[2]	Private control cost uncertainty[3]	Static/dynamic	Instruments assessed	Asymmetric public and private information
Milon (1987)	Honey Creek Basin, OH	Pesticides and nutrient	Field	Yes	Yes	No	Static	Input standard	No
Braden et al. (1991)	Pipestone Creek, Galien River, MI	Sediment and pesticides	Field	No	No	No	Static	Input standard	No
Johnson et al. (1992)	Columbia Basin, OR	Nitrate	Farm	Yes	Yes	No	Dynamic	Estimated emissions tax/standards, Input tax/standards	No
Taylor et al. (1992)	Willamette, Valley, OR	Nitrogen, phosphorus and sediment	Farm	Yes	Yes	No	Static	Input tax, estimated emissions standards and input standards	No
Moxey and White (1994)	Northern England	Nitrate	Field	Yes	Yes	No	Static	Emissions standards, input standards	No
Helfand and House (1995)	Salinas Valley, CA	Nitrates	Field	No	Yes	No	Static	Input taxes and standards	No
Hopkins et al. (1996)	Eastern Con Belt Plain and Erie Huron Lake Plain, OH	Nitrate-nitrogen, phosphorus, erosion, pesticide	Field	No	Yes	No	Static	Input taxes, emissions taxes, emissions standard, design standards	No
Huang et al. (1996)	Kanawha, IA	Nitrates	Field	No	No	Yes	Dynamic	Input standards	No
Larson et al. (1996)	Salinas Valley, CA	Nitrates	Field	No	Yes	No	Static	Input taxes	No
Lankoski and Ollikainen (2003)	Southern Finland	Nitrogen	Farm	Yes	Yes	No	Static	No	No
Lankoski et al. (2006)	Southern Finland	Nitrogen phosphorus	Farm	Yes	Yes	No	Static	Conventional and No-till technology	No
SAPIM Finland	Southern Finland	Nitrogen, phosphorus,	Farm	Yes	Yes	No	Static	Buffer norm, Buffer payment, Input tax on N fertilizer, Targeted A-E, Auction	Yes (auction)
SAPIM Switzerland	Switzerland	Nitrogen, phosphorus, Ammonia, GHG	Farm	No	Yes	No	Static	Standard on N applic., Input tax on N fertilizer, Input tax on N applic.	No
SAPIM USA	Corn belt	Nitrogen, phosphorus, Soil erosion	Farm	Yes	Yes	No	Static	Input tax, Buffer norm, N appl. Limit, Auction	Yes (auction)
SAPIM Japan	Honshu island	Nitrogen, GHG	Farm	No	Yes	No	Static	Input tax on N fertilizer, N limit+payment, Minimum organic+ payment, per unit payment related to organic applic.	No

1. Impact heterogeneity is present if there are explicitly modelled multiple polluters and the marginal environmental impacts of non-point emissions differ by sources.
2. Cost heterogeneity is present if there are multiple non-point source units in the study area and the different source units have different costs.
3. Private control cost uncertainty is unrelated to random variations in emissions, which in a technical sense would lead to uncertainty in the level of pollution control and hence control costs. Instead, we are referring to other types of uncertainty, such as uncertainty in markets or production (*e.g.* caused by weather, or imperfect understanding of technological relationships) that affect economic returns to producers and hence the cost they incur (in terms of reduction in economic returns) in undertaking measures to reduce emissions.

Sources: Horan and Shortle (2001); OECD Secretariat.

Note

1. This part of the literature review draws on the review in Lewandrowski *et al*. (2004).

Chapter 4
Finland: Crop production and entry/exit options with forestry

This case study examines the formulation of agri-environmental policies aimed at multiple environmental outputs in the presence of spatial heterogeneity. A theoretical framework is developed that establishes socially optimal joint provision of agricultural commodities and environmental goods and services (specifically, water quality and biodiversity) as a benchmark. As regards spatial heterogeneity the focus is on differential land productivity that leads to differing fertilizer intensities and differing nutrient runoff. Moreover, biodiversity benefits are dependent on land allocation/use choices between different crops as well as between agricultural land and other land uses, such as forestry.

This framework is then used to examine conceptually and empirically the effects of different types of agricultural and agri-environmental policies commonly used in many OECD member countries, including crop area payments, fertilizer taxes, mandatory minimum width of field edges or buffer strips, payments for establishing buffer strips, and finally alternative types of conservation auctions.

The remainder of this chapter is structured as follows. First, a general theoretical framework is developed for both traditional policy instruments as well as green/conservation auctions. This is followed by a description of empirical framework, policy simulations and results for traditional agri-environmental policy instruments. Green auctions are then compared with flat-rate agri-environmental payments.

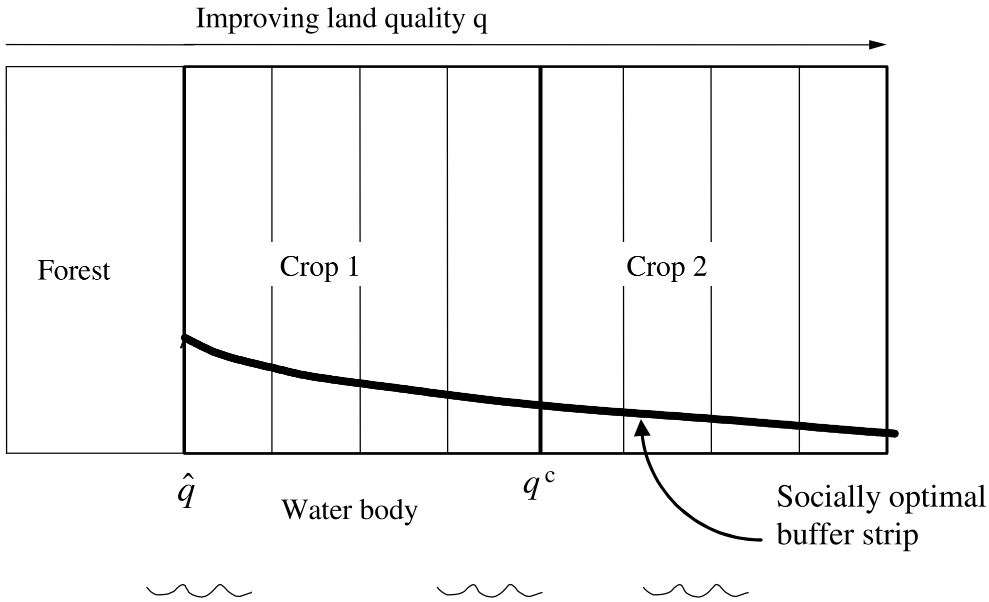
Theoretical framework

The theoretical framework and empirical specifications of the core equations used in this case study are based on Lankoski and Ollikainen (2003), Lankoski *et al.* (2004), and Cattaneo *et al.* (2007). Lankoski and Ollikainen (2003) develop an analytical framework for analysing joint production of commodity outputs and positive and negative externalities under heterogeneous land quality. They show that the optimal design of policy requires the use of spatially targeted and tailored instrument combinations. Lankoski *et al.* (2004) extend the framework by adding entry and exit margin and characterise conceptually and empirically the inefficiencies arising from the use of existing policy instruments such as crop area payments, environmental cross-compliance schemes and agri-environmental payments relative to the social optimum in the provision of multiple outputs. Cattaneo *et al.* (2007) examine to what extent jointness and environmental heterogeneity affect the performance of different auction designs *vis-à-vis* conventional flat rate policies. They analyse whether the possibility of exploiting environmental practices that provide multiple environmental benefits has an impact on the relative performance of different policy designs. They specifically focus on an agri-environmental payment programme that promotes two environmental targets, nutrient runoff reduction and biodiversity provision.

Theoretical framework for socially optimal input use and land allocation

Theoretical model presented here is based on Lankoski *et al.* (2004). The Finnish application of SAPIM describes agricultural production under heterogeneous land quality. Figure 4.1 shows the spatial structure of this application.

Figure 4.1. The spatial properties of the SAPIM



Source: Lankoski *et al.* (2004).

Following Lankoski *et al*. (2004) this study considers agricultural production under heterogeneous land quality in a region where farms are located along a river that drains the area. The land is divided into parcels which are of the same size and homogeneous in land quality (Figure 4.1). Land quality differs over parcels. It is assumed that it can be ranked by a scalar measure *q*, with the scale chosen so that minimal land quality is zero and maximal land quality is one, *i.e.*, $0 \le q \le 1$. Let *G(q)* denote the cumulative distribution of *q* (acreage having quality *q* at most), while *g(q)* is its density. It is further assumed that *g(q)* is continuous and differentiable. The total amount of land in the region is thus

$$G = \int_0^1 g(q)dq \,. \tag{1}$$

It is supposed for simplicity that there are only two crops grown in this region, $j = 1,2$. Both crops are produced under constant returns to scale technologies. Without loss of generality it is assumed that crop 1 is better suited to lower quality land. Output of each crop per unit of land area, y_j, is a function of land quality q and the fertilizer application rate (fertilizer per unit of land area) l_j, $y_j = f^j(l_j;q)$. The production function is increasing and concave in fertilizer and land quality, that is, $f_l^j(l_j;q) > 0$, $f_{ll}^j(l_j;q) < 0$, $f_q^j(l_j;q) > 0$, $f_{qq}^j(l_j;q) < 0$. Let p_j and c denote the respective prices of crops and fertilizer. It also allows for the possibility that some part of the land is left uncultivated and allocated to a residual use such as fallow or it can be allocated outside of the sector (such as forestry in the empirical application). The revenue per unit of land area generated by residual use is π^0; it is assumed that it is independent of soil quality. Let $L_j(q)$ denote the share of land of quality q allocated to use j. The total amount of land allocated to each use is thus $H_j = \int_0^1 L_j(q)g(q)dq$, $j = 0,1,2$.

Crop production has two environmental effects: it affects water quality *via* nutrient runoff and biodiversity *via* wildlife habitat provided by cropped areas and field margins. The focus here is on a special type of field margin, a buffer strip located between crop land and the river that is left uncultivated and covered by perennial vegetation. Buffer strips help reduce nutrient runoff as well as promote biodiversity. Let $m_j(q)$ denote the share of a parcel of quality q allocated to crop j that is retained as a buffer strip. Since buffer strips are maintained only on cropland, the total area of buffer strips in the region is $M = \int_0^1 \sum_{j=1}^2 m_j(q)L_j(q)g(q)dq$. Each type of land use contributes to biodiversity by providing wildlife habitat. The contributions of different land uses need not be the same; hence it is assumed that biodiversity benefits generated by the region are an increasing, concave function of the (aggregate) areas of each type of land use, $k(H_0, H_1, H_2, M)$.

Crop production also generates negative environmental externalities *via* nutrient runoff. It is assumed that runoff for each parcel of land is a function $v_j((1-m_j(q))l_j(q), m_j(q))$ that depends on the crop, j, the amount of fertilizer applied to the parcel, $(1-m_j(q))l_j(q)$, and the area of the buffer strip running the length of the parcel along the river, $m_j(q)$. For convenience, runoff from residual use is assumed to be zero. By assumption runoff is uniformly mixed in the river, so that pollution damage depends on aggregate runoff, $Z = \int_0^1 \sum_{j=1}^2 v_j((1-m_j(q))l_j(q), m_j(q))L_j(q)g(q)dq$. Let D(Z) denote the convex damage function from runoff ($D'(\cdot) > 0; D''(\cdot) > 0$).

The social welfare maximization problem can now be expressed as

$$\max_{l_j, m_j, L_j, z, H_j} \int_0^1 \left[\sum_{j=1}^2 (1-m_j(q))[pf^j(l_j(q), q) - cl_j(q)]L_j(q) + L_0(q)\pi^0\right] g(q)dq$$

$$-D(z) + k(H_0, H_1, H_2, M)$$

s.t.

$$H_j = \int_0^1 L_j g(q) dq$$

$$M = \int_0^1 \sum_{j=1}^2 m_j(q) L_j(q) g(q) dq$$

$$Z = \int_0^1 \sum_{j=1}^2 v_j((1-m_j)l_j(q), m_j(q)) L_j(q) g(q) dq$$

$$L_0(q) + L_1(q) + L_2(q) \le 1 \ \forall \ q$$

Let λ_j be the Lagrange multiplier associated with the crop land (H_j) constraint, μ the Lagrange multiplier associated with the buffer area (M) constraint, ζ the Lagrange multiplier associated with the runoff (Z) constraint, and δ the Lagrange multiplier associated with the constraint $L_0(q) + L_1(q) + L_2(q) \le 1$. Then the first order conditions defining the optimal use of fertilizer, the size of the buffer strip and allocation of land among alternative uses in the social optimum are:

$$l_j : \left[p \frac{\partial f^j}{\partial l_j} - c \right] - \zeta \frac{\partial v_j}{\partial l_j} \le 0 \tag{2a}$$

$$m_j : -(pf^j - cl_j) + \mu - \zeta \frac{\partial v_j}{\partial m_j} \le 0 \tag{2b}$$

$$L_j : (1-m_j)[pf^j - cl_j] + \lambda_j + \mu m_j - \zeta v_j - \delta \le 0 \text{, j = 1,2} \tag{2c}$$

$$L_0 : \pi^0 + \lambda_0 - \delta \le 0 \tag{2c'}$$

$$H_j : \frac{\partial k}{\partial H_j} - \lambda_j = 0 \tag{2d}$$

$$M : \frac{\partial k}{\partial M} - \mu = 0 \tag{2e}$$

$$Z : \ -D'(z) + \zeta = 0 \tag{2f}$$

plus the aforementioned set of constraints.

Condition (2a) is the standard condition that fertilizer should be applied at a rate such that marginal revenue equals its unit price plus the marginal social cost of damage from runoff. The latter is the last term on the right hand side of condition (2a), as can be seen from condition (2f), which says that the shadow price ζ equals $D'(Z)$. It should be noted that fertilizer application per unit area will vary over parcels and crops. After substitution using equations (2e) and (2f), it can be seen that condition (2b) says that the size of the buffer strip should be chosen to equate the opportunity cost of allocating land to buffer strips, which equals the loss of crop rent, $-\left[pf^j - cl_j\right]$, with the marginal agro-biodiversity benefits and reductions in runoff damage, $\frac{\partial k}{\partial M} - D'(z)\frac{\partial v_j}{\partial m_j}$. Equation (2c) defines the condition for the critical land quality dividing the land area between the two crops. After substitution from equations (2d)–(2f), it can be seen that all land of quality q should be allocated to the use with the highest social return, including crop rent

$(1-m_j)[pf^j - cl_j]$, marginal contributions to biodiversity from both the crop and buffer strip, $\frac{\partial k}{\partial H_j} + \frac{\partial k}{\partial M} m_j$, and marginal runoff damage, $-D'(z)v_j$.

Under certain regularity assumptions, condition (2c) also defines two unique critical land qualities: a minimum quality of land allocated to crop production and a critical quality at which the land allocation switches from one crop to another (see for example Lichtenberg 2002). Land of quality $q < \hat{q}_1$ is allocated to the residual use. Land of quality $\hat{q}_1 \le q < q^c$ is allocated to crop 1. Land of quality $q \ge q^c$ is allocated to crop 2.

This land allocation is illustrated in Figure 4.1. Furthermore, one can easily show by totally differentiating conditions (2a) and (2b) that fertilizer application is increasing in land quality ($\frac{\partial l_j}{\partial q} > 0$) while buffer strip area is decreasing in land quality ($\frac{\partial m_j}{\partial q} < 0$), also as shown in Figure 4.1.

In the absence of government intervention profit-maximizing farmers' decisions may not take into account either negative (runoff) or positive (agro-biodiversity) externalities from agriculture. It is easy to see from condition (2b) that farmers will not maintain buffer strips in such cases because they receive no compensation for the lost rent (hence condition (2b) holds as a strict inequality). The privately optimal fertilizer application rate similarly ignores marginal runoff damage

$$l_j : p\frac{\partial f^j}{\partial l_j} - c = 0 \tag{3a}$$

while land of each quality is allocated to the use that generates the highest rent without consideration of runoff damage or biodiversity benefits

$$L_j : pf^j - cl_j - \delta \le 0. \tag{3b}$$

These conditions have been analysed in detail in Lichtenberg (1989; 2002), and in Lankoski and Ollikainen (2003).

Background and theoretical framework for green auctions

As regards agri-environmental payment programmes in OECD countries, most of them are based on fixed flat rate payments provided to farmers who comply with a predetermined set of management practices/criteria, such as reduced tillage or limits on the intensity and timing of fertilizer, manure and pesticide applications. The obvious problem with this type of flat rate payment approach is that heterogeneity in neither farmers' adoption costs nor site-productivity of environmental goods supplied are taken into account in policy design and implementation. This may reduce cost-effectiveness of the programme even if the policy-related transaction costs are taken into account.

Auctions could reduce information asymmetry between a farmer and a policy maker and thus reduce the problem of hidden information and adverse selection. Auctions have been recently applied to environmental conservation in agriculture (Latacz-Lohmann and

Hamsvoort, 1997; Stoneham *et al.,* 2003; Vukina *et al.,* 2006). In conservation auctions, farmers bid competitively for a limited amount 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 adoption costs and as a result it will reduce farmers' information rents and improve the cost-effectiveness of an agri-environmental programme. It should be noted that conservation auctions may imply higher policy-related transaction costs than flat-rate payment approaches.

In this case study the environmental effectiveness and cost-effectiveness of conservation auctions relative to flat-rate agri-environmental payment is assessed. With regard to auction design this application focuses on an agri-environmental payment programme promoting two environmental targets, nutrient runoff reduction and biodiversity provision applying a sealed bid green auction (discriminatory first-price sealed bid auction). A score index is used to rank the farmers' applications. The score index is a product of two aspects: environmental performance and the monetary size of the bid. By giving weights to different environmental aspects in the programme, the index establishes the relations between these aspects as concerns their relative priority (like the Environmental Benefits Index [EBI] in the United States). As index weights are typically known prior the auction, the index helps farmers to assess how their actions affect their chances of getting their bids accepted. Unlike previous index applications, where the weights given to the chosen environmental aspects are based on expert assessment, the weights of runoff reduction and biodiversity conservation are developed using Finnish valuation studies concerning biodiversity benefits and nutrient runoff damages. Hence, the weights in the environmental index reflect the nature of social welfare function.

Following Cattaneo *et al*. (2007) it is assumed that the government announces an agri-environmental programme to promote nutrient runoff reduction and biodiversity provision. Farmers are asked to present a combination of environmental management actions and related bids. To guide the bids, the government indicates the weights given to the environmental performance and the size of the bid. Moreover, farmers promising to reduce fertilizer application are required to make a costly soil nutrient test and to report the actual amount of applied fertilizer to prevent moral hazard associated with non-point pollution. Drawing on the farmers' bids, a single score value (I) will be computed for each application. The applications will be accepted according to the score value in so far as the scores exceed a cut-off value (I^c), which is defined endogenously after the bids have been submitted.

To formalise this bidding procedure, it is first defined how the environmental performance of each bid is assessed. It is assumed that environmental performance includes two components: an improvement in agricultural biodiversity (BD) and in water quality by reducing nutrient runoff (BZ). In working lands (cultivated lands), biodiversity is mostly promoted by field margins, which provide semi-natural habitat for wildlife. Reducing nutrient runoff can be made by many means. The most obvious way is to reduce fertilizer application, another is provided by establishing buffer strips between fields and waterways. Also, a choice of tillage practice, such as reduced tillage or no-till may be chosen to reduce nutrient runoff.

The focus here is solely on fertilizer reduction and buffer strips as the means of reducing nutrient runoff. Fertilizer application is denoted by l and buffer strips by m. Then, biodiversity and water quality improvement benefits can be expressed relative to the maximum improvement obtainable in a given parcel as $BD(m)$ and $BZ(l,m)$, respectively. Assumption 1 characterises the environmental performance in the programme.

Assumption 1. Environmental performance, E

Environmental performance is a linear combination of biodiversity and water quality improvement benefits, $E(l,m) = \alpha BD(m) + \beta BZ(l,m)$ with $0 < \alpha, \beta < 1$ and $\alpha + \beta = 1$ and $0 < E(m,l) \leq 1$. Moreover,

A.1 $\dfrac{dE}{dl} \equiv E_l = \beta BZ_l < 0$;

A.2 $\dfrac{dE}{dm} \equiv E_m = \alpha BD_m + \beta BZ_m > 0$

From assumptions A.1 and A.2 it can be seen that decreasing fertilizer application and increasing buffer strip size reduce nutrient runoff. Moreover, there is a trade-off between them: by increasing the size of the buffer strips one can allow for higher fertilizer application in the bid for obtaining the same score. This substitution possibility plays a crucial role in this model.

Next the score value I is defined. By assumption, it depends on the environmental performance E and the payment r required by the farmer relative to the maximum payment as a function of environmental benefit provided, $R(E)$. Moreover, the score value is defined as a share of the maximum obtainable score value, denoted by $\bar{I}$. Let ω_e and ω_r denote the weights given to the environmental performance and the payment required, respectively. Like above, $0 < \omega_e, \omega_r < 1$ and $\omega_e + \omega_r = 1$. Now, the score value can be expressed as,

$$I = \left[\omega_e E + \omega_r (1 - \frac{r}{R(E)}) \right] \bar{I} \,. \tag{4}$$

Thus, equation (4) says that the score value of each bid is a share ($0 < I \leq \bar{I}$) of the maximum obtainable score value. Clearly, it depends positively on the weight given to the environmental performance and negatively on the payment required for the bid.

Farmers form their bids following the above rules. To become accepted into the programme a farmer's application's index score must be above the endogenously determined cut-off value. Obviously, the farmer's bidding strategy will be guided by expectations about this cut-off value. It is assumed that the farmers are risk-neutral, so that they focus on expected values only. Thus, the farmer will submit a bid if the expected profit from participating exceeds the profits under the private optimum. The expected profits depend on the probability of being accepted in the programme. Let $\underline{I}$ denote the minimum index value to have a chance at entering the programme. Then the probability of being accepted to the programme is defined by

$$P(I > I^c) = \int_{\underline{I}}^{I} f(I) dI = F(I) \,, \tag{5}$$

where density function $f(I)$ and distribution function $F(I)$ characterise farmers' expectations about the cut-off value. Let $\pi_0^* = pf(l^*) - cl^*$ denote the farmer's restricted profits under the privately optimal solution, with l^* the optimal fertilizer application, p crop price and c fertilizer price. Furthermore, the profits under the agri-environmental payment programme conditional on the choices of actual fertilizer rate l, buffer strip m and soil nutrient test (NC) are expressed as $\pi_1 = (1-m)[pf(l) - cl] - NC$.

Even though the soil nutrient test can be thought to reduce the moral hazard problem associated with fertilizer application, the farmer still has hidden information concerning the revenues and costs among others. This information rent (see Latacz-Lohmann and Van der Hamsvoort, 1998) is reflected in the size of the bid for the payment of providing environmental benefits to the programme. In the presence of hidden information, $r \geq \pi_0^* - \pi_1$. Recall, the environmental benefits are produced by a combination of fertilizer application and buffer strips. Therefore, it is assumed that the farmer reflects their relative impact on the bid when producing the environmental benefits by choosing l and m. More specifically, the following assumption can be made.

Assumption 2. The farmer's bid, r

The farmer's bid r, depends on the size of the buffer strip and fertilizer application:

A.3 $r = r(l,m)$ with $\dfrac{dr(l,m)}{dl} = r_l < 0$ and $\dfrac{dr(l,m)}{dm} = r_m > 0$

Assumption A.3 follows from the fact that a higher fertilizer application rate reduces, but a larger buffer strip increases the difference $\pi_0^* - \pi_1^*$ and thereby the payment required as compensation to participate in the programme. Hence, unlike previous auction studies, r reflects the inherent trade-off between buffer strips and fertilizer application.

Now, the farmer's expected profits can be expressed as,

$$E\pi \equiv \Pi = [\pi_1(l,m) - \pi_0^* + r(l,m)]F(I). \tag{6}$$

The economic problem of the farmer is to choose l and m (and thereby the bid r) so as to maximise the expected profits (6) from the bid subject to (4) and the obvious constraints $E(l,m) \leq 1$ and $r \leq R$. The Lagrangean for the problem reads as,

$$L = [\pi_1(l,m) - \pi_0^* + r(l,m)]F(I) + \lambda_r(R-r) + \lambda_E(1-E) \tag{7}$$

At an interior solution the Lagrange multipliers are zero and the first-order conditions can be expressed as

$$l^0: \quad (1-m)[pf_l - c] + r_l = -\left[\omega_e E_l + \omega_r \frac{rR_l}{R^2}\right]\frac{F'(I)}{F(I)}\Phi\bar{I} \tag{8a}$$

$$m^0: \quad -[pf(l) - cl] + r_m = -\left[\omega_e E_m + \omega_r \frac{rR_m}{R^2}\right]\frac{F'(I)}{F(I)}\Phi \bar{I}, \tag{8b}$$

where $\Phi = (1-m)[pf(l) - cl] - NC + r(l,m) - \pi_0^*$.

In both necessary conditions for the optimum, the left hand side (LHS) term indicates the economic costs of providing environmental goods to the programme and the right hand side (RHS) term indicates the expected return, that is, the effects of l and m on the score index and on the acceptance probability. It should be noted that in (8a), RHS bracket term is positive, so that that the LHS bracket term must be positive, too, and greater than r_l, which is negative. This is intuitive. The farmer reduces fertilizer application and, due to the concave response function, the value of marginal product pf_l exceeds the net costs of fertilizer use. In (5b), the RHS bracket term is negative, so that the negative LHS bracket term is greater than r_m. It should be recalled from Assumptions A.1 and A.2 that $\frac{dE}{dm} \equiv E_m = \alpha BD_m + \beta BZ_m > 0$, which indicates that buffer strips really perform in equation (8b) the double function of promoting biodiversity and water quality improvement at the same time.

Empirical applications using Finnish data

This application utilises data from the Uusimaa and Varsinais-Suomi provinces in southern Finland. The economic data originate from regional economic and employment development centres, while the ecological data come from studies conducted on the catchment area that approximately corresponds to these two provinces. The cultivated agricultural land in the region was 402 300 ha in 2002, which represents approximately 20% of cultivated land in Finland. The average farm size in 2002 was 38 ha. Agriculture in the region is predominantly crop production. The most representative crops in 2002 were barley (31%), spring wheat (26%), oats (16%), and rape (6%) (*Yearbook of Farm Statistics,* 2003). Spring wheat and rape were selected as modelled crops on the basis of their relative differences with regard to fertilizer intensity and biodiversity. The predominant soil type in this region is clay and the predominant tillage method is conventional tillage (mouldboard plough tillage). About two thirds of the total cultivated land in the region is drained with subsurface drains (Finnish Field Drainage Centre, 2002).

The Finnish application of SAPIM model consists of a quadratic Nitrogen response function, an exponential nitrogen runoff function, a damage function from nitrogen runoff, and agro-biodiversity valuation function. The core equations of this application are based on empirical specifications developed in Lankoski and Ollikainen (2003) and Lankoski *et al.* (2004).

The private profits from agriculture in the absence of government intervention are

$$\pi^i = (1-m_i)\left[p_i\left(a_i + \alpha_i l_i + \beta_i l_i^2\right) - cl_i - wn_i\right] - rk_i \quad \text{for } i = 1,2, \tag{9}$$

where m_i denotes the buffer strip as a share of parcel (the minimum width of the buffer strip is 0.5 m which corresponds to normal field edge), p_i refers to the prices of crops and c to the fertilizer (nitrogen) price, w to wage rate per hour and r to the cost of capital. It is assumed in (9) that cultivation requires employing per parcel a constant amount of labour input (measured in working hours) and capital, and they are denoted by n_i and k_i, respectively. Thus, the wage cost per parcel is fixed (as working hours are fixed) and depends on the actual cultivated share of the parcel. Capital cost is another fixed cost term but it is independent of the size of the buffer strip. Both fixed cost terms affect the analysis: the size of the buffer strips is dependent on labour costs, and both labour and capital will affect directly land allocation and, hence, the social optimum.

The model employs a quadratic Nitrogen response function, $y_i = a_i + \alpha_i l_i + \beta_i l_i^2$, which has been estimated for rape (crop 1) and spring wheat (crop 2) in clay soils by Heikkilä (1980) and Bäckman *et al.* (1997), respectively. The land quality is incorporated into the response function through the intercept parameter a_i and slope parameter α_i by calibrating the Nitrogen response function to reflect actual yields in clay soils in southern Finland in years 2000-02.

$$\begin{aligned} a_1 &= e_0 + e_1 q \qquad & \alpha_1 &= \mu_0 + \mu_1 q \\ a_2 &= h_0 + h_1 q \qquad & \alpha_2 &= \eta_0 + \eta_1 q \end{aligned} \tag{10}$$

All prices and costs are from year 2002 (see Table 1 for parameter values). For the estimation of labour and capital costs a standard activity set for field operations is developed: primary tillage, seedbed tillage, planting, and herbicide application.[1] Labour cost is based on estimated hours/ha for different operations and farmer's wage rate per hour. Capital cost is based on machinery required for aforementioned field operations and machinery expense per hectare (which is measured by depreciation cost).

Besides rents from agriculture, π^i, the social welfare function contains nitrogen runoff damages and agro-biodiversity benefits. The social welfare function for agriculture is expressed as

$$SW = \int_0^1 \sum \pi^i - 3.57Z + 54M^{0.0977} \tag{11}$$

In the second term Z denotes the total nitrogen runoff and the social value of marginal damage from nutrient runoff is EUR 3.57 per kg of N runoff, which is estimated on the basis of Yrjölä and Kola (2004),[2] who used the contingent valuation method to reveal the Finnish consumers' attitudes towards multifunctional agriculture and agriculture in general and their willingness to pay for them.[3] Linear damage estimate implies that marginal damage from nutrient runoff is constant. Per parcel nitrogen runoff function is $z_i = [1 - m_i^{0.2}]\phi\, e^{-0.7[1-0.01(1-m_i)l_i]}$. The first term on the right-hand side of equation, $1\text{-}m_i^{0.2}$, models nitrogen uptake by buffer strips. The calibration is based on Finnish experimental studies on grass buffer strips (Uusi-Kämppä and Yläranta, 1992; 1996, Uusi-Kämppä *et al.*, 2000). The term $\phi\, e^{-0.7[1-0.01(1-m_i)l_i]}$ represents nitrogen runoff from crop *i* generated by a nitrogen application rate of l_i per hectare when buffer strips take up a share of land m_i. The parameter ϕ calibrates this expression so that it equals the level of nitrogen runoffs generated by a nitrogen application rate of 100 kg per hectare in the absence of buffers strips (Simmelsgaard, 1991). On the basis of Finnish experimental studies on the runoff of nitrogen (Turtola and Jaakkola, 1987; Turtola and Puustinen, 1998), the parameter ϕ is set at 15 kg N/ha.

The third term denotes the agro-biodiversity valuation. The buffer strip areas are linked to species diversity with the help of a study by Ma *et al.* (2002). They describe the relationship between floral species richness and buffer strip area by $S = \psi\Delta^{\varphi_\alpha} W^{\varphi_\beta}$, where $\varphi_\alpha\,(\varphi_B)$ is an estimate for the average change in species richness due to an increase in the length (width) of the area while keeping the width (length) of the area constant ($\psi = 1.6331$, $\varphi_\alpha = 0.0009$, $\varphi_\beta = 0.0977$). Since the length of the area is fixed in this application, the buffer strip size *m* uniquely defines the width of buffer strip and thus, after having solved for *m*, the floral species richness can be assessed by using the coefficients estimated by Ma *et al.* (2002). The estimate for agro-biodiversity valuation function is given in terms of buffer strip hectares and taken from Yrjölä and Kola (2004), which suggests EUR 54 as average WTP per hectare for biodiversity.

Non-agricultural land use in this application is forestry. This is an obvious choice, as forests are the natural cover of the Finnish landscapes. It is assumed that if a parcel of forest is converted to agriculture, there is a lump sum conversion cost but the yields obtained from this converted land will reflect typical agricultural yields. If a previously cultivated land is forested, it will take a long time for this parcel to produce regular forest income. From Finnish studies an estimate of EUR 47.8 per hectare annual forest income over one rotation period of trees in reforested agricultural land has been obtained. Hence, this annual forest income EUR 47.8 per hectare provides reference profits for entry and exit margin of agricultural land.

Overall landscape diversity is measured *ex post* using the Shannon Diversity Index, $SHDI = -\sum_{i=1}^{n}(P_i * \ln P_i)$, where P_i equals the proportion of the region covered by patch type *i* (see for example Eiden *et al.*, 2000). There are four patch types in the model: the areas allocated to forest, rape (crop 1), wheat (crop 2), and the total buffer strip area. Since over 80% of land area in Finland is covered by forest, changes in the forest cover are assumed to contribute nothing to the social value of biodiversity. Landscape diversity is not included in the maximization procedures.

Parameter values for the model are reported in Table 4.1. The arable land area is assumed to be 40 ha (the width of the field area, that is, the distance from the water border to the other edge of each parcel is 200 m and the length, that is, the border along the waterway, is 2 000 m, so that the length of each parcel is 50 m). The base case of the model represents the private market solution (without taxes and subsidies) for cereals and oilseeds in Finland in 2002. Average prices of marketing year 2002 within the European Union are used. They may differ somewhat from the world market prices; however, as they are used in all calculations, such differences do not cause any bias when comparing alternative policy interventions.

Policy simulations and results

The model is used to estimate government budget outlays and social welfare as well as crop production, nitrogen runoff, and biodiversity under several policy scenarios.

Policy alternatives

Two benchmark scenarios are used to compare performance of alternative agri-environmental policies: i) the private optimum, obtained assuming producers maximise profits ignoring positive and negative externalities; and ii) the social optimum incorporating positive and negative externalities in the computation of societal benefits and costs. The policy scenarios to be analysed are compared to the private and social benchmarks, and are meant to be representative of the instruments available to policy makers. The options range from imposing regulations, to providing indirect incentives or, if feasible, internalising positive and negative externalities targeting them directly with payments and taxes, as follows:

Table 4.1. Parameter values in the numerical application

Parameter	Symbol	Value
Price of rape	p_1	EUR 0.255/kg
Price of wheat	p_2	EUR 0.13/kg
Price of nitrogen fertilizer	c	EUR 1.2/kg
Parameter α		
Basic level of response for rape	μ_0	9.72
Basic level of response for wheat	η_0	30.8
Slope of the response change for rape	μ_1	0.01
Slope of the response change for wheat	η_1	0.05
Parameter β		
Parameter of quadratic Nitrogen response function	β	-0.0324 for rape -0.094 for wheat
Parameter a		
Initial level of productivity for rape	e_0	700
Initial level of productivity for wheat	h_0	680
Slope of the productivity change for rape	e_1	10
Slope of the productivity change for wheat	h_1	23
Nitrogen leakage at average nitrogen use	ϕ	15 kg/ha
Farmer's wage rate per hour	w	EUR 11.35/h
Farmer's labour input per hectare	n	6.57 h/ha
Capital cost	rk	EUR 144/ha

All prices and costs are from the year 2002. The price of nitrogen is calculated on the basis of a compound nitrogen, phosphorus, potassium (NPK) fertilizer.

Source: Lankoski *et al.* (2004).

- Regulatory means – imposing a minimum buffer size on all cultivated parcels (7.1 metres [m] wide).
- Indirect instruments – internalising costs and benefits external to the private market:
  - Tax on polluting inputs – reflecting the polluter pays principle but applying it indirectly using a nitrogen tax (tax rate 29%)
  - Buffer payment – paying a fixed rate per metre of buffer strip up to a depth of 10 m (EUR 1.5 per metre).
- Targeted agri-environmental incentives – providing a payment that is proportional to the environmental benefit associated with agricultural biodiversity (compensation for providers of a public good) combined with a runoff tax based on damage estimates from nitrogen runoff (polluter pays principle). Tax is 96% of runoff damage and payment is 100% of biodiversity benefit.[4]

The levels of policy instruments were chosen in each case so as to replicate the aggregate nitrogen runoff of the social optimum solution. Thus, these levels may not correspond to the one that leads maximum social profit for each instrument since biodiversity benefits are not taken into account when setting the level of policy instrument. The policy scenarios described above assume private profit maximisation by producers. The results are compared to the private and social optimum without any corrective policies. A policy instrument that is effective in incorporating the externalities will resemble as closely as possible the social optimum. At the limit, if a policy is the first-best solution to account for the externalities involved, the outcome for that policy should replicate exactly the social optimum. As it stands, none of the policies proposed here are first-best solutions. Due to land quality heterogeneity, to be first-best the policies should be differentiated by parcel because marginal adoption cost and benefits of cultivation and buffer provision will differ by parcel. The assumption made here is that, due to transaction costs (information gathering and monitoring costs), policies are most likely to be set uniformly across heterogeneous landscapes.

As noted before, agri-environmental policies are rarely imposed in a setting where other agricultural policies are environmentally neutral. For this reason we repeat the scenarios presented above assuming that farmers receive crop area payments of EUR 50 per hectare of cultivated area. This policy does not distort relative production between the two crops; however, it affects the entry-exit margin since it is not provided if land is used in forestry.

For an analysis of the private and social profitability of flat-rate agri-environmental payments *vs* green auctions, please see the section below.

Comparing private and social optimum

First the impact of market failure on land use is analysed both in terms of extensive and intensive margins. Thanks to higher productivity of wheat, the private optimum favours wheat production (20 parcels) relative to rape (3 parcels), and leaves a substantial share of lower quality land in forest (17 parcels). The social optimum on the other hand favours less intensive use of fertilizer on an expanded agricultural area (Table 4.2). Forest area is reduced (12 parcels) and production of rape is expanded because it makes less intensive use of fertilizer than wheat (Table 4.3).

Table 4.2. Input use and land allocation: Comparing private and social optima

Policy	Fertilizer use, (kg/ha)		Buffer strips, (m)		Numbers of forest parcels	Numbers of rape parcels	Numbers of wheat parcels
	Rape	Wheat	Rape	Wheat			
Private optimum	80.3	122.8	-	-	17	3	20
Social optimum	72.6	116.5	10.0	9.8	12	14	14

Source: Author's calculations.

From an environmental standpoint, whereas no buffers are introduced under the private optimum and nitrogen runoff is high (367 kg), once externalities are factored in the social optimum as shown by equation (11), 10-m buffers are put in place (slightly less for wheat areas because of higher opportunity cost of not cultivating) and the total nitrogen runoff is reduced by over 50% to 170 kg. Similarly, the species richness improves nearly threefold, and a more diverse landscape is provided (as denoted by a higher Shannon Diversity Index, Table 4.3). The implication, for this Finnish case, is that extensive agriculture is the preferred outcome once biodiversity and nitrogen runoff damages are taken into consideration.[5] Although food security considerations are not specified as an objective, a side effect of the social optimum is a better balance between the two crops, and a broader agricultural area on which intensification could be possible if a shock in the commodity markets were to occur.

Table 4.3. Production and environmental effects: Comparing private and social optima

Policy	Total production, (kg)		Nitrogen runoff, (kg)	Species richness, (head)	Shannon Diversity Index
	Rape	Wheat			
Private optimum	4 431	78 682	367	15	0.90
Social optimum	19 205	52 698	170	44	1.21

Source: Author's calculations.

Quantifying in monetary terms the trade-off between profits and non-market benefits, the results indicate that the social optimum entails a 13% decrease in profits, whereas runoff damages are reduced by 54% and biodiversity benefits increase by 250%.[6] In the balance, social welfare increases by 60% under a social optimum, thanks to a more extensive approach to agriculture (Table 4.4).

The impact of agri-environmental policies in isolation

The simplest policy instrument to implement, namely imposing a *minimum buffer* size, does not affect fertilizer intensity nor does it significantly alter entry and exit into agriculture relative to the private optimum. The functioning of this instrument relies on the imposed buffer to reduce aggregate fertilizer use and to capture nitrogen runoff while providing biodiversity benefits. Providing *incentives to introduce buffers* by paying for a flat rate per metre of buffer (EUR 1.5 per metre) up to a certain depth (10 m) is similar to the buffer norm in terms of impact on land use with the exception of some additional entry into agriculture since these payments are received only if a parcel is under crops. Whereas the *minimum buffer* or the buffer payment does not affect the intensive margin, the nitrogen tax (29% fertilizer tax) acts directly on fertilizer intensity. In so doing, the profitability of the two crops is affected, both in absolute and relative terms, leading to considerable exit from agriculture (26 parcels in forestry, Table 4.5). In relative terms, the nitrogen tax favours the production of rape, as it uses fertilizer less intensively than wheat.

Table 4.4. Profits and social welfare: Comparing private and social optima

Policy	Profit	Runoff damage	Biodiversity benefit	Social welfare	SW/SO*
	(EUR)				
Private optimum	2 869	1 309	322	1 882	0.62
Social optimum	2 498	607	1 127	3 019	1.00

* SW/SO represents the social welfare ratio of the scenario relative to that of the social optimum.
Source: Author's calculations.

Table 4.5. Input use and land allocation: Agri-environmental policies in isolation

Policy	Fertilizer use, (kg/ha)		Buffer strip, (m)		Number of forest parcels	Number of rape parcels	Number of wheat parcels
	Rape	Wheat	Rape	Wheat			
Private optimum	80.3	122.8	-	-	17	3	20
Social optimum	72.6	116.5	10.0	9.8	12	14	14
Buffer norm	80.4	122.8	7.1	7.1	18	2	20
Buffer payment	80.2	122.8	10	10	15	5	20
Nitrogen tax	60.9	110.5	-	-	26	9	5
Targeted A-E	73.1	116.7	8.6	9.5	12	14	14

Source: Author's calculations.

Targeted agri-environmental incentives (A-E), which take into consideration both the benefits of reducing runoff (charge 0.96 of runoff damages) and of improving biodiversity (by providing full payment of biodiversity benefits), is the policy instrument that most resembles the social optimum in terms of fertilizer intensity, and of land-use allocation (Table 4.6).[7] Where the targeted agri-environmental incentives scenario deviates from the social optimum is in the size of the buffer strips, with rape cultivation using a smaller buffer than is socially optimal. Despite this deviation, A-E payments operate through a balanced combination of land-use change incentives (entry-exit from agriculture), fertilizer-use intensity, and buffer adoption.

The fact that for targeted agri-environmental incentives case buffers are installed more extensively for parcels cultivated in wheat rather than in rape is slightly counter-intuitive as one would expect land to be withdrawn from production (as buffer) on parcels with lower opportunity costs (rape). The explanation of this outcome is that it is relatively more profitable to manage runoff from rape by reducing fertilizer intensity, while for wheat beyond a given level of reduction in fertilizer intensity it is more profitable to limit nitrogen damages by introducing a larger buffer rather than further reduce intensity.

Table 4.6. Production and environmental effects: Agri-environmental policies in isolation

Policy	Total production, (kg)		Nitrogen runoff, (kg)	Species richness (head)	Shannon Diversity Index
	Rape	Wheat			
Private optimum	4 431	78 682	367	15	0.90
Social optimum	19 205	52 698	170	44	1.21
Buffer norm	2 861	75 905	173	42	0.93
Buffer payment	6 961	74 748	172	44	1.08
Nitrogen tax	13 507	20 097	172	14	0.88
Targeted A-E	18 109	52 816	169	41	1.16

Source: Author's calculations.

The production effects of the four agri-environmental policy instruments are radically different: on the one hand the buffer norm resembles a slightly scaled down private optimum in terms of quantities produced, while on the other hand targeted agri-environmental incentives provide an outcome closer to the social optimum (Table 4.6). The buffer payment favours rape production, but is still in the general area of the private optimum. The nitrogen tax is far from both private and social optima, with aggregate production considerably reduced, and rape being favoured in relative terms because it makes less intensive use of fertilizer.

The nitrogen tax is also the least desirable when it comes to species richness, which is even less than in the private optimum, due to the reduction of land in agriculture under this scenario. Furthermore, land use diversity is also at its lowest when the nitrogen tax is applied (Table 4.6). The buffer norm and targeted agri-environmental incentives both perform well in terms of species richness and landscape diversity, with targeted A-E incentives best approximating land use diversity of the socially optimum solution.

Farmers' profits under the different policy instruments can be interpreted as an indication of the extent to which famers might favour the different policies. In this respect, the farmers' preferences would be for targeted agri-environmental incentives followed by buffer payments (Table 4.7), both with post-payments profits that are higher than the private optimum (subject to government budget constraints). The buffer norm follows in third place, but entails a decrease of more than 10% in profits relative to the private optimum without constraints.

Table 4.7. Profits, budget outlays and social welfare: Agri-environmental policies in isolation

Policy	Profit (EUR)	Profit + payments (EUR)	Budget outlays (EUR)	Runoff damage (EUR)	Bio-diversity benefits (EUR)	Social welfare (EUR)	SW/SO
Private optimum	2 869	2 869	-	1 309	322	1 882	0.62
Social optimum	2 498	2 498	-	607	1 127	3 019	1.00
Buffer norm	2 562	2 562	-	617	880	2 825	0.94
Buffer payment	2 578	2 953	375	612	1 007	2 973	0.98
Nitrogen tax	2 588	2 202	-386	613	196	2 171	0.72
Targeted A-E	2 531	3 045	514	604	1 082	3 009	1.00

Source: Author's calculations.

In terms of farmers' interests, the ease of implementation of the policy instrument should also be considered. Whereas the A-E incentives envisioned here require considerable information gathering in terms of externalities and monitoring of impacts, a buffer norm or a buffer payment are easy to implement and monitor, requiring very little information. This is an element of interest to policy makers designing the programmes, but may also affect farmers' perception of which option is preferable, that is their willingness to accept a buffer norm or targeted A-E incentives package.

The nitrogen tax would probably also not appeal to policy makers because by providing for very few biodiversity benefits the social welfare improvement obtained is quite limited relative to the private optimum and considerably below the other two policy options (Table 4.7). These results suggest that, since a nitrogen tax affects mainly the intensive margin and targets only one objective, it should be paired with another policy instrument affecting the extensive margin and targeting biodiversity.

From an aggregate welfare perspective, the A-E incentives approach comes very close to the social optimum.[8] However, taking into consideration how much simpler the buffer payment and buffer norm are to design and implement, the social welfare obtained by providing a buffer payment (EUR 2 973) or imposing a minimum buffer (EUR 2 825) is surprising. From a practical standpoint, this means that if the additional transaction costs (information and monitoring) of setting up the targeted A-E incentives programme are greater than EUR 0.9 per hectare then a buffer payment may be a more attractive choice in terms of aggregate social welfare. Even the buffer norm could outperform the targeted A-E incentives if the additional transaction costs of the latter exceed EUR 4.6 per hectare, with the additional advantage of requiring no budget outlays from the government.

The impact of agri-environmental policies in the presence of area payments

Even in the current policy environment where efforts are being made to decouple agricultural support from production, agri-environmental policies, although increasing in importance, remain relegated at the margin of many countries' agricultural policy portfolios. Here "agricultural support payments" are characterised as area payments (AP), which do not affect relative returns of different crops, but do impact the extensive margin (entry-exit from agriculture). Land use under crop area payments (Table 4.8) highlights that they entail a significant shift into agriculture, transferring into rape production all parcels that were previously under forest in the private optimum. Wheat production is not affected. The expansion of the extensive margin entails a decreased fertilizer intensity stemming from production on lower land quality parcels.

Table 4.8. Input use and land allocation: Interaction between area payments and AEPs

Policy	Fertilizer use, (kg/ha)		Buffer strip, (m)		Number of forest parcels	Number of rape parcels	Number of wheat parcels
	Rape	Wheat	Rape	Wheat			
Private optimum	80.3	122.8	-	-	17	3	20
Social optimum	72.6	116.5	10.0	9.8	12	14	14
Area payment (AP)	79.0	122.8	-	-	0	20	20
AP + buffer norm	79.0	122.8	7.1	7.1	0	20	20
AP + buffer payment	79.0	122.8	10	6.5	0	20	20
AP + N-tax	59.5	110.5	-	-	7	28	5
AP + targeted A-E	71.5	116.3	7.9	7.4	0	27	13

Source: Author's calculations.

The four agri-environmental instruments cannot counteract the shift induced by the area payments. The impact of imposing a minimum buffer policy or providing a buffer payment is limited because land allocation and fertilizer intensity are not affected. Area payments limit the effectiveness of the buffer payment for high productivity parcels (the ones in wheat) because the higher opportunity cost of introducing a buffer is compounded with that of losing the area payment for th e share under buffer.

The nitrogen tax does counteract in part the impact of area payments by shifting back into forestry a subset of the affected parcels, and favouring rape production. Rape production is also favoured by the targeted agri-environmental incentives option, which also leads to the introduction of buffers, and lower fertilizer intensities than with area payments alone. However, none of these policies come even close to the social optimum, as can be seen by the amounts of commodities produced and nitrogen runoff (Table 4.9).

The over-production associated with area payments leads to a 56% increase of nitrogen runoff relative to the private optimum. Even with a minimum buffer size or targeted agri-environmental incentives, which are most effective in limiting the increase in runoffs, the runoffs are approximately 110 kg and 77 kg, respectively, above the social optimum. The buffer payment, which in the absence of area payments performed similarly to imposing a minimum buffer, does not perform as well in reducing runoffs. The nitrogen tax performs particularly poorly in terms of runoff damages. Finally, also in terms of land use diversity, all scenarios with area payments are inferior to the private optimum due to the shift away from forestry and into agriculture.

Table 4.9. Production and environmental effects: Interaction between area payments and AEPs

Policy	Total production, (kg) Rape	Total production, (kg) Wheat	Nitrogen runoff, (kg)	Species richness, (head)	Shannon Diversity Index
Private optimum	4 431	78 682	367	15	0.90
Social optimum	19 205	52 698	170	44	1.21
Area Payment (AP)	27 580	78 682	572	16	0.69
AP + buffer norm	26 606	75 905	280	44	0.82
AP + Buffer payment	26 201	76 197	318	45	0.84
AP + N-tax	38 955	20 097	372	15	0.81
AP + Targeted A-E	35 780	49 703	247	45	0.77

Source: Author's calculations.

While area payments clearly improve farmers' after-payment profits, pre-payment farm profits from production are considerably below the private optimum, and below the social optimum except for the case of the nitrogen tax (Table 4.10). The implication of these reduced on-farm profits is that, with area payments, scarce government funds are being used to shift agricultural production towards a lower welfare solution even by private optimisation criteria. When also taking into account the added damages from nitrogen runoff associated with area payments, which greatly counteract the biodiversity improvement from expanding agriculture, the reduction in social welfare relative to the private optimum (47% reduction) is an indication that area payments are not an efficient use of government funds.

Table 4.10. Profits, budget outlays and social welfare: Interaction between area payments and AEPs

Policy	Profit	Profit + payments	Budget outlays	Runoff damage	Biodiversity benefits	Social welfare	SW/SO
	(EUR)						
Private optimum	2 869	2 869	-	1 309	322	1 882	0.62
Social optimum	2 498	2 498	-	607	1 127	3 019	1.00
Area Payment (AP)	2 491	4 491	2 000	2 044	560	1 007	0.33
AP + buffer norm	2 199	4 129	1 929	1 000	1 558	2 757	0.91
AP + buffer payment	2 170	4 583	2 413	1 135	1 428	2 463	0.82
AP + N-tax	2 513	3 385	872	1 328	462	1 648	0.55
AP + Targeted A-E	2 198	4 776	2 647	883	1 572	2 818	0.93

Source: Author's calculations.

Although the agri-environmental instruments considered in this paper cannot remedy the distortion introduced by area payments, the social welfare outcome for the buffer norm and for the targeted A-E incentives is much better than with area payments alone. In fact, these two policy options generate a social welfare that is less than 10% below the social optimum. The bulk of the benefits obtained by either buffer norm or targeted A-E incentives are linked to additional biodiversity coming from the introduction of buffers for both rape and wheat parcels, combined with an expanded area under these two crops associated with the area payments. Despite these added biodiversity benefits, runoff damages are considerably higher, indicating that area payments shift the balance of non-market benefits obtained with the different agri-environmental policies.

The buffer payment, which in the absence of area payments outperformed the minimum buffer norm in terms of aggregate welfare, becomes a less preferable option when area payments are part of the policy package.

Introducing area payments significantly shifts land towards more extensive agriculture, favouring rape relative to wheat (Figure 4.2). The agri-environmental policies that were effective without area payments cannot counterbalance this shift. In the presence of area payments, the agri-environmental policies provide a different mix of benefits, from those provided without area payments (Figure 4.3). This different mix is due to the inability of these policies to counteract the push towards the extensive margin due to area payments, leading the policies to act mainly through the introduction of buffers and/or fertilizer intensity. The only option to counteract this expansion of agriculture is to impose a nitrogen tax; however, the social welfare attained by this policy is inferior to the other three policies considered.

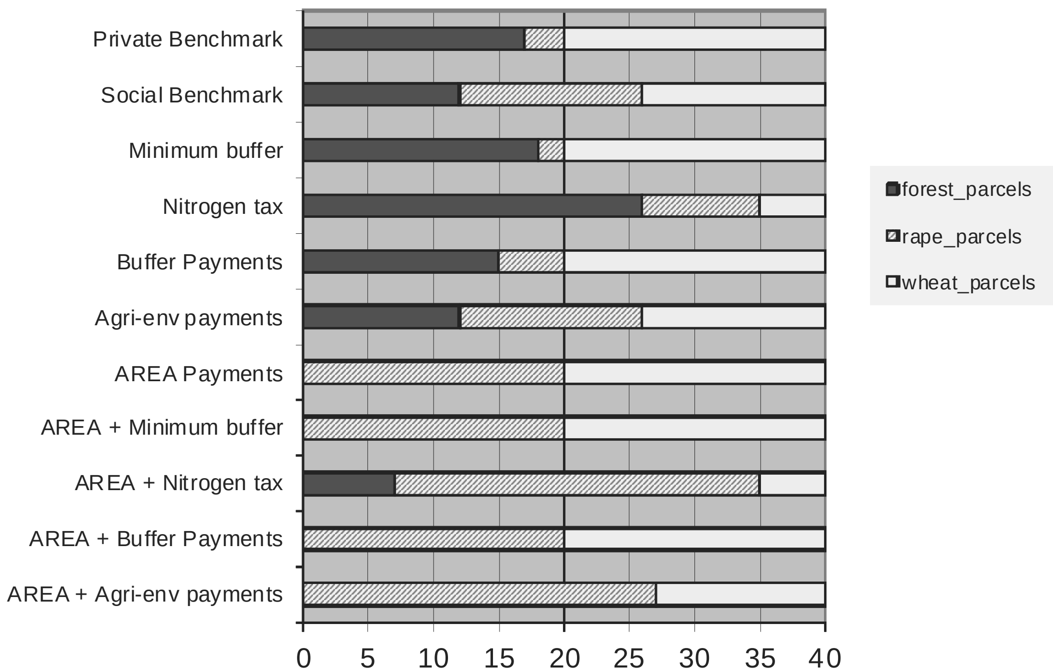
Figure 4.3 presents the contribution to social welfare of the different components (profits, runoff, and biodiversity).[9] Agri-environmental policies without area payments all have identical runoffs (by construction), with varying levels of profits and considerable differences in biodiversity benefits. The introduction of area payments affects the performance of policies in terms of the absolute and relative contribution of the three components of social welfare. First, nitrogen runoff is greater than with the A-E policy in isolation. This is not surprising, but what is surprising is that the nitrogen tax scenario, aiming to reduce nitrogen runoff damages, when combined with area payments, performs poorly in reducing runoffs compared to the other A-E instruments in the same context. Furthermore, on-farm profits are higher in the nitrogen tax scenario under area payments than for the other two instruments, which was not the case with the policies in isolation.

Despite having different land-use allocation, buffer sizes, and fertilizer intensities, the minimum buffer and the A-E payment scenarios (both with and without area payments) have very similar profiles in terms of the welfare components. The main difference when area payments are part of the policy package is the greater contribution in terms of biodiversity benefits, which counterbalances in part the negative environmental impact of greater nitrogen runoffs. When compared to the social optimum, these two solutions over-provide biodiversity and under-provide nitrogen runoff mitigation. This is not captured by the aggregate social welfare number.

Flat-rate payment approach vs green auctions

In this section an economic and environmental performance of a green auction is compared to the traditional flat-rate agri-environmental payment. The green auction mechanism relies on the weighted linear average of the environmental performance index and size of payment proposed. The environmental index in turn is a weighted linear average of nitrogen runoff reduction and agro-biodiversity promotion. The weights are empirically estimated by mapping the social welfare surface using a sample of 100 points on the surface. The lowest social welfare value was taken to coincide with the zero-value of the environmental index, and the weights were estimated using a constrained least-square estimator so that the sum of the weights equals 1. The weights obtained adopting this procedure were 0.43 for biodiversity and 0.57 for runoff reduction (for a more detailed description of the procedure and estimation results, see Annex B).

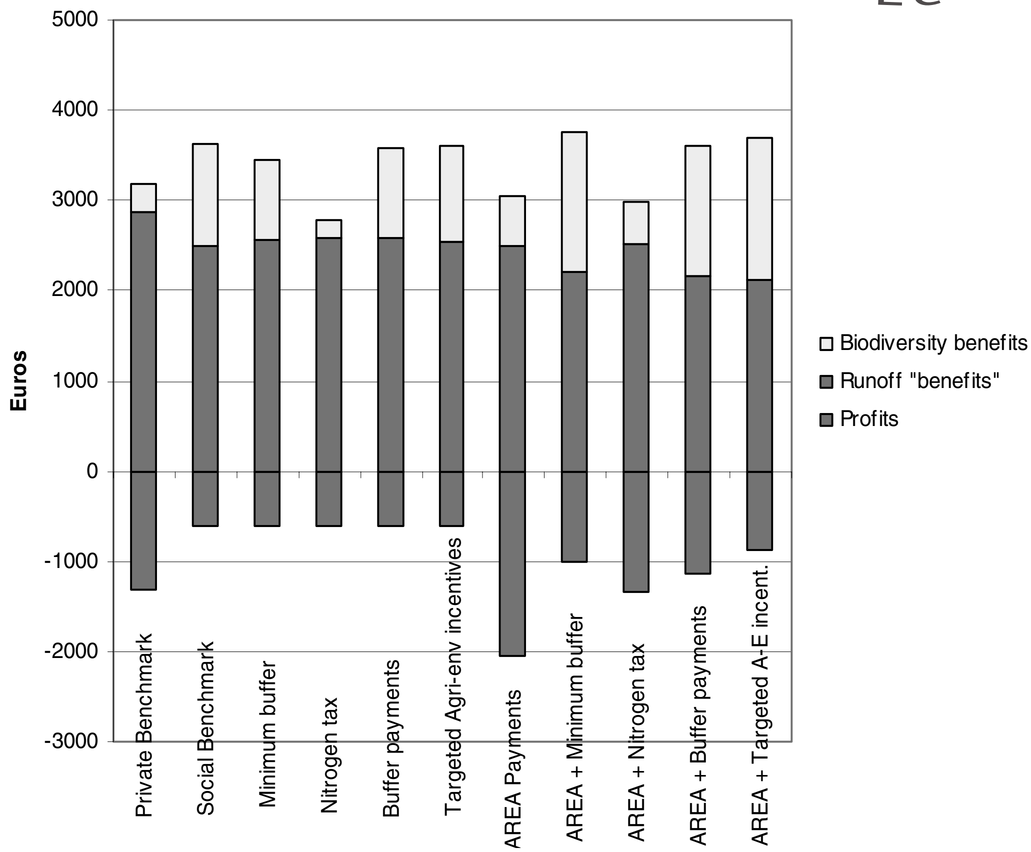
Figure 4.2. Land-use decisions under different policies (over the 40 parcels)



Source: Author.

In this analysis both land productivity and environmental sensitiveness of land are heterogeneous, so that the empirical model contains 40 differential land productivities and four different slopes of parcels towards watercourse (Table 4.11). The size of each field parcel is one hectare so that the total land area is 160 ha and it is allocated to three alternative uses: forestry, crop 1 (rape) and crop 2 (spring wheat). The empirical model is now used to estimate the outcome of the two alternative green auction designs, which are compared with a current flat-rate payment approach. Two types of green auction are examined: (1) green auction ranking by environmental score, (2) green auction ranking by environmental score and cost saving component (with a 0.2 cost-weight). The performance of green auctions is compared with the current Finnish agri-environmental payment programme that provides a flat-rate payment to participating farmers on the basis of forgone profits of establishing 3 m-wide buffer strips and reducing nitrogen intensity from privately optimal level to the maximum allowed level in the agri-environmental programme. All policy options are compared with two benchmark cases of model: farmer's private optimum (without government intervention) and the social optimum.

Figure 4.3. Contribution to social welfare under different policies



Source: Author.

The alternative policies we focus on are defined as follows:

1. Current flat-rate agri-environmental payment (Flat Rate): *A 3 m-wide buffer strip is required together with the maximum allowed nitrogen fertilizer application rate 100 kg/ha for rape and 120 kg/ha for wheat. The uniform agri-environmental payment as a compensation for forgone profits amounts to EUR 21/ha.*

2. Green auction ranking by environmental score (EnvMax): *The bids are selected according to their environmental score. The private optimum is a reference to calculate the benefits from nitrogen use reduction. The budget is assumed to be restricted to the amount of the current flat-rate payment approach described above.*

3. Green auction ranking by environmental score and cost-saving component (CostSave): *The bids are selected according to their environmental score and cost-saving component, which is given a weight 0.2. The private optimum is a reference to calculate the benefits from nitrogen use reduction. The budget is assumed to be restricted to the amount of the current flat-rate payment approach.*

Table 4.12 provides input use and land allocation results for private optimum, social optimum, and for all three policy designs. Comparison of the privately and socially optimal solutions shows that due to higher productivity of wheat, the private optimum favours wheat production (80 parcels) relative to rape (12 parcels), and leaves a substantial share of lower quality land in forest (68 parcels). The social optimum reduces nitrogen use intensity for both crops and allocates more land to rape which uses less intensively nitrogen fertilizer than wheat. Moreover, land allocation is shifted towards less polluting forest and thus, area under forest is increased from 42.5% in the private optimum to 47.5% in the social optimum. No incentives are provided in the private optimum to establish buffer strips and thus this solution only provides the normal field edge of 0.5 m width in every parcel so that total area under field edges is 0.23 ha.

Table 4.11. Additional parameter values in the numerical application of green auctions

Parameter	Symbol	Value
Nitrogen runoff		
Nitrogen leakage at average nitrogen use under 4 different field slopes (from <0.5% to >3%)	ϕ_j	11.4; 13.3; 16.7; 20.0 kg/ha
Share of surface runoff from total runoff	γ	90%
Labour and capital inputs		
Farmer's wage rate per hour	w	EUR 11.35/h
Farmer's labour input per hectare	n	6.57 h/ha
Capital cost	rk	EUR 144/ha
Forestry		
Annual forest income		EUR 47.8/ha
Nitrogen runoff		N 2 kg/ha
Nitrogen runoff damage		EUR 7.15/ha
Biodiversity benefits		EUR 16/ha
Exogenous social rent		EUR 56.65/ha

Source: On the basis of field slope distribution in Finland, Finnish experimental studies on the runoff of nitrogen (Turtola and Jaakkola, 1987, Turtola and Puustinen, 1998) and ICECREAM model results, the parameter ϕ_j is set to four different levels to reflect the effect of field slope on the propensity for nitrogen runoff.

Table 4.12. Input use and land allocation: Flat rate *vs* alternative auctions

(average input use is reported in bold and range in parenthesis)

Policy	Nitrogen intensity, (kg/ha)		Buffer strip, (m)		Number of forest parcels	Number of rape parcels	Number of wheat parcels
	Rape	Wheat	Rape	Wheat			
Private optimum	**80.3** (80.2-80.5)	**122.8** (120.3-125.4)	**0.5**	**0.5**	68	12	80
Social optimum	**70.0** (67.3-72.5)	**114.2** (111.5-117.1)	**9.2** (7.3-11.8)	**9.9** (7.2-13.7)	76	40	44
Flat rate	**79.8** (79.1-80.5)	**122.8** (120.3-125.4)	**3.0**	**3.0**	40	40	80
EnvMax	**72.8** (66.7-80.5)	**122.2** (111.2-125.4)	**6.3** (0.5-15.0)	**1.9** (0.5-17.3)	67	37	56
Cost Save	**67.0** (64.0-70.3)	**121.1** (110.7-125.4)	**16.8** (11.3-24.3)	**4.8** (0.5-20.8)	68	57	35

Source: Author's calculations.

In the social optimum the total area under buffer strips is 4.0 ha and the average width is increased from 0.5 m in the private optimum to 9.5 m. Both nitrogen use intensity and the width of buffer strips vary over land productivity and environmental sensitiveness of land as is shown by figures in the parenthesis. In the private optimum variation in input use is driven solely by differences in land productivity whereas in the social optimum both land productivity and environmental sensitiveness of land (slope differences) affect the variation of optimal input use. However, environmental heterogeneity has more profound effects on the variation of socially optimal input use. For example, in the case of wheat cultivation and considering only one land productivity class (q = 35), the socially optimal buffer strip width varies from 7.8 m in the gentlest slope to 13.0 m in the steepest slope, and the optimal nitrogen intensity varies from 116.0 kg/ha in the gentlest slope to 112.0 kg/ha in the steepest slope.

With regard to the social optimum all policies allocate too much land to agriculture and among agriculture they fail to produce the optimal allocation of land between rape and wheat. In terms of input use the flat-rate payment results in too high nitrogen intensity for both crops and too narrow buffer strips. Both green auctions result in higher than socially optimal nitrogen intensity for wheat. Ranking by environmental score (EnvMax_joint) results in reasonably wide buffer strips in the rape cultivation whereas in those parcels, which are allocated to wheat cultivation the width of buffers fall clearly short of the socially optimal ones. The green auction with cost saving component (CostSave_joint) promotes wider buffers in the wheat cultivation as well and in sum this policy most closely resembles the socially optimal input use and land allocation.

Table 4.13 compares the social welfare performance of alternative policies relative to private and social optima. Reported are total profits, budget outlays, the number of accepted and rejected bids in two auctions, nitrogen runoff damage, biodiversity benefits, social welfare, and social welfare of solution relative to that of social optimum. Social welfare estimate also incorporates the social cost of public funds (marginal cost of taxation). In Finland the marginal cost of taxation has been estimated to be 10-20% of government payments (VATT). We assume a 15% cost of public funds and correct the social welfare estimate of policy alternatives with this estimate of marginal cost of taxation. The environmental payment is set at level EUR 21 per hectare in a flat-rate payment and the overall agri-environmental budget for green auctions is defined by the budget in the flat-rate payment (total budget is EUR 2 482).

Table 4.13. Profits, budget outlays and social welfare: Flat rate *vs* alternative auctions

Policy	Profit, (EUR)	Budget outlays, (EUR)	Number of accepted/ rejected bids	Runoff damage, (EUR)	Bio-diversity benefits, (EUR)	Social welfare, (EUR)	SW/SO
Private optimum	11 422	-	-	5 564	1 288	7 748	0.67
Social optimum	10 216	-	-	2 622	3 348	11 615	1.00
Flat rate	13 001	2 482	-	5 234	4 223	9 861	0.85
EnvMax	13 469	2 561	35/63	4 468	2 192	9 224	0.79
CostSave	12 581	2 489	67/25	3 140	3 171	10 726	0.92

Source: Author's calculations.

All policies increase total profits (these profits include government payments) relative to private and social optima. However, the green auction with cost-saving component (CostSave) is the least profitable from the farmers' viewpoint. When comparing the number of accepted and rejected bids under alternative auction designs one can see that the number of accepted bids is clearly higher under the auction with cost saving component because the unit-cost of each bid is lower (that is, the lower bid requested by applicants improves their chances of being accepted). All three policies are welfare-enhancing relative to the private optimum; however, their performance differs substantially in terms of aggregate welfare and environmental performance. The green auction with the cost-saving component (CostSave) clearly dominates the other policy design options in terms of reducing runoff damage, whereas the flat-rate payment is second to none in biodiversity promotion. Ranking exclusively by environmental score (EnvMax) increases total profits even more than the flat-rate payment while providing only moderate improvements in biodiversity promotion. In social welfare terms, the flat rate payment is preferable to an auction adopting a pure environmental ranking. However, a word of caution is in order because this result is achieved by the flat rate payment programme counterbalancing high runoff damages with an over-provision of biodiversity benefits relative to the social optimum. Moreover, it should be noted that the social welfare may be overestimated for the auction cases as it is implicitly assumed that the farmers' assessment of their costs to implement environmental measures is correct. As it is likely not to be the case, the decisions of the farmers may not be optimal given their true adoption costs.

Summary of the Finnish case study

This case study investigated how environmental regulations, environmental taxes and voluntary agri-environmental payments perform in a heterogeneous landscape. The focus was on crop production and differential land productivity that implies differential input use intensities as well as differential adoption costs with regard to agri-environmental measures. The effects of alternative policy instruments on nutrient runoff and biodiversity were taken into account through their impact on input use and land allocation choices.

The results obtained with stylised model for Finnish agriculture indicate that different agri-environmental policy instruments lead to very different outcomes in terms of land use, production, and environmental externalities, but the different instruments cannot be compared directly. Furthermore, the policy context in which these agri-environmental policies are put in place impacts the effectiveness of such policies. These differences can be appreciated especially at the extensive margin, expressed as entry-exit from agriculture and the relative allocation of land to different crops. Targeted agri-environmental incentives in the absence of area payments represent the scenario that most closely resembles the social optimum. However, the minimum buffer norm and the buffer payment are very close from the optimum and are likely to entail significantly less transaction costs than targeted instruments so they may be preferred. The minimum buffer norm and the nitrogen tax scenarios both lead to under-utilisation of agricultural land in favour of forestry, accentuating this aspect that was already present in the private optimum in the absence of policies. When agri-environmental policies are introduced, not in isolation but in the presence of area payments, the relative effectiveness of the policies is affected, and the bundle of environmental benefits provided changes.

The green/conservation auction (*i.e.* auction in which farmers bid for a limited amount of conservation contracts) with the cost saving component outperforms other agri-environmental payment programmes, but the difference is relatively small and the auction result may be overestimated due to farmers' potential estimation errors as regards their adoption costs. The outcome is quite close to the social optimum. In contrast, the green auction ranking by environmental score performed worse, and was even less welfare-enhancing than the flat-rate payment. This demonstrates the importance of the cost-saving component in environmental policy design. Ranking the bids exclusively by their environmental score creates incentives to over-provide environmental benefits for a fixed payment to increase their chances of acceptance.

Notes

1. It is assumed here that machinery is same for both crops but the number of tillage operations (*e.g.* seedbed tillage by harrowing) may differ between crops.

2. The study by Yrjölä and Kola provides the mean annual WTP estimate for multifunctional agriculture in Finland and it provided starting point for deriving the valuation of nutrient runoff damage and biodiversity benefits. Information on the Finnish population aged between 18 and 75 years of age (3 770 652) in 2002 and the total amount of cultivated land (2.2 million ha) was used to define the average willingness to pay per hectare of cultivated land. This willingness to pay estimate was then decomposed to biodiversity and nutrient runoff damage.

3. Contingent valuation (CV) method is based on consumer surveys whose questions elicit the consumer preferences for public goods by constructing a hypothetical market for public goods. The aim of a CV is to estimate consumers' willingness to pay for public goods by asking them how much they would pay for certain government actions (Carson, 2000).

4. This type of agri-environmental payment differs from the agri-environmental payments used, for example, in the EU, where farmers are compensated for income losses and cost increases.

5. However, these results are, at least in part, dependent on the fact that no biodiversity value is assigned to forestry since the marginal contribution of an additional hectare of forest is small, given the large portion of the country already under forest cover. No biodiversity value is assigned to buffer strips in forested areas. This latter assumption may be revised in light of information on the value of openings in forested areas, probably leading to a slightly less extensive agriculture as the one given in social optimum.

6. Biodiversity benefits for the private optimum (EUR 322) is the baseline value of biodiversity without any buffers.

7. The tax imposed on runoff is less than the marginal damage because buffers jointly produce biodiversity and runoff reduction.

8. The social optimum could be attained only using a first-best policy, which, in the case of an agri-environmental incentive programme, would have to allow for differentiated payment rates to take into consideration land heterogeneity. If runoff were affected only by fertilizer intensity, and biodiversity by the buffer area, paying for the benefits dollar-per-dollar would replicate the social optimum. However, since buffers jointly provide both benefits, the optimal payment rate will depend on the marginal cost of reducing fertilizer intensity *vis-à-vis* the marginal cost of an extra unit of buffer for limiting nitrogen runoff.

9. The top end of the bars in Figure 4.2 does not represent social welfare. To obtain social welfare the runoff "benefits" must be subtracted from the cumulative total of profits and biodiversity benefits.

Chapter 5
Switzerland: The environmental effects of dairy production

This chapter develops a dairy farm model which is designed to match the general approach of SAPIM. In the dairy model, the representative farmer maximises revenue from dairy production subject to manure policies. Given the orientation towards policy analysis, the complexities of milk production are described in a simplified way using a concave milk response function with two variable inputs. Dairy production is linked to land use decisions, because the farm always has an opportunity to produce either feed for livestock or crop for sale, or both. Manure emerges as a side-product in the dairy production and can be used as a fertilizer input in the crop production.

Governments restrict the use of manure and chemical fertilizer in agriculture by imposing either a nitrogen standard or a phosphorus standard. The grounds for using these standards arise usually from different environmental concerns. Nitrogen standards are typically used if nitrogen leaching from agriculture and livestock production to groundwater is excessive, making the nitrogen concentration of the groundwater so high that pumping water for drinking purposes imposes a health risk to people. In contrast to nitrogen, phosphorus standards are generally applied when runoff to surface waters deteriorates water quality leading to eutrophication and algae blooms.

Schnitkey and Miranda (1993) provide an analysis of phosphorus policies at a farm level. Their model exhibits both dynamic and spatial features. Dynamic analysis is needed, because phosphorus behaves dynamically in crop production. Particularly, the crop yield depends on the state of soil phosphorus, which changes slowly in time and depends on the past application rates of phosphorus. To analyse the phosphorus dynamics, Schnitkey and Miranda impose a carryover function of phosphorus, which links the current application rates to the accumulation of soil phosphorus. The spatial dimension is introduced to reflect the increasing transportation/hauling costs of the manure, because they require that the amount of manure produced must not exceed the land area available for manure application. Schnitkey and Miranda solve the steady-state optimum of the dairy farm subject to the requirement that the amount of phosphorus applied may not exceed the given standard, set either per parcel or on the whole farm. They demonstrate that there is a critical radius at which the profits from the use of the manure and the commercial chemical fertilizer are equal. While manure is applied inside the radius, the chemical fertilizer is applied outside the radius. They find that neither the point-wise control, which restricts soil phosphorus at each location (parcel), nor a whole-farm control, which restricts the average soil phosphorus (over all parcels), affects the critical radius.

Regional level optimization models extend the farm level approach to cover a given agricultural region. These models have some obvious advantages over the farm models. First, they allow one to focus on the supply of and demand for manure and the resulting equilibrium in the region. Moreover, these models allow for the determination of the optimal number of the farms in the region, which is a crucial determinant of the aggregate nitrogen and phosphorus runoff. These aspects cannot be focused properly in the pure farm models. A drawback of the regional models is that it is difficult to find decentralised regional policies to implement the regional optimum that is developed under the assumption that a sole owner makes all decisions in the region. Although the regional level is beyond the scope of the dairy farm model to be developed in this document, it is useful to summarise the basic findings of the regional models.

Innes (2000) provides a rich theoretical analysis of livestock waste regulation at the regional level. The model includes three environmental effects: spills from manure storage, nutrient runoff from the application of manure to crop lands and direct ambient pollution including gases and pests. He examines the effects of policies comprising scale regulations to animal inventories, fertilizer taxes and waste storage and handling standards. The main finding of the analysis is that, despite the imposed standards on manure management, the livestock farms have a tendency to produce too many animals either due to too large facilities, or due to too many facilities (*i.e.* too high entry). The farm-based instruments cannot achieve the optimal regional production scale. Thus, the key question for policy is how to enhance efficiency by regulating the livestock facility sizes and entry. Finding such efficient policies is difficult in practice.

While Innes (2000) derives theoretical results, Kaplan *et al.* (2004) and Smith *et al.* (2006) provide empirical analysis of regional manure policies. These papers demonstrate that the effects of manure policies depend crucially on the transportation costs of manure and substitution possibilities between manure and chemical fertilizers. Manure policies typically lead to reduced production, higher prices and decreased runoff. Also it turns out that the willingness of crop producers to accept manure to their crop lands affects the costs of manure management. Ribaudo *et al.* (2003) apply the model by Fleming *et al.* (1998) to investigate three alternative manure management practices (waste management, waste utilisation, manure transfer) to meet the manure nutrient standards at a regional level. They also determine the crop land area required to meet the standard on the basis of the willingness of the farmers to accept manure on their crop lands. Both papers find that imposing a phosphorus standard implies much higher costs than a nitrogen standard.

Drawing on previous literature, some "stylised facts" concerning the impacts of manure policies can be outlined (Ollikainen, 2006). First, the farms can generally adapt to the tightening nutrient standards by reducing the number of production animals, changing the diet, changing the pattern of cropping in the farmlands, or by finding new crop lands outside the farm; what combination of means is actually chosen, depends on their relative costs. Second, the adoption costs and profitability of dairy production depend crucially on manure management alternatives, especially on the hauling costs and the willingness of outside farmers to accept manure on their crop lands. Third, the type of the nutrient standard imposed clearly matters for the adoption costs. Adoption costs of achieving a nitrogen-based standard seem to be lower than those of meeting a phosphorus-based standard. Fourth, shifting from a farm-level to a regional level provides greater flexibility for manure management policies but finding decentralised solutions may be difficult (Ollikainen, 2006).

The remainder of this chapter is as structured as follows. First, a description of the theoretical framework is presented. This provides the basis for the empirical application on the basis of Swiss data. Policy simulations and results are then presented.

Theoretical framework

This theoretical framework is based on Ollikainen (2006). This model considers a dairy farm which has a given amount of agricultural land. By assumption, milk production is the main production line of the farm but as a side product the farm provides also meat. For a given number of dairy cows, there is an annual meat production associated with the outflow of older cows and inflow of calves. Revenue from selling the meat minus the cost of buying calves creates lump-sum net revenue reflecting the chosen number of milk cows. Moreover, manure produced as a side product of milk production must be managed, that is, stored and then used as an input in crop production.

The dairy farm allocates its arable land to the most profitable use. By assumption the quality of the arable land varies. It is assumed that within each parcel the land is homogenous but it differs across parcels, so that parcels can be arranged from the lowest quality to the highest quality parcels. Three possible land-use forms are considered. The land can be allocated to *cereals* production, *silage* production or to *pasturing*. Cereals produced can either be sold at the market or used for feeding the cows. Silage is typically produced for the use in the farm, because no general silage market is developed due to steeply increasing transportation costs, which make large scale silage trading impossible. However, local trade may exist. Pasturing is assumed to take place in the same field parcels where grass silage is cultivated. Pasturing is a viable option to feed cows during the summer season, although it is not mandatory for milk production. Pasturing can also be thought to promote animal welfare.

Basic framework: Dairy production

The farmer optimizes jointly milk production, land allocation, and manure management by choosing the relevant inputs and other required actions. To formalise this decision one needs to make assumptions concerning relevant features of production activities. Framework starts with the basic features of milk production and then necessary assumptions concerning land-use are made.

Production function of milk (per cow)

The response function of milk can generally be expressed as $Q = f(x_1,...x_n)$, where Q is the output per cow, and x_i:s describe all possible inputs in milk production: concentrate, silage, labour and others. In empirical experiments, from which response functions are derived, typically only a few inputs are controlled, while others are kept constant. To reflect these aspects, it is assumed that the main inputs in milk production are silage (s) and concentrate (v), *i.e.* $Q = g(s,v)$. In line with empirical research we assume that these inputs are imperfect substitutes in production.

Production of meat

The farm produces meat as a side product. For any number of dairy cows there is a constant outflow of older cows and inflow of young calves. Thus, the farm receives a steady flow of revenue from selling older cows to meat production and a steady flow of costs of buying calves to replace old cows. We denote this net revenue associate with any level of stocking, H, by Φ. Depending on relative prices this term may be positive or negative. Note also that a large increase in meat price leads to a change from dairy production to beef production. It is assumed here that changes are not big enough to change the production line of the farm.

Side-product: manure

As side-products milk production produces manure and urea, which contain nitrogen (N), phosphorus (P), and potassium (K) – all necessary macronutrients in crop production. However, surplus of nitrogen or phosphorus over crop requirements may lead to nutrient runoff and leaching into surface and ground waters. The share of manure and urea is treated as constant and both are expressed under the label manure (w). The amount of manure per cow depends on their feeding $w = w(s, v)$ .The actual share of N, P, and K in the manure depends on the ratio of silage and concentrate used. While the farmer may not be especially interested in the overall amount of nutrients and their shares in manure, society definitely is, because N and P may have negative environmental effects. Therefore, when specifying the runoff function from fields this aspect will be given special attention.

Manure causes costs. A manure storage system must be installed. Its size depends on the amount of dairy cows. Hence, this cost can be treated as an initial investment, because it is a crucial part of the barn design. Manure can be used to complement chemical fertilizers in production of silage or cereals. As the farm's crop lands are dispersed in the surroundings of the manure storage, the farm needs to invest in the capacity to transport manure to field and to spread it on the fields. It is assumed that spreading cost of manure, β, is constant in each parcel but let the transportation cost of manure be increasing. Let *e* denote the overall spreading and transportation cost. It depends on the maximum amount of manure that can be transported with one ride to the farm crop lands, *X*, and on the distance of the fields in the farm, r

$$e(m, r) = \beta m + h(X, m, r) \tag{1}$$

Due to capacity, the transportation costs become linked to per-hectare application of manure.

Sometimes it may be optimal for a dairy farm to expand its milk production to a scale where the farm crop lands cannot consume all the manure produced at the farm. Then, the farm must find crops lands outside the farm for the excess manure. Transporting manure outside the farm requires by assumption an investment in a bigger transportation capacity. To describe the transportation costs to outside crop lands, denote the overall manure produced in the farm by *wH*. Moreover, let *M* denote the total amount of manure applied in the farm crop lands, so that the excess manure produced is given by ($wH - M$). Denote the distance to the relevant outside fields by *t* and define the transportation costs to outside crop lands as $C = C(wH - M, t)$, Naturally, the shape of this cost function will affect the number of cows that is optimal in the long run for the dairy farm. The steeper the cost function, the more binding the farm's crop land area for the choice of the number of heads and resulting manure.

Prices of dairy inputs

Concentrate is bought at a constant competitive unit price p_v. Defining price for silage is slightly more complicated. There are no well-developed markets for silage because transportation costs are often prohibitively high. Thus buying silage outside the farm may be difficult. However, in the long run a local market may exist and silage can be bought at a constant unit price, p_s. In the presence of a local market, the farm values the silage produced at the market price. Finally, it is assumed that labour and nursery costs are constant per head.

Crop production

The farm has a given amount of crop land available, denoted by A. This land can be allocated to the production of cereals and silage. The land area allocated to silage production can also be used for pasturing. Given that the land quality varies, interior or corner solutions for the land use are possible: all land may be allocated entirely to either silage or cereals, but both land-use types may coexist.

Let y_i denote the output of silage and cereals (*e.g.* barley or oats), i = 1 and 2, where 1 denotes silage and 2 cereals. Both crops are produced by using a fertilizer input. The dairy farm applies both manure and chemical fertilizer. Chemical fertilizer applied per hectare is denoted by *l*, and the manure applied per hectare by *m*. Whether *l* and *m* are perfect or imperfect agronomic substitutes in production or not, depends on the nutrient status and soil texture of the land and on the yields the inputs produce. It is assumed here that both nutrient inputs contribute to the yield in a similar way. Moreover, by assumption the farmer targets specifically nitrogen application but when using chemical fertilizer the farmer also adds an annual steady-state amount of phosphorus. Assuming the farmer uses a compound NP fertilizer (in which nutrients are in fixed proportions), nitrogen applied is given by $N = \varepsilon l$, while $(1-\varepsilon)l$ denotes the phosphorus content. Recall, nitrogen content in manure was defined as $N = \alpha m$. Using above definitions the response function of the crops in terms of nitrogen applied in the form of chemical fertilizer and manure is given by $y^i = f^i(N)$, where i = s,v. This equation reflects the possibility that the farmer applies in each parcel either manure or chemical fertilizer depending on which produces higher profits. As the yield response function and land quality is the same for both inputs in any given parcel, the choice of using manure or chemical fertilizer will depend on the (constant) unit cost of the chemical fertilizer, manure spreading and hauling costs and on the parcel's location.

To keep the analysis simple, it is assumed that cereal production is more profitable in the higher quality parcels, while silage and pasturing takes place in the lower quality parcels. Moreover, given that the transportation costs matter, it is assumed that the higher quality parcels locate close to the barn and the land quality weakens monotonically in the distance from the barn. This allows for a clear and intuitive interpretation of the analysis, as the quality and distance will have a joint compounding effect on the optimal level of fertilizer use between the parcels. In an empirical model, all kinds of spatial configurations can be easily introduced.

Privately optimal dairy production

The long-run economic problem of the dairy farm is to choose the amount of cows and their feeding so as to maximise the profits from milk production, to allocate each quality of land to the most valuable use and to choose the optimal fertilizer use in each land quality. In the long-run, the amount of farmland, *A*, is a variable input, too. This choice, however, is omitted here and the optimal dairy production is solved for any arbitrary size of the arable land. This is not a drawback, because the main production line is by assumption dairy production. The costly transportation of the manure to crop lands outside the farm are allowed, which provides a flexible mechanism to maintain the nutrient balance in farm crop lands.

How will the model be solved for the optimal number of dairy cows? Generally, the number of cows, denoted by H, depends on many things, such as silage, nursery, labour and related costs. These unit costs are kept constant in the long-run solution. Increasing investment costs, $c(H)$, comprising the costs of the barn and milk processing system and the manure processing system $c(H) = c_1(H) + c_2(wH)$ close the long-run model.

It is assumed first that the government imposes neither manure regulations, nor taxes or subsidies on crop production. Under these assumptions, the farmer receives revenue from the produced milk and meat and pays for the costs of the inputs and investment in barn and manure systems. It is assumed that the revenue per head is constant $\pi^3 = \left[p_M g(s,v) - p_s s - p_v v - K + \Phi\right]H - c_1(H) - c_2(wH) - C(wH - M, t)$.

A dairy farm faces the following long run economic problem. It must choose the optimal number of cows and optimise their feeding and manure management. Moreover, the farm must allocate the farmland into cereals, silage and pasturing in full accordance with manure produced and silage needed in milk production. Optimality conditions for the private optimum in absence of government intervention are the following:

$$\Omega = c_1'(H) + (c_2' + C_H)w, \quad \text{where} \quad \Omega = \left[p_M g(s,v) - p_s s - p_v v - K + \Phi\right] \tag{2a}$$

$$\frac{g_v}{g_s} = \frac{p_v + (c_2' + C_w)w_v}{p_s + (c_2' + C_w)w_s} \tag{2b}$$

$$\pi^{*1} = \pi^{*2} \tag{2c}$$

From (2a) the optimal number of cows, H^0, is obtained when the marginal revenue (in terms of milk and meat) from the last cow brought in production equals to the marginal investment and manure transportation costs caused by this last cow.

From (2b) milk is produced by using an optimal input combination of pasture, silage, and concentrate that is defined by a condition where the value of their marginal contributions to milk production equals their unit prices adjusted by the marginal manure management costs.

From (2c) optimal land allocation is achieved where profit from producing cereals equals to profit from silage production when input use intensities are optimal for both crops.

Conditions (2a) – (2c) characterise the production decision of a dairy farm in the absence of government intervention.

Nutrient application standards

Nutrient application standards may be levied in a piece-wise manner as the maximum allowable amount of nitrogen or phosphorus per hectare, or as an average overall standard per farm. Here the effects of a nitrogen standard per hectare are examined. The impacts of a phosphorus standard would be analysed analogously, so that this analysis holds for phosphorus, too, as will be discussed at the end of this section.

Suppose now that a nitrogen standard is imposed on each hectare separately reflecting the land quality, crop nutrient needs and nutrient runoff damages. The nitrogen content of manure is defined by $N^m = \alpha m$ and nitrogen of chemical fertilizer in turn by $N^l = \varepsilon l$. Thus, for each parcel the farmer has the following nitrogen restrictions on the use of chemical fertilizer and manure,

$$\alpha m \leq \overline{N} \quad \text{and } \varepsilon l \leq \overline{N} \tag{3}$$

The nitrogen standard reduces the fertilizer application in the farm. As the total amount of land is given, the nitrogen standard (1) implies also an overall restriction on the aggregate application of manure (denoted by *M*). Thus, due to the imposition of the nitrogen standard, the farm faces two alternative situations. First, the overall manure produced by the farm (*wH*) may still remain within the limits of the aggregate maximum allowable manure ($wH < M$). Alternatively, it exceeds this amount ($wH > M$), which requires using some additional measures, such as reducing the number of heads, changing the diet or transportation to crop lands outside the farm.

The analysis starts with assuming that $wH < M$. Obviously, there is no need to adjust the number of heads in milk production, or the diet, nor transporting manure outside the farm. Thus, the dairy production is separable from the crop production in the short-run. Therefore, one can focus directly on the application rates of manure and chemical fertilizer to the farm's crop lands and trace out how they and the cropping decision changes. Intuition suggests that relative to the privately optimal solution the farmer now applies the excess manure to the fields that were previously subject to the chemical fertilizer application, that is, the farmer extends the critical radius to outer fields. Also, a change in the land allocation takes place. A formal analysis below shows that this indeed is the case.

In the presence of (3), the farmer has the following constrained maximisation problem to solve for each parcel and both crops,

$$Max \;\; \pi^i = p_i f^i(N) - cl_i, \text{ or } \pi^i = p_i f^i(N) - e(m_i, r); \quad i = 1,2 \tag{4a}$$

$$\text{subject to } \varepsilon l_i \leq \overline{N}_i, \text{ or } \alpha m_i \leq \overline{N}_i \;; \qquad i = 1,2 \tag{4b}$$

Writing down the Lagrangian function, Λ, and differentiating it with respect to nitrogen input produces

$$\Lambda_{l_i} = p_i f_N^i - c - \mu = 0\,; \;\; \Lambda_{m_i} = p_i f_N^i - (\beta + h_m) - \mu = 0 \tag{5}$$

Setting the shadow price, μ, equal to zero in equation (5) produces the privately optimal input use without government intervention. Optimality conditions under government intervention in (5) thus differ from the respective privately optimal conditions without government intervention. The shadow price, μ, indicates the costs caused by the tightness of the nitrogen standard. It is the same for both the manure and chemical fertilizer. Relative to private optimum without government intervention, the per hectare application rate is reduced. Therefore, the marginal in-farm transportation costs decrease and the optimal radius for manure transportation shifts outwards to cover (some of) the fields, which previously were subject to the chemical fertilizer application.

When the aggregate amount of manure produced in the farm becomes too high relative to the farmland area, the farm has some excess manure and three basic options to adjust to the nitrogen standard. First, the farm may reduce the number of cows to eliminate the excess manure. Second, changing the diet to reduce the nitrogen content of manure to eliminate the excess manure is possible. Third, the farm can also try to find crop lands outside the farm, where farmers are willing to accept manure on their lands.

Consider now the decision of the farm, for which $wH > M$ after the imposition of the nitrogen standard. In the short-run, the investments in the milk production capacity and the number of heads are given and transporting the excess manure outside the farm is the chosen reaction. Thus, the conditions governing the privately optimal milk production remain unchanged. Also, the first-order conditions for the application of manure and chemical fertilizer are identical to equation (5). The critical radius covers all farmlands. Thus, the farm reduces fertilization to match the required nitrogen standard and transports the excess manure outside the farm. Hence, the transportation of the excess manure causes the farm simply a lump sum cost of the size $C(\overline{w}\overline{H} - \overline{M}, t)$, where $\overline{w}\overline{H}$ indicates the fixed amount of manure and $\overline{M}$ indicates the aggregate nitrogen standard for the dairy farm.

Let us ask, finally, what happens to the land allocation in the presence of the nitrogen standard. It changes the relative profitability of cereal and silage production changes. The condition for the interior solution in land allocation requires that

$$\overline{\pi}^1 = \overline{\pi}^2, \qquad (6)$$

where bar refers to the profits from cereal and silage production under the nitrogen standard. The reduced nitrogen application reduces the profitability of both crops but more the profits from cereals grown on the high quality lands (for details, see Lankoski and Ollikainen, 2003). Thus, we find that the nitrogen standard increases the land area devoted to silage production, which is in line with the findings of the previous studies presented in Chapter 3 that indicated changes in the cropping pattern as a response to nitrogen standards.

In sum, the imposition of a binding and per quality adjusted nitrogen standard has the following implications. When $wH < M$, the milk production remains unchanged but crop production changes. The critical radius allocating parcels to manure and chemical fertilizer application is extended in favour of manure application and the amount of land allocated to silage production is increased. If $wH > M$, the means of adaptation chosen depends of their relative costs. If transporting is the cheapest way to manage the excess manure, the following impacts are obtained. The milk production remains unchanged, fertilizer use is reduced and the transportation of the excess manure causes an additional lump sum cost and a greater share of land is devoted to silage.

Consider, finally, the impact of a phosphorus standard on the fertilization rates. Suppose the farmer uses on some fields the chemical NP-fertilizer. Suppose further that the farmer can add pure nitrogen to complement the reduced amount of NP-fertilizer. Given that this additional nitrogen can be applied together with the NP-fertilizer, the impacts of the standard on fertilizer remain rather small. If additional nitrogen input is not feasible, costs are higher, yet smaller than in the case of manure.

The impact of the phosphorus standard on manure management is more complicated given that the share of phosphorus in manure is smaller than the share of nitrogen. Therefore, imposing a restriction to phosphorus leads to a greater reduction to manure application than the nitrogen restriction (provided, naturally, that nitrogen restriction is not extremely high). If no other nitrogen source is available, the phosphorus standard creates *de facto* a binding nitrogen standard for the farmer who targets nitrogen application. This was analysed above and the only difference to the above analysis is that now the restriction is tighter and the reduction in profits is greater. The farmer has, however, an opportunity of adding some chemical nitrogen to complement the restricted manure application. The technology of chemical fertilization is different from that of manure spreading. Adding chemical nitrogen causes extra costs and thus, adding chemical nitrogen may or may not be profitable in practice.

The analysis revealed that the impacts of manure policies depend much on the relative cost-burden on the alternative means of adapting to tightening regulation. When dairy farms are heterogeneous in terms of their cost structure, one can expect that all types of adaptation can take place: reducing the number of production animals, changes in the diet and changes in manure management together with altered cropping patterns.

Empirical application on the basis of Swiss data

Milk production is very important for Swiss agriculture, accounting for 23% of the value of agricultural production. According to data from the *Station fédérale de recherches en économie et technologie agricoles* (FAT), an average Swiss dairy farm (traded milk) has 15.7 dairy cows, 17.9 ha of utilised agricultural area and 1.6 farm work units. Between 1990 and 2004 the number of milk producers decreased from 50 334 to 33 072, and over the same period the average quota per dairy farm increased from 59 to 92 tonnes of milk. Federal expenditures for dairy farming have been decreasing in recent years, going from CHF 716 million in the year 2000, to CHF 504 million in 2004. Of these funds, 70% were used for cheese, 14% for butter and 15% for milk powder.

As regards agricultural policy the AP 2007 agricultural policy reform programme provided the basic legislative framework governing agricultural policy for the period 2004-07. From 2008, the new policy package is gradually being implemented under the Agricultural policy reform 2011 (AP 2011). The key feature of AP 2011 is a further reduction of 30% in budgetary expenditures for market price support (2008-11 in comparison with 2004-07). The savings are being used for direct payments for services (*e.g.* preserving culturally valuable landscape or animal welfare) and to compensate for difficult production conditions. All remaining export subsidies for agricultural commodities are to be eliminated, and customs duties on imported animal feed and cereals for human consumption are to be reduced.

There are two main categories of direct payments. *General Direct Payments* are mainly granted in the form of general area and headage payments, and to a lesser extent also include payments to farmers operating in less-favoured conditions. *Ecological Direct Payments* are mainly granted in the form of area and headage payments to farmers who voluntarily apply stricter farming practices than those required by the regulations and the farm environmental management practice requirements (*PER – prestations écologiques requises*). All Direct Payments are based on the condition that farmers comply with the PER.

The structure of the programmes and the eligibility conditions applied within the *General Direct Payments* and the *Ecological Direct Payments* categories have remained largely unchanged under the new AP 2011 (implemented from 2008). Outlays to farmers for these two categories remained rather stable in 2007 and 2008. About 80% of the total is granted under ***General Direct Payments***, although they have declined by 5% in 2008. Area payments per hectare of arable land and permanent cropland were reduced, but remain the most important single category and account for 60% of general direct payments. The other important category of general payments is the payment per livestock unit (LU) for roughage-consuming animals, and these payments were increased by 37% in 2007 to compensate for a reduction in milk market support. Additional payments are granted for livestock under difficult conditions (*e.g.* mountains). Headage payments for roughage-consuming animals and animals raised in difficult conditions together accounted for 33% of general direct payments. The remaining 5% of General Direct Payments are paid to cultivate the steep slopes in mountain regions.

Ecological Direct Payments increased overall by 3% and about 44% of these payments are provided to improve animal welfare. Payments for animal friendly husbandry systems and headage payments for animals kept outdoors increased by 8.5% and 2.4% respectively. Around one quarter of ecological payments are granted for "ecological compensation" (payments for extensive meadows, dryland areas to produce litter, hedges, floral and rotation fallow, extensive area strips and high-stem fruit trees) and another 10% is paid for “contributions to environmental quality” *(Contributions au sens de l’ordonnance sur la qualité écologique – OQE)*. In 2008, the level of ecological compensation decreased by 3%, while the contributions for environmental quality increased by 40% (although from a lower base). The remaining ecological payments for extensive farming and organic farming were reduced by 6.5% and 12.5% respectively.

From May 2009, the ***milk quota*** system was abolished for all dairy farmers, although until May 2015 they will only be able to sell milk under the terms of existing contracts drawn up with buyers (exempted are those farmers who sell their milk directly to final consumers). Price support expenditures for dairy products were reduced in 2007 by 17% compared to 2006, to reach CHF 361 million. The expenditures budgeted for 2008 was reduced by another 5%. Payments for the price supplement paid to processors for milk transformed into cheese and the premium for milk produced without silage feed were reduced during 2007 and 2008, while domestic market support for butter slightly increased. In 2007, export subsidies for cheeses and other milk products were 60% lower than in 2006. In 2008, they were further reduced (50% lower than in 2007) especially for the other milk products. By 1 July 2009, all price support expenditures for dairy products will be abolished except the price supplement paid to processors for milk transformed into cheese and the premium for milk produced without silage feed.

The main environmental challenges facing agriculture were identified in 2002 by the Federal government which established a number of agri-environmental targets for 2005 (from a 1990-92 base), including: reducing surplus nitrogen (23%) and phosphorus (50%); lowering pesticide use (30%) and ammonia emissions (9%); achieving 10% of farmland as *ecological compensation areas* and cultivating 98% of farmland according to ecological compliance or organic farming standards; and requiring 90% of drinking water in agricultural areas to have a nitrate level below 40mg/l.

Dairy production in Switzerland

The heterogeneity of dairy production farms in Switzerland poses considerable challenges for defining a representative production system. The International Farm Comparison Network (IFCN), which compares costs and production systems across countries, addresses such issues by providing information on typical farms. For Switzerland the data are assembled at FAT by relying on participating farms and on expert opinion of what constitutes a representative production system.

Four dairy farm types have been identified in the context of IFCN for Switzerland, representing all regions where milk production is significant: two types in the Plain region, one in the Hill region, and one in the Mountain region (Gazzarin, 2002). These representative farms differ in herd size (from 26 in the mountains to 70 in the larger of the two plain farms), in their cost structure and land requirements (highest in the mountains), and final use for the milk (with milk in the hill region being used to produce traditional cheese which requires not using any silage). These differences require focusing any modelling exercise to a specific region, since a representative farm across regions would not make any sense. The modelling effort undertaken by the Secretariat focuses therefore on the Plain region.

The Plain region was chosen because it appears to be the region where dairy production is most competitive, and the production systems present are the most similar to those typically found in other OECD member states. In a recent study, FAT has developed a detailed representation for a set of potential production systems in the Plain region (Gazzarin and Schick, 2004). In the framework adopted by FAT various elements and processes define a production system. These are presented in Annex B (Table B.1), and include, for example, the production of fodder, feeding systems, and herd type and size.

A considerable number of permutations are possible by changing characteristics of the factors included in this table. Here we focus on herd type and size, and feeding strategy (feed-mix) as factors of variation while other parameters are assumed as given (Table 5.1). For example, all systems considered here adopt free-stall housing, with a horizontal silo, perforated ground, and evacuation of the manure by scraper.

Table 5.1. Characteristics of systems with different herd types

	Milk yield	Labour costs	Capital costs	Land costs	Stocking rate
	(kg ECM/ cow/yr)	(CHF/cow/yr)	(CHF/cow/yr)	(CHF/cow/yr)	(head/ha)
Reference cow (grazing and silage)	6 700	2 050	623	348	1.87
Medium-productivity cow	7 700	2 087	678	354	1.81
High-productivity cow	10 000	2 460	830	390	1.82
Reference cow (all grazing)	6 700	1 869	623	362	1.80
Seasonal grazing	6 500	1 736	611	384	1.76

ECM stands for Energy Corrected Milk, which takes into account variations in fat content.
Source: Obtained using data from FAT report # 608, Table 8. Values are converted from costs per kg of milk to costs per cow.

Among the physical inputs required, the combination of fresh forage, grass and maize silage, and concentrate feed clearly play an important role in determining both milk yield obtained per cow and nutrient content of manure.

Milk production per cow

In the analytical model the main inputs in milk production $Q = g(s, v)$, silage (s) and concentrate (v), were assumed to be imperfect substitutes in production. The production of milk in the empirical model is represented as conversion of dry matter, protein and energy from 3 sources: pasture and grass silage, maize silage and feed crop (barley). These three sources of feed are imperfect substitutes, meaning that one cannot substitute one for the other in fixed proportions to obtain a given yield. Since the modelling framework is static, the amount of herbage intake associated with a hectare of pasture/grass includes all grass-based outputs produced on that hectare, whether grazed grass or grass silage.

A milk production function was calibrated on the basis of data linking milk yield to feed mix. As can be seen in Table 5.2, the annual milk yield per cow is 6 500 kg using a diet of mainly grass and maize silage together with a small amount of concentrate. However, to achieve a yield of 10 000 kg, the share of concentrate increases while that of grass decreases. Energy content in MJ was estimated for all feed mixes and then a quadratic specification was fitted to describe how milk yield responds to the share of concentrate x_s on feed mix, $y = \alpha + \beta x_s - \gamma x_s^{\ 2}$. The estimated milk production function describes quite well this non-linear relationship – higher yields are obtained by increasing the share of concentrate (in terms of total energy).

Table 5.2. The effects of alternative feeding mixes on the amount of slurry (m^3) and excreted nitrogen and phosphorus (P_2O_5), kg per cow per year

Milk yield	Con-centrate	Maize silage	Grass and hay, total	Slurry, un-diluted	Slurry, diluted	Excretion N	Excretion P_2O_5
(kg ECM)	(kg)	(kg)	(kg)	(m^3)	(m^3)	(kg/cow/yr)	(kg/cow/yr)
6 500	157	1 652	4 832	22.6	45.1	112.8	40.0
8 000	357	1 689	4 958	24.2	48.4	121.0	42.9
10 000	971	2 409	4 304	26.4	52.8	132.0	46.8

Source: Menzi (2006) private communication on the basis of data from the *Haute école suisse d'agronomie* (HESA).

Manure management

The data in Table 5.2 also indicates the amount of manure excreted according to the diet, as well as the consequent nitrogen and phosphorus content of the manure. This information, together with the optimal level of fertilizer and manure application on fields permits the calculation of nutrient balances for the mixed dairy-crop farm.

The other important aspects of manure management are the technology used of housing the animals, storing the manure, and finally spreading the manure. These are captured respectively by the fixed and variable cost assumptions in the model. In the Plain region, approximately 65% of housing systems are tied and 35% are loose housing systems (cubicle houses). It is assumed here that a representative manure management system is based on a slurry system, which represents over 50% of manure management systems in dairy farms in the Plain region. This system stores undiluted, untreated manure in tanks or pits until manure is applied to cropland. In the Plain region over 80% of slurry storages are covered. Depending on the size of manure storage (from 235 to 452 m^3) the investment cost of covered storage is from CHF 55 000 to 86 000 (Ammann, 2005). Manure spreading is mainly based on broadcasting technique which represents 87% of dairy farms. However, trail hose application is analysed as an alternative to broadcasting technique. Depending on the capacity of manure spreading machinery (from 2.5 to 8.5 tonnes) the investment cost is from CHF 15 000 to 40 000 (Ammann, 2005).

Crop production

Due to lack of experimental data and parameter estimates for multi-nutrient crop response functions, crop yield response is modeled through nitrogen. It is assumed here that the farmer applies compound NPK chemical fertilizer, in which the main nutrients (nitrogen, phosphorus, and potassium) are in fixed proportions (*e.g.* 20-5-10). By assumption the farmer chooses the level of nitrogen application (N_i) for each crop, and because nutrients are in fixed proportions, nitrogen fertilizer intensity determines also the amount of phosphorus and potassium used. Therefore, the amount of phosphorus applied is *e.g.* $0.25*N_i$ and the model can be expressed in terms of N_i. It should be noted that assumption relating to fixed proportions of main nutrients is only valid for chemical fertilizer application. In the case of manure the ratio between nitrogen and phosphorus could be affected by the feeding mix but data used for this application (Table 5.2) show that the ratio between nitrogen and phosphorus is not affected by different feeding rations. Per-hecatre nitrogen application intensity, $N_i = N_i^f + \varphi N_i^m$, is the sum of two nitrogen sources; chemical fertilizer and manure. Depending on relative prices the application of nitrogen in a given parcel may come fully from chemical fertilizer or from manure or from both. Effective nitrogen value in manure or nitrogen available for crop, φ ($0 < \varphi \leq 1$) depends on the manure spreading technique (*e.g.* injection *vs* broadcast technique). A Mitscherlich nitrogen response is employed for describing crop response to nitrogen, $y_i = \alpha(1 - \gamma e^{-\beta N_i})$, where y_i is crop yield (kg/ha), N_i is applied nitrogen (kg/ha) and α, γ, and β are parameters. Differences in soil productivity (land quality) can be incorporated through the parameter α and γ in the Mitscherlich Nitrogen response function. Crop Nitrogen response functions are calibrated on the basis of data concerning average nitrogen use for modeled crops (pasture/ grass silage, maize silage, and barley) and crop yields in the Plain region.

Environmental effects of dairy production

Ammonia emissions, GHGs and nitrogen and phosphorus surpluses are the environmental issues constituting the main focus of the empirical application. Ammonia emissions will reflect the impact of dairy production on air quality, GHGs show the impact on climate and nutrient surplus calculations serve as proxies for potential load of nutrients to watercourses and in the case of nitrogen also emissions to air.

Ammonia emissions are based on Swiss data and expert opinions on the effect of different feed-mixes or feeding strategies on nitrogen content of manure and the effect of alternative manure storage and spreading options on ammonia emissions during manure storage and spreading (Kulling *et al*., 2002; Hindrichsen *et al*., 2006; Reidy *et al*., 2008). Annex B (Table B.2) describes NH_3 emission factors for different combinations of housing system, manure storage and manure spreading (Menzi, 2006).

The effects of alternative feeding mixes on the amount of undiluted and diluted slurry (m^3) and excreted nitrogen and phosphorus (P_2O_5) are given in Table 5.2 (Menzi, 2006). Methane (CH_4) and nitrous oxide (N_2O) emissions are modeled on the basis of the manure volume under different feeding mixes and estimates provided by Kulling *et al*. (2002) and Hindrichsen *et al*. (2006). As regards methane emissions both emissions from manure storage and enteric emissions are taken into account as well as nitrous oxide emissions from manure storage.

The soil surface nitrogen and phosphorus balances for each crop are calculated as the difference between the total quantity of nitrogen or phosphorus inputs entering the soil and the quantity of nitrogen or phosphorus outputs leaving the soil. The soil surface balances (surplus/deficit) for nitrogen and phosphorus in each parcel in the model were calculated by adding the total nutrient content of chemical fertilizer application and manure application and by subtracting the nutrient content of the crop output. The calculated nutrient surplus (kg/ha) provides an indicator of the production intensity, and of the potential nutrient losses and environmental damage to surface and ground waters. In addition to soil surface balance we also report farm-gate balance as a general indicator of farm's environmental load. Farm-gate balance is calculated by adding nutrient content of inputs entering to farm (feed, chemical fertilizer, animals, etc.) and by subtracting nutrient content of outputs leaving farm (crops, milk, animals, etc.).

Coefficients for nutrient content of manure are based on Menzi (2006) while coefficients for nutrient content of different crops are taken from DBF (2001), which provides detailed information on both the recommended level of nutrient use for each crop and nutrient content of crop output.

Policy simulations and results

The following policy simulations are as conducted:

- General agricultural policy and its reform:
  - Abolishment of dairy quota and its impact on milk price. This scenario applies a 16% reduction in the milk price as reported in the Ferjani (2008).
- Regulatory instruments:
  - Manure/nutrient application standard (based on N) required for achieving a zero N surface balance.
  - Imposing trail hose application technique for manure application on fields.
  - Maximum stocking rate of 1.2 livestock units per hectare. Given the farm has 20 ha in total, this translates into a maximum upper limit of 24 cows.

- Economic instruments:
  - Tax (25%) on the price of commercial chemical fertilizer.
  - Tax on the price of nitrogen actually applied (25%). Both chemical fertilizer and manure application spreading are taxed.

The policy scenarios described above assume private profit maximization by dairy farmer. The outcome of the different policy instruments is expressed both with respect to environmental effectiveness and cost-effectiveness for a given instrument. The environmental effectiveness of policy scenario is measured as the reduction of the given negative environmental impact relative to the Baseline. Cost-effectiveness combines farm profit reduction (farmer's adoption cost or profits forgone) with the environmental outcome – that is, the change in profits relative to change in environmental outcome.

Agri-environmental policies in Switzerland are implemented in a setting where other agricultural policies are present and for this reason the scenarios assume that farmers receive area payments of CHF 1 200 per hectare of cultivated land and headage payments of CHF 856 per livestock unit.

Table 5.3 shows the economic and production impacts of alternative policy scenarios.

Table 5.3. SAPIM simulation results: The effects of different policy scenarios on profits, production and land-use decisions

	Policy scenarios						
	Baseline	**Milk quota abolishment**	**Maximum stocking rate**	**Trail-hose application**	**Standard on N application**	**Tax on N fertilizer**	**Tax on N application**
Farm profits (CHF/yr)	82 083	47 993	62 231	81 854	74 984	79 607	76 445
Numbers of cows	34	32	24	34	34	34	34
Stocking rate (cows/ha)	1.7	1.6	1.2	1.7	1.7	1.7	1.7
Milk (litres/yr)	331 735	317 505	236 123	331 735	331 735	331 735	331 735
Crop production (kg)							
Grass silage	181 342	181 342	181 342	77 933	148 750	258 219	250 542
Maize silage	94 713	94 713	94 713	219 824	79 861	-	-
Barley	-	-	-	-	-	-	-
Land allocation (ha)							
Pasture/ grass silage	14	14	14	6	14	20	20
Maize silage	6	6	6	14	6	0	0
Barley	0	0	0	0	0	0	0

Source: Author's calculations.

The first column in Table 5.3 portrays the outcome for the Baseline without agri-environmental policy interventions. Farm profits include revenues from both dairy and crop production activities (20 ha). The baseline is a highly-intensive dairy production system with 34 cows which translates into a stocking rate of 1.7 dairy cows per hectare. Land is allocated in large part to pasture/grass silage (70%) while the rest of available land is allocated to production of maize silage for feed (30%). Both grass silage and maize silage are produced in greater quantities than consumed as feed and thus surplus is exported from farm (36 tonnes of grass silage and 14 tonnes of maize silage). Barley is not produced, as with given relative feed prices the most profitable crops are grass and maize silage. For an optimised diet feed grain (barley) is imported and total imports are 32 tonnes per year.

Imposing environmental constraints – through either constraints on production practices or relative prices – reduces dairy farm profits in alternative policy scenarios relative to the Baseline. General agricultural policy reform, and more specifically decrease of milk producer price by 15% due to abolishment of milk quota, reduces farm profits by almost 42%. The optimal herd size is reduced from 34 to 32 dairy cows with corresponding decrease both in the stocking rate and total milk production. A constraint on the maximum stocking rate (1.2 dairy cows per hectare) effectively reduces the herd size from 34 to 24 and as a result milk production is decreased by 29% and profits by 24%. Hence, while removal of price support reduces farm profits significantly and has an impact on the stocking rate, the regulation on the stocking rate has a huge impact on the stocking rate, but a much smaller impact on farm profits. Although milk production is significantly affected by these two policy scenarios land allocation between alternative crops and optimal level of crop production are not affected because neither of these two scenarios affect relative prices and thus profits in crop production.

The rest of the policy scenarios – nitrogen tax on chemical fertilizer, tax on nitrogen application (both chemical and manure), trail hose application technology standard and standard/upper limit on nitrogen application per hectare – are neutral in terms of dairy production (dairy cow number and milk production) but they reduce overall farm profits relative to the Baseline. However, in the case of trail hose application the overall farm profits are reduced only very slightly (0.3%). In summary, this suite of scenarios illustrates how different agri-environmental policy instruments may have differential impacts on farm profits. Optimal land allocation and input use in crop production, including chemical fertilizer and manure application rates, are all affected by these policy instruments. The scenarios taxing nitrogen fertilizer and nitrogen application results in all land allocated to pasture/grass silage. The production of feed grain is not profitable in any of the reported policy scenarios.[1]

Manure produced, its nutrient content, and the application rates of both manure and chemical fertilizer are reported in Table 5.4.

Total amount of manure produced on the farm is naturally linked to the number of cows and consequently in the case of milk quota abolishment and the constraint on the stocking rate there is a reduction of the total manure produced on the farm (4% and 29%, respectively). These policy scenarios do not affect the optimal level of manure application, but manure exports outside the farm are reduced relative to the Baseline. The remaining scenarios do not affect the total manure produced on the farm but they have an impact on the amount of manure applied on farmland and therefore on manure exports. Both standard on trail-hose application technique and tax on chemical N fertilizer give strong incentives to shift application of manure to field parcels further away from farm centre. The critical radius for manure transportation and spreading increases. However, the intensity of nitrogen applied in manure is not affected because in the case of the fertilizer tax the relative price of chemical fertilizer has changed but not price of nitrogen from manure. In the trail-hose application scenario the cost of applying nitrogen from manure is reduced and thus it provides an incentive to increase the amount applied. Both nitrogen standard and a tax on nitrogen application effectively reduce the amount of manure applied and therefore result in increased exports of manure outside of the farm.

Table 5.4. SAPIM simulation results: The effects of different policy scenarios on manure production, manure application, manure exports, nutrient content of manure, and nitrogen and phosphorus fertilizer application

	Policy scenarios						
	Base-line	**Milk quota abolish-ment**	**Maxi-mum stocking rate**	**Trail-hose applica-tion**	**Standard on N applica-tion**	**Tax on N fertilizer**	**Tax on N applica-tion**
Manure produced (m^3)	884	847	629	884	884	884	884
Manure application (m^3)	483	483	483	677	218	687	397
Manure exported (m^3)	401	364	146	207	666	197	487
Nitrogen in manure (kg/ha)	221.0	211.8	157.3	221.0	221.0	221.0	221.0
Phosphorus in manure (kg/ha)	34.2	32.7	24.3	34.2	34.2	34.2	34.2
N fertilizer application (kg/ha)	114.6	114.6	114.6	82.5	48.3	48.4	85.8
P fertilizer application (kg/ha)	17.9	17.9	17.9	13.9	7.6	6.0	10.6

Source: Author's calculations.

Chemical nitrogen and phosphorus application rates are not affected by the first two policy scenarios, namely milk quota abolishment and maximum stocking rate, while the rest of the policy scenarios have quite significant impact on the optimal level of chemical nitrogen and phosphorus application rates. In particular, the scenarios with tax on nitrogen fertilizer and standard on nitrogen application reduce the amount applied by 58%.

The environmental impacts of alternative policy scenarios, in terms of nitrogen and phosphorus balances, ammonia emissions, and CO_2-eq emissions are reported in Table 5.5.

Table 5.5. SAPIM simulation results: Nutrient balances, ammonia emissions, GHG emissions

	Policy scenarios						
	Base-line	Milk quota abolish-ment	Maximum stocking rate	Trail-hose applica-tion	Standard on N applica-tion	Tax on N fertilizer	Tax on N applica-tion
	Surface nutrient balance (kg/ha)						
Nitrogen	81.2	81.2	81.2	73.8	0.0	85.3	59.9
Phosphorus	-2.2	-2.2	-2.2	1.8	-15.8	-6.4	-11.8
	Farm-gate nutrient balance (kg/ha)						
Nitrogen	-4.5	-11.9	-55.0	3.3	-19.3	-99.0	-52.0
Phosphorus	1.4	0.4	-5.3	-2.2	-1.9	-10.5	-4.7
	Other environmental impacts						
Ammonia emissions (kg/head)	38.0	38.0	38.0	26.2	38.0	38.0	38.0
CO_2-eq emissions (kg/head)	3 887	3 881	3 887	3 887	3 887	3 887	3 887

Source: Author's calculations.

In Table 5.5 two different nutrient balance methods are reported, namely surface balance (kg/ha) and farm- gate balance (also reported as kg/ha). Using the surface balance method the Baseline implies a small phosphorus deficit and a substantial nitrogen surplus. Because nitrogen can be applied either as chemical fertilizer or as manure, the nitrogen surplus is poorly addressed in most of the policy scenarios except when a quantitative standard on nitrogen application is imposed, or when a tax is applied on both fertilizer and manure application. The scenario for N standard application requires an average reduction of 132 kg/ha of nitrogen applied in order to achieve a zero Nitrogen surface balance. The tax on nitrogen application reduces nitrogen surface balance by 26%. Soil surface phosphorus balance is generally negative in most scenarios because of the predominance of grass silage in the composition of crops planted. It is only positive in the trail-hose application scenario, where there is a large shift in cropping in favour of maize silage. This shift in cropping occurs because of lower nitrogen prices (from manure) implied in this scenario make maize silage the most profitable crop – these results are clearly conditioned on the initial assumptions about yield response to nitrogen application for the various crops.

The farm-gate nutrient balance measures the difference between the nutrient content of inputs (chemical fertilizer and net trade in feed) and output (milk produced). Measured in this way, the farm-gate balance in the Baseline is negative for nitrogen, and slightly positive for phosphorus. Nitrogen surplus is negative in all policy scenarios except when technology standard on trail-hose application method is adopted. Most of the policy scenarios overshoot environmental objectives leading to further negative balances (deficits) for both phosphorus and nitrogen relative to the Baseline. This is particularly case in the scenario with nitrogen fertilizer tax, but also for the scenarios imposing a standard on nitrogen application and maximum stocking rate. Setting these standards too stringently also has implications in terms of profits, and in fact, these two options are among the ones with the lowest profits.

Concerning ammonia emissions, these are closely linked to the amount of nitrogen in manure, the housing system, type of manure storage and manure spreading technique. Therefore, the only policy scenario where ammonia emissions per head of dairy cow are significantly affected is adoption of trail-hose manure application method. Ammonia emissions are reduced over 30% relative to the broadcast application technique used in the Baseline. Similarly, the CO_2-equivalent emissions show little variation between alternative policy scenarios when measured per head of dairy cow.

Table 5.6 shows that the average cost-effectiveness of alternative policies to reduce nutrient surpluses varies quite a lot within and across nutrients and balance methods. Generally it is more expensive to reduce phosphorus surplus than nitrogen surplus. Cost-effectiveness ratio is not applicable for those cases that result in increase of negative environmental effect relative to Baseline (*e.g.* the case of trail-hose application and surface balance for P).

Table 5.6. SAPIM simulation results: The cost-effectiveness of different policy scenarios on the reduction of nitrogen and phosphorus surpluses

	Policy scenarios			
	Trail-hose application	Standard on N application	Tax on N fertilizer	Tax on N application
Profit forgone relative to the Baseline, CHF	229	7 099	2 475	5 638
Surface balance N, CHF/kg	2	4	n.a.	13
Surface balance P, CHF/kg	n.a.	26	30	29
Farm-gate balance N, CHF/kg	n.a.	24	1	6
Farm-gate balance P, CHF/kg	3	107	10	46

n.a. = not applicable

Source: Author's calculations.

Summary of the Swiss case study

In this chapter a stylised dairy farm model has been developed to describe dairy production and its environmental effects. Theoretical results were presented for nutrient standards with and without excess manure production. In the former, milk production remains unchanged but crop production changes, and the critical radius is extended in favour of manure application and the amount of land allocated to silage production is increased. When excess manure is produced, the adaptation chosen depends on the relative costs of different options. If transporting is the cheapest way to manage the excess manure, milk production remains unchanged, fertilization is reduced and the transportation of the excess manure causes an additional lump sum cost and a greater share of land is devoted to silage.

As regards environmental effects, the empirical application focuses on ammonia emissions, GHGs, and nitrogen and phosphorus surpluses. Analysed policy instruments ranged from general agricultural policy reforms to more targeted environmental policy instruments, including both regulations and economic instruments. What emerges from the empirical results is that general agricultural policy reform in the form of milk quota abolishment has significant impact on the profitability of production – through a milk price decrease of 16% – and less so on herd size and the production volumes. The opposite holds for regulation imposing a maximum stocking rate.

Most of the policy scenarios do not affect total manure produced on the farm, but they have an impact on the amount of manure applied on farmland and therefore on manure exports. Both standard on trail-hose application technique and tax on chemical N fertilizer give strong incentives to shift application of manure to field parcels further away from farm centre. Thus, the critical radius for manure transportation and spreading increases. Both nitrogen standard and a tax on nitrogen application effectively reduce the amount of manure applied and therefore result in increased exports of manure outside of the farm. Chemical nitrogen and phosphorus application rates are affected significantly by some of the policy instruments, notably tax on nitrogen fertilizer and standard on nitrogen application.

Because nitrogen can be applied either as chemical fertilizer or as manure, the nitrogen surplus is poorly addressed in most of the policy scenarios except when a quantitative standard on nitrogen application is imposed, or when a tax is applied on both fertilizer and manure application. This result clearly demonstrates the well known problem of substitution of unregulated activities, whether chemical fertilizer or manure.

Note

1. We have also simulated a policy scenario in which grain price was increased by 25% and in that scenario 4 ha out of 20 were allocated to feed grain.

Chapter 6
United States: The environmental effects of crop production and conservation auctions

This case study focuses on the economic and environmental performance of conservation auctions *vs* traditional agri-environmental policy measures in the US. The economic and environmental effects are, however, not aggregated in this case study and no social benefit function is computed. The three alternative land-use types analysed in this application are: land retirement for environmental purposes, such as partial field buffer strips, and two alternative tillage methods to produce a cultivated crop; no-till and conventional mouldboard tillage. The Conservation Reserve Program Continuous Signup (CCRP), for example, provides partial field retirement through vegetative buffer installation along water courses to capture nutrient runoff and provide other environmental amenities. No-till and conventional tillage represent here management under the working lands programmes (such as the Environmental Quality Incentives Program, EQIP). In this application the sources of heterogeneity are both differential land productivity and environmental sensitivity of the parcels. Environmental heterogeneity is represented here by differing slopes of field parcels towards watercourse. Different field slopes result in variation in the propensity of soil to erode and nutrients and herbicides to runoff from different field parcels.

Following Aillery (2006), land retirement is usually best suited for those parcels where environmental damage due to erosion and related sediment, nutrient, and herbicide runoff would be high relative to the value of agricultural commodity production. While land retirement usually results in large environmental benefits per hectare, the programme cost of land retirement could be high since payment rate should reflect the full agricultural value of the land. This means that under a budget constraint for the agri-environmental or conservation programme larger overall environmental benefits may be obtained from a working lands programme, since it allows land to remain under production and compensation payment rates do not need to reflect the full agricultural value of land.

For working lands programmes the performance of no-till *vs* conventional mouldboard tillage is an important element of a conservation plan. Relative to conventional tillage, no-till farming is generally found to provide considerable environmental benefits in terms of reduced soil erosion, nitrogen runoff, and particulate phosphorus runoff. However, not all environmental effects of no-till are favourable relative to conventional tillage, since many studies report that dissolved (orthophosphate) phosphorus runoff may increase under no-till due to the accumulation of phosphorus in the soil surface. Moreover, no-till may increase the abundance of perennial weeds thus requiring a higher use of herbicides which may eventually increase herbicide runoff relative to conventional tillage, and no-till may also increase potential for leaching of nutrients and pesticides to groundwater. Furthermore, with regard to greenhouse gases no-till farming contributes to carbon sequestration but it may increase the emissions of nitrous oxides. Thus, from society's viewpoint no-till involves important environmental trade-offs that need to be incorporated into the analysis.

From the farmer's viewpoint no-till seems to provide unambiguous cost reductions because of lower labour requirements and fuel consumption. Capital investment and maintenance costs may also be reduced, although upfront capital requirements for new equipment can represent a barrier to adoption. Furthermore, relative to conventional tillage no-till may provide potential revenue from carbon credits in the context of carbon markets. However, no-till yields may be lower than those under conventional tillage, in particularly during the transition period (usually up to five years) to no-till before soil structure develops so that it starts to support no-till yields (*e.g.* number of macropores). Moreover, no-till requires specialised equipment, including direct planter, and it also affects the timing of planting, since "covered" soils usually take longer to dry and warm after winter period. Thus, a switch from conventional tillage to no-till farming implies a learning curve for a farmer. Thus from a farmer's viewpoint no-till involves some important economic trade-offs.

With regard to policy instruments to be analysed in this case study the main focus will be on:

- Environmental and economic performance of land retirement programme *vs* working land programmes;
- Environmental and economic performance of green auctions *vs* flat-rate agri-environmental payments; and
- The cost-effectiveness of traditional policy instruments, such as input use taxes, input application standards, and payments for conservation tillage practices and structural practices, including buffer strips between field parcels and watercourses.

For assessing the trade-offs between land retirement and working lands programmes, theoretical and empirical frameworks are developed in order to explicitly analyse relative costs and benefits. To our knowledge Feng *et al.* (2006) is the only study where land retirement and working lands programmes are analysed within a joint framework. Feng *et al.* analyse how the existence of a pre-fixed budget allocation between CRP and EQIP affects the potential environmental benefits obtained from alternative policy implementation schemes. In their empirical application based on data from Iowa they found that a working lands programme is more cost-effective than land retirement for low levels of environmental benefits measured by an index of multiple benefits. Only at high target levels of environmental benefits would it be cost-effective to enrol land into a land retirement programme. They also find that there can be large efficiency losses due to the pre-fixing of conservation budgets, regardless of whether a simultaneous or sequential implementation strategy is followed for these two programmes.

Moreover, in this application we are also interested in assessing the cost-effectiveness gains from auctions. More specifically we want to investigate the relative importance of gains received from environmental targeting (through the Environmental Benefits Index – EBI) *vs* gains received through adoption cost revelation through competitive bidding. This provides important policy implications, since if environmental targeting is the main source of cost-effectiveness gains then policy makers could implement also *e.g.* regionally differentiated payments on the basis of performance screens, such as an environmental benefit index, without bidding.

This chapter is organised as follows. The theoretical framework is developed and presented next. This is followed by a description of case study area (the US Corn Belt) and policy simulations and the results. The chapter concludes with a summary of main results.

Theoretical framework

The theoretical framework for this case study builds on that developed for green auctions in the Finnish case study and presented in Chapter 4. Thus, we follow Cattaneo *et al*. (2007) and assume that the government announces an agri-environmental payment in the form of a conservation auction programme, in which farmers bid competitively for a limited amount of conservation contracts. The programme aims to promote water quality improvements through reduction of sediment, nitrogen and phosphorus runoff from farm fields to watercourses.

To guide the bidding, the government reveals the weights given to the environmental performance, *E*, and the maximum bid payment, *R*. On the basis of farmers' bids, a single score value (*I*) will be computed for each application and the applications exceeding a cut-off value (I^c) will selected. Cut-off value is defined endogenously after the bids have been submitted.

The environmental performance of each bid with respect to surface water quality includes three measures/indicators: reduction of sediment, nitrogen and phosphorus runoff. Nutrient runoff can be reduced through reducing N and P fertilizer application rates or by establishing buffer strips between farm fields and waterways. Farmers may also adopt conservation tillage practices, such as no-till, in order to reduce both sediment and nutrient runoff.

The focus here is on practice adoption – including fertilizer application intensity, tillage practices and establishment of buffer strips – as the means of reducing both nutrient and sediment runoff. Nitrogen runoff in a given land productivity class *i*, $Z^i_{j_N} = g^i_{j_N}(l^i_j, m^i_j)$, for tillage practice/crop rotation/erodibility combination *j* can be expressed as a function of fertilizer use l^i_j and the share of parcel allocated to buffer strip m^i_j. Phosphorus runoff is defined similarly by $Z^i_{j_P} = g^i_{j_p}(l^i_j, m^i_j, \varphi^i_j)$ as a function of fertilizer use and buffer strip but it also depends on soil phosphorus status φ^i_j. Sediment runoff is given by $Z^i_{j_S} = g^i_{j_S}(m^i_j, \theta^i_j)$, where θ^i_j is the slope of the parcel.

Environmental performance, *E*, is a linear combination of water quality improvement benefits from reduction of nitrogen, phosphorus and sediment runoff,[1]

$$E^i_j(l,m) = \alpha Z_N(l,m) + \beta Z_P(l,m) + \gamma Z_S(m), \qquad (1)$$

with $0 < \alpha, \beta, \gamma < 1$ and $\alpha + \beta + \gamma = 1$ and $0 < E(l,m) \leq 1$.

Moreover,

$$E_l = \alpha Z_l + \beta Z_l < 0 \tag{2a}$$

$$E_m = \alpha Z_m + \beta Z_m + \gamma Z_m > 0 \tag{2b}$$

As in the Finnish case study, the score value I depends on the environmental performance E and the payment r required by the farmer relative to the maximum payment as a function of environmental benefit provided, $R(E)$. Moreover, the score value is defined as a share of the maximum obtainable score value, denoted by $\bar{I}$. Let ω_e and ω_r denote the weights given to the environmental performance and the payment required, respectively. Like above, $0 < \omega_e, \omega_r < 1$ and $\omega_e + \omega_r = 1$. Now, the score value can be expressed as,

$$I = \left[\omega_e E + \omega_r (1 - \frac{r}{R(E)}) \right] \bar{I}\ . \tag{3}$$

Thus, equation (3) says that the score value of each bid is a share ($0 < I \leq \bar{I}$) of the maximum obtainable score value. Clearly, it increases with environmental performance and decreases with bid.

Farmers form their bids following the above rules. To become accepted into the programme a farmer's application's index score must be above the endogenously determined cut-off value. Obviously, the farmer's bidding strategy will be guided by expectations about this cut-off value. It is assumed that the farmers are risk-neutral, so that they focus on expected values only. Thus, the farmer will submit a bid if the expected profit from participating exceeds the profits under the private optimum. The expected profits depend on the probability of being accepted in the programme. Let $\underline{I}$ denote the minimum index value to have a chance at entering the programme. Then the probability of being accepted to the programme is defined by

$$P(I > I^c) = \int_{\underline{I}}^{I} f(I)dI = F(I)\ . \tag{4}$$

Now, the farmer's expected profits can be expressed as,

$$E\pi_j^i \equiv \Pi = \left[\pi_1(l,m) - \pi_0^* + r(l,m)\right] F(I)\ . \tag{5}$$

Let $\pi_0^* = pf(l^*) - cl^* - \Omega - \mathrm{K}$ denote the farmer's profits under the privately optimal solution, with l^* the optimal fertilizer application, *p* denotes crop price, *c* denotes fertilizer price, Ω represents other variable costs of cultivation except fertilizer, and K denotes fixed capital costs. The profits under the working lands agri-environmental payment programme are conditional on the choices of fertilizer application rate *l* and buffer strip size *m* and are given as $\pi_1 = (1-m)\left[pf(l) - cl - \Omega\right] - \mathrm{K}$. Fixed capital costs of cultivation are thus not dependent on buffer strip size.

In the case of working lands payment programme the economic problem of the farmer is to choose l and m (and thereby the bid r) for a given land productivity class i and production system j so as to maximise the expected profits (5) from the bid subject to (3) and the obvious constraints $E_j^i(l,m) \leq 1$ and $r_j^i \leq R$. The Lagrangean for the problem is,

$$L = \left[\pi_1(l,m) - \pi_0^* + r(l,m)\right]F(I) + \lambda_r(R - r) + \lambda_E(1 - E) \qquad (6)$$

At an interior solution the Lagrange multipliers are zero and the first-order conditions can be expressed as

$$l^0: \quad (1-m)\left[pf_l - c\right] + r_l = -\left[\omega_e E_l + \omega_r \frac{rR_l}{R^2}\right]\frac{F'(I)}{F(I)}\Phi\bar{I} \qquad (7a)$$

$$m^0: \quad -\left[pf(l) - cl - \Omega\right] + r_m = -\left[\omega_e E_m + \omega_r \frac{rR_m}{R^2}\right]\frac{F'(I)}{F(I)}\Phi\bar{I} \qquad (7b)$$

where $\Phi = (1-m)\left[pf(l) - cl - \Omega\right] - \mathrm{K} + r(l,m) - \pi_0^*$. In both necessary conditions for the optimum, the LHS term indicates the economic costs of providing environmental goods to the programme and the RHS term indicates the expected return, that is, the effects of l and m on the score index and on the acceptance probability. In (7a), RHS bracket term is positive, so that the LHS bracket term must be positive, too, and greater than r_l, which is negative. In (7b), the RHS bracket term is negative, so that the negative LHS bracket term is greater than r_m. Conditions (7a) and (7b) provide interior solution for optimal input-use intensity under the working lands programme, that is, a programme that provides incentives for adjusting input use towards more environmentally friendly practices and outcomes on cultivated lands.

Given the above framework for green auctions in working lands raises a question whether a land retirement type of agri-environmental payment programme could be incorporated into this same theoretical frame? The answer is yes since for each land productivity class i and production system j a farmer compares the profits from participation in working lands programme $\pi_1(l^*,m^*) + r(l^*,m^*)$ with corresponding profits from participation in land retirement programme, $\pi_2(m=1, l=0) + r(l,m)$. Thus, in the case of land retirement we end up with a corner solution where whole parcel is allocated to "buffer", that is *m=1* and fertilizer use is zero *(l=0)*[2].

In the first stage, in each differential land productivity and environmental heterogeneity parcel, the farmer compares profits obtained from participating in land retirement compared to the working lands programme and then selects the option with highest profits. In the second stage, the farmer compares those profits with the profits obtained in the private optimum, $\pi_0^*(m=0, l^0)$ and decides whether or not he or she participates in the agri-environmental payment programme.

Empirical application on the basis of the US Corn Belt

The Corn Belt has been selected as the US case study area for the following reasons:

- A good mix of no-till and conventional moldboard tillage in the region, with substantial no-till and conventional/moldboard tillage acreage in major crops (corn and soybean) produced.
- A significant amount of CRP area in the region.
- A single region in the Regional Environment and Agriculture Programming Model (REAP) hosted by Economic Research Service of the United States Department of Agriculture.

The REAP model defines representative crop rotations by region, which are used to capture differences in yield, cost, and environmental coefficients. Two representative crop rotations for the Corn Belt region are selected: continuous corn and corn-soybean. The REAP model defines acreage share in Highly Erodible Lands (HEL) and Non-Highly Erodible Lands (NonHEL), with differentiation in model yield, cost and environmental coefficients.

Cost data for this case study were obtained from the ARMS cost-of-production estimates and other data sources, such as the World Resources Institute (WRI). Most of the cost items vary by tillage practice (no-till and conventional moldboard tillage). Primary cost items include: fertilizer, herbicide, machinery, fuel, and labour costs, and land rents.

With regard to environmental effects and parameters the data is generated by the EPIC (Environmental Policy Integrated Climate Model). EPIC is a crop biophysical simulation model that is used to estimate the effect of management practices on crop yields, soil quality, and environmental effects at field parcel level. Environmental data provided by EPIC includes: i) soil erosion; ii) nitrogen lost in solution; iii) nitrogen lost in sediment; iv) total nitrogen loss; v) phosphate lost in solution; vi) phosphate lost in sediment; vii) total phosphate loss; viii) pesticide runoff; and(ix) changes in soil carbon.

On the basis of above data the basic model and two auction models for working lands simulations has been developed as follows.

Crop production

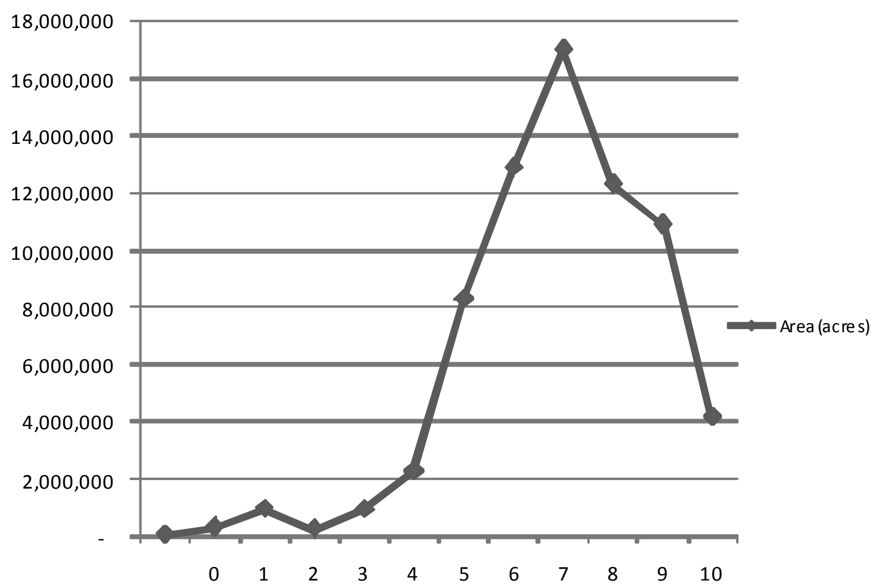
Crop nitrogen response functions (quadratic specification: a+bN-cN^2) estimated with US data were calibrated with data for eight crop/rotation/tillage/erodibility combinations with known nitrogen application level and with known yield level (see Table 6.1 for a description of eight production systems). The original value of parameter *b* from published research was retained in Nitrogen response function and then parameters *a* and *c* were solved to correspond to known nitrogen (N) application level and yield level for each combination. Thus eight Nitrogen response functions were obtained (one per each combination). It was then assumed that these response functions (and parameters) represent mean level of productivity in the case study region. On the basis of land productivity distribution in the study area (see Figure 6.1) seven land productivity classes were developed (by combining six first classes). These seven productivity classes were incorporated into the model as follows. The above-mentioned calibrated response functions were assumed to represent the mean productivity. By keeping *a* and *c* at their levels solved, the parameter *b* was solved so that there would +/- 5% change of yield (productivity) per one index point so that productivity difference would range from -15% to +15% around the mean. This provides altogether around 60 differential combinations of land productivity/crop/rotation/tillage/erodibility combinations.

Table 6.1. Descriptive abbreviation for different crop/tillage/erodibility combinations

Descriptive abbreviation	Crop(s)	Tillage method	Erodibility classification
HEL_MLD_Corn	Corn	Mouldboard	Highly erodible
HEL_NLL_Corn	Corn	No-till	Highly erodible
HEL_MLD_Corn/soy	Corn/soy	Mouldboard	Highly erodible
HEL_NLL_Corn/soy	Corn/soy	No-till	Highly erodible
NonHEL_MLD_Corn	Corn	Mouldboard	Non-highly erodible
NonHEL_NLL_Corn	Corn	No-till	Non-highly erodible
NonHEL_MLD_Corn/soy	Corn/soy	Mouldboard	Non-highly erodible
NonHEL_NLL_Corn/soy	Corn/soy	No-till	Non-highly erodible

Source: Author's classification.

Figure 6.1. Soil productivity by National Commodity Crop Productivity Index Land Class



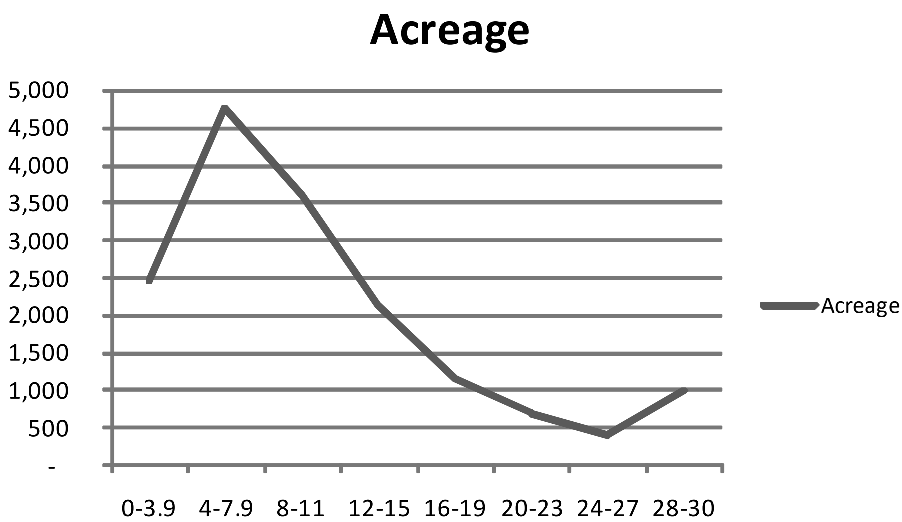
Source: Author's calculations.

Environmental process functions

On the basis of EPIC data the Secretariat has estimated functional expressions for nitrogen runoff, P-in solution runoff and P-in sediment runoff, and general sediment runoff. These functions provide the core of the environmental component of the model.

To connect the data on soil productivity and soil erodibility we apply the shares of HEL/nonHEL (Erodibility Index ≥8) and nonHEL (Erodibility Index <8) area and assume that this corresponds to the category 8-8.9 annual tonnes of soil erosion in the USLE range for the study region (see Figure 6.2). This gives a distribution in which the share of nonHEL land 88% and HEL land is 12%. Based on this information, the total acreage of HEL is about 8.5 million acres and the total acreage of nonHEL land 62 million acres.

Figure 6.2. Distribution of acreage by the USLE soil-loss category (HEL land)



Source: Author's calculations.

Environmental benefit index (EBI)

Corn Belt specific EBI weights (Cattaneo *et al.*, 2005) were used to construct a surface water quality based EBI, which is based on the weights given for Nitrogen (weight 0.22), Phosphorus (weight 0.22) and Sediment (weight 0.56) runoff plus each parcel's relative impact on these three types of runoff (as a function of nitrogen and phosphorus application intensity, tillage practices, and buffer strip widths).

We follow Cattaneo *et al.* (2005) and Claassen *et al.* (2007) and derive relative damage estimates (RDEs) for each type of runoff (nitrogen, phosphorus, and sediment) on the basis of edge-of-field runoff. Production systems with low relative damage estimates (RDEs) indicate more environmentally friendly practices and those with high estimates contribute higher quantities of pollutant runoff to watercourses. Relative damage estimates are converted to a 0-1 impact index (I_{kij}) for each runoff type:

$$I_{kij} = \left(\frac{RDE_{kij} - \min(RDE_j)}{\max(RDE_j) - \min(RDE_j)} \right) \tag{8}$$

where min(RDEj) and max(RDEj) are the minimum and maximum damage estimates across all production systems *i* and parcels *k* for the j^{th} runoff type (Cattaneo *et al.,* 2005).

Environmental indices can be specified as indices of potential environmental gain or environmental performance (Claassen *et al.*, 2007). For indices that measure potential gain, a high value shows high potential for environmental damage in the absence of abatement measures or alternatively potential for lost opportunity to improve environmental performance in the absence of environmental measures. Environmental performance index is a mirror image since index value is high when there is small opportunity for environmental gain (Claassen *et al.*, 2007). That is, when environmental performance is high, further environmental gain from measures is low.

In our analysis a performance based index is used; however, basic index calculations are for potential gain type index. Thus we follow Claassen *et al.* (2007) and convert environmental gain index to performance index as given by equation (9)

$$S_f = \max(I) - I_f \tag{9}$$

where S_f is performance-based index value for farm *f*, I_f is the potential environmental gain index value for farm and max(*I*) is the largest possible value of *I*.

Policy simulations

Alternative policy experiments in this case study are listed and described in Table 6.2. The level of the instruments is fixed arbitrarily (unless otherwise stated).

As can be seen from Table 6.2, all together 10 different policy instruments/instrument combinations are analysed and compared to the benchmark of private optimum.

Results

We start by reporting the results for the benchmark case of private optimum. The US case study model incorporates all variable and fixed costs related to each of eight different combinations of crop/rotation/tillage/erodibility (production units) and on the basis of these farmers' profits are calculated. Choice variable is nitrogen application and phosphorus application is determined on the basis of nitrogen application by assumption of fixed proportions (different for each combination) of these main nutrients in fertilizers as given by our data. Table 6.3 shows both the variable and fixed production cost items for representative production units under mean level of land productivity.

As can be seen from Table 6.4, representative production systems/units vary greatly as regards different production cost items. As empirical research has shown no-till farming entails much smaller energy (fuel) and labour costs than conventional tillage. On the other hand chemical costs are higher due to increased need to control perennial weeds under no-till.

Table 6.2. Policy experiments

Policy	Characteristics
Benchmark	
Private Optimum	No government policy intervention. Serves as a benchmark for policy experiments as regards profits and environmental performance.
Traditional regulatory and economic policy instruments	
Mandatory buffer	Regulation mandating a 2.5% buffer strip between field parcel and watercourse.
Nitrogen fertilizer tax	Fertilizer tax of 25% on the price of chemical nitrogen fertilizer.
Combination of nitrogen fertilizer tax and mandatory buffer strip	Fertilizer tax of 25% on the price of chemical nitrogen fertilizer combined with 2.5% mandatory buffer strip.
Nitrogen fertilizer application limit	Nitrogen fertilizer application limit of 100 lbs/acre.
Nitrogen fertilizer application limit and mandatory buffer	Nitrogen fertilizer application limit of 100 lbs/acre and mandatory 2.5% buffer strip.
Conservation auctions	
Conservation Auction I	Discriminatory pricing auction focusing on buffer strip establishment and fertilizer use reduction on working lands.
Conservation Auction II	Uniform pricing auction focusing on fertilizer use reduction on working lands.
Conservation Auction IIIa	Discriminatory pricing auction focusing on fertilizer use reduction on working lands with equal weights (0.5) for environmental benefits and cost factors.
Conservation Auction IIIb	Discriminatory pricing auction focusing on fertilizer use reduction on working lands with differential weights for environmental benefits (0.99) and cost factors (0.01).
Conservation Auction IIIc	Discriminatory pricing auction focusing on fertilizer use reduction on working lands with differential weights for environmental benefits (0.01) and cost factors (0.99).

Source: Author's classification.

Table 6.3. Variable and fixed costs of cultivation for different production systems/units under mean productivity

Descriptive abbreviation	Variable costs USD/acre					Fixed costs USD/acre	
	Phosphate	Energy	Chemical	Labour	Other	Land	Other
HEL_MLD_Corn	11.32	9.86	12.60	10.27	73.46	119.00	53.96
HEL_NLL_Corn	13.07	4.82	16.20	6.92	87.02	119.00	51.28
HEL_MLD_Corn/soy	2.14	8.81	11.66	8.53	68.83	119.00	48.82
HEL_NLL_Corn/soy	13.73	4.20	15.37	5.59	75.09	119.00	46.76
NonHEL_MLD_Corn	11.06	9.86	22.04	10.27	73.12	119.00	54.61
NonHEL_NLL_Corn	13.07	4.82	17.16	6.92	74.47	119.00	51.08
NonHEL_MLD_Corn/soy	3.36	8.81	11.66	8.53	60.45	119.00	48.69
NonHEL_NLL_Corn/soy	13.73	4.20	15.37	5.59	63.50	119.00	46.56

Source: Author's calculations.

Table 6.4. Private optimum: Input use, production, profits and environmental impacts under mean productivity

Production system	Crop yield	Nitrogen applied	Nitrogen runoff	Phosphorus applied	Phosphorus runoff	Soil erosion	Profit
	tonnes/acre	lbs/acre	lbs/acre	lbs/acre	lbs/acre	tonnes/acre	USD/acre
HEL_MLD_Corn	2.51	104	15.3	0.3	0.5	9.3	119.6
HEL_NLL_Corn	3.76	120	2.0	29.7	6.8	2.4	333.5
HEL_MLD_Corn/soy	4.10	54	6.3	4.9	0.9	9.8	303.8
HEL_NLL_Corn/soy	4.66	68	1.2	31.2	7.3	2.8	360.9
NonHEL_MLD_Corn	4.27	145	25.0	25.1	7.6	8.0	410.3
NonHEL_NLL_Corn	4.27	120	2.1	29.7	8.4	1.8	448.8
NonHEL_MLD_Corn/soy	5.14	54	6.4	7.6	5.2	8.2	494.7
NonHEL_NLL_Corn/soy	5.66	68	1.4	31.2	8.7	2.0	528.3

Source: Author's calculations.

Table 6.4 shows input use, production, profits and environmental impacts under the privately optimal solution without government intervention. In reviewing yields obtained under different production systems/units, an interesting feature is that no-till yields are higher than yields under conventional tillage when comparison is made within the same rotation and erodibility category. On the other hand yields are higher for NonHEL lands than HEL lands. As regards input-use intensity there is a significant variation between different production systems so that no-till is more intensive as regards nitrogen application than conventional tillage in all but one case (NonHEL_MLD_Corn). Phosphorus application is determined on the basis of nitrogen application by assumption of fixed proportions, but it is different for each production system and thus there are quite significant differences across systems as regards phosphorus application intensity.

Table 6.4 shows that despite higher nitrogen application intensity in no-till farming, the nitrogen runoff is clearly lower than under conventional tillage, while the opposite holds in the case of phosphorus runoff. Soil sediment erosion is naturally much lower in NonHEL lands than under HEL lands and it is also lower in no-till farming relative to conventional tillage. Farmers' profits are consistently higher under no-till than conventional tillage, and they are also much higher for NonHEL lands than HEL lands.

Analysis of traditional policy instruments

As regards the analysed policy instruments and their environmental and economic impacts, Table 6.5 compares two basic policy instruments to address surface water quality issues, namely the establishment of mandatory buffer strips (2.5% of cultivated area) and setting a tax on chemical nitrogen fertilizer (25%).

Table 6.5. Results: 2.5% buffer strip requirement and 25% tax on fertilizer price

	Buffer strip (2.5 %)				Fertilizer tax (25 %)			
Production system	Nitrogen runoff	Phosphorus runoff	Soil erosion	Profits	Nitrogen runoff	Phosphorus runoff	Soil erosion	Profits
	lbs/acre	lbs/acre	tonnes/acre	USD/acre	lbs/acre	lbs/acre	tonnes/acre	USD/acre
HEL_MLD_Corn	10.5	0.4	6.5	116.6	15.0	0.5	9.3	119.4
HEL_NLL_Corn	1.4	5.8	1.7	325.1	2.0	6.6	2.4	333.4
HEL_MLD_Corn/soy	4.3	0.8	6.8	296.2	6.2	0.9	9.8	303.8
HEL_NLL_Corn/soy	0.8	6.3	1.9	351.9	1.2	7.2	2.8	361.0
NonHEL_MLD_Corn	17.1	6.5	5.5	400.1	24.4	7.6	8.0	410.2
NonHEL_NLL_Corn	1.4	7.2	1.2	437.6	2.1	8.4	1.8	448.8
NonHEL_MLD_Corn/soy	4.4	4.5	5.7	482.3	6.4	5.2	8.2	494.6
NonHEL_NLL_Corn/soy	0.9	7.5	1.4	515.1	1.3	8.7	2.0	528.4

Source: Author's calculations.

As can be seen from Table 6.5, the establishment of mandatory buffers of 2.5% of cultivated area in each field parcel is quite effective as regards both nutrient runoff reduction and erosion control.[3] When compared to private optimum, farmers' profits are reduced by 2.5% while nitrogen runoff decreases over 31%, phosphorus runoff by almost 15% and soil erosion by almost 31%. Thus, mandatory buffer seem to provide quite cost-effective policy intervention for addressing water quality issues. The story is quite different in the case of fertilizer tax, however, and indeed our results just confirm the well established empirical result that fertilizer taxes need to be quite high to be effective as regards nutrient runoff reduction. As Table 6.5 shows, a 25% tax on nitrogen fertilizer has almost zero impact on farmers' profits while reducing nitrogen runoff on average by only 1.3% and phosphorus runoff less than 1%.

Table 6.6 shows the effectiveness of instrument mixes/combinations to reduce nutrient runoff and soil erosion. In theory both of these instrument-mixes should perform well, since the instruments combined do complement each other in reducing nutrient runoff, that is, fertilizer tax or application limit reduces fertilizer application while buffer strips reduce the surface runoff nutrients. However, results in Table 6.6 show that the instrument mix combining mandatory buffer and fertilizer tax mainly relies on buffer strips as regards the environmental effectiveness because additional gain over the single instrument policy of mandatory buffer is quite marginal (additional gain is only 1% or so). The combination of nitrogen application limit (100 lbs/acre) and mandatory buffer provides the instrument combination with much more additional gain over the single-instrument mandatory buffer strip. With on average 9% reduction of farmers' profits, nitrogen runoff is reduced by 39.5% and phosphorus runoff by 21.2%.

Table 6.6. Results: Combination of nitrogen tax and buffer strip and combination of nitrogen application limit and buffer strip

Production system	Nitrogen tax and buffer strip				Nitrogen application limit and buffer strip			
	Nitrogen runoff	**Phosphorus runoff**	**Soil erosion**	**Profits**	**Nitrogen runoff**	**Phosphorus runoff**	**Soil erosion**	**Profits**
	lbs/acre	lbs/acre	tonnes/acre	USD/acre	lbs/acre	lbs/acre	tonnes/acre	USD/acre
HEL_MLD_Corn	10.3	0.4	6.5	116.4	9.8	0.4	6.5	115.2
HEL_NLL_Corn	1.4	5.7	1.7	325.1	1.1	4.4	1.7	293.3
HEL_MLD_Corn/soy	4.3	0.8	6.8	296.2	4.3	0.8	6.8	296.2
HEL_NLL_Corn/soy	0.8	6.1	1.9	352	0.8	5.8	1.9	351.9
NonHEL_MLD_Corn	16.7	6.5	5.5	400	8.8	5.5	5.5	258.7
NonHEL_NLL_Corn	1.4	7.1	1.2	437.6	1.2	6.4	1.2	405.8
NonHEL_MLD_Corn/soy	4.4	4.5	5.7	482.3	4.4	4.5	5.7	482.3
NonHEL_NLL_Corn/soy	0.9	7.4	1.4	515.2	0.9	7.3	1.4	515.1

Source: Author's calculations.

Table 6.7 presents average abatement costs (USD/lb of N runoff) for alternative basic policy instruments including mandatory buffer, nitrogen application limit, and the instrument combinations of nitrogen tax with buffer strip, and nitrogen application limit with buffer strip.

Table 6.7 shows that there is a huge variation in the average abatement costs both across production systems and across policy instruments. However, one should note that if nitrogen application limit is not binding (shown by zero adoption cost) then the average abatement cost is mainly driven by adoption cost of establishing mandatory buffer. This can be seen for example, in the case of following production systems: HEL_MLD_Corn/soy, HEL_NLL_Corn/soy, NonHEL_MLD_Corn/soy, and NonHEL_NLL_Corn/soy. However, clearly the average abatement costs are much higher for the more profitable tillage practice no-till.

Analysis of conservation auctions

After analysing conventional policy instruments and instrument combinations it is time to analyse how new policy approaches, namely alternative types of conservation auctions, perform relative to the private optimum and traditional agri-environmental policy instruments.

Table 6.7. Average abatement cost (USD/lb of N runoff) for alternative policy scenarios

Production system	Average abatement cost, USD/lb of N runoff			
	Mandatory buffer	N application limit	N Tax + buffer	N limit + buffer
HEL_MLD_Corn	1	1	1	1
HEL_NLL_Corn	13	92	13	46
HEL_MLD_Corn/soy	4	0	4	4
HEL_NLL_Corn/soy	24	0	24	23
NonHEL_MLD_Corn	1	12	1	9
NonHEL_NLL_Corn	17	102	17	49
NonHEL_MLD_Corn/soy	6	0	6	6
NonHEL_NLL_Corn/soy	31	0	30	30

Source: Author's calculations.

Table 6.8 shows basic results for *Conservation Auction II* that employs uniform pricing payment format and focuses on nitrogen application reduction in different production units under mean productivity and erosion. It is supposed that the farmers estimated adoption costs are equal their true adoption costs which may not always be the case in practice.

Table 6.8. Results for uniform pricing auction

Production system	Private N application	Auction N application	CC_env	Bid value USD	Environ-mental performance	Benefit/ cost ratio
HEL_MLD_Corn	104	100	1.4	1.4	0.342	0.2
HEL_NLL_Corn	120	100	32.6	32.6	0.608	0.0
HEL_MLD_Corn/soy	54	54	0.1	0.1	0.379	5.0
HEL_NLL_Corn/soy	68	65	1.0	1.0	0.560	0.6
NonHEL_MLD_Corn	145	100	145.0	145.0	0.259	0.0
NonHEL_NLL_Corn	120	100	32.6	32.6	0.577	0.0
NonHEL_MLD_Corn/soy	54	53	0.1	0.1	0.334	5.7
NonHEL_NLL_Corn/soy	68	66	0.3	0.3	0.543	1.8

Source: Author's calculations.

As Table 6.8 shows, the uniform price auction reveals farmers' true adoption costs since all production units bid exactly the amount of their true adoption cost for nitrogen use reduction. As regards the last two columns dealing with environmental performance and benefit-cost ratio (B_i/C_i) of each bid one can see that relative ranking of bids would be different if targeting would be based on environmental performance or benefits instead of a benefit-cost ratio.

Tables 6.9 and 6.10 introduce spread (range) of the basic results presented in Table 6.8 for increased (+15%) and decreased (-15%) land productivity with mean erosion.

Table 6.9. Uniform price auction with -15% decrease in land productivity

Production system	Productivity	Private N application	Auction N application	CC_env	Bid value USD	Environ-mental performance	Benefit/ cost ratio
HEL_MLD_Corn	-15%	100	98	0.4	0.4	0.346	0.9
HEL_NLL_Corn	-15%	114	100	17.1	17.1	0.608	0.0
HEL_MLD_Corn/soy	-15%	50	49	0.1	0.1	0.384	6.0
HEL_NLL_Corn/soy	-15%	63	60	0.8	0.8	0.579	0.7
NonHEL_MLD_Corn	-15%	139	100	108.5	108.5	0.259	0.0
NonHEL_NLL_Corn	-15%	114	100	15.4	15.4	0.577	0.0
NonHEL_MLD_Corn/soy	-15%	48	47	0.1	0.1	0.340	6.6
NonHEL_NLL_Corn/soy	-15%	62	60	0.3	0.3	0.558	2.2

Source: Author's calculations.

Table 6.10. Uniform price auction with +15% decrease in land productivity

Production system	Productivity	Private N application	Auction N application	CC_env	Bid value USD	Environ-mental performance	Benefit/ cost ratio
HEL_MLD_Corn	15%	107	100	5.4	5.4	0.342	0.1
HEL_NLL_Corn	15%	125	100	53.1	53.1	0.608	0.0
HEL_MLD_Corn/soy	15%	59	58	0.1	0.1	0.373	4.2
HEL_NLL_Corn/soy	15%	73	69	1.3	1.3	0.538	0.4
NonHEL_MLD_Corn	15%	151	100	186.9	186.9	0.259	0.0
NonHEL_NLL_Corn	15%	126	100	56.3	56.3	0.577	0.0
NonHEL_MLD_Corn/soy	15%	59	58	0.1	0.1	0.327	4.8
NonHEL_NLL_Corn/soy	15%	74	72	0.4	0.4	0.528	1.5

Source: Author's calculations.

Tables 6.9 and 6.10 show that increase in land productivity (from -15% to +15% around mean productivity) increases privately optimal fertilizer application and thus it increases opportunity costs of environmental measures (adoption costs). Thus, farmers' bids are much higher (on average over 200% higher) while environmental performance is decreased 3% and benefit-cost ratio is weakened by 51%.

Table 6.11 combines 11 simulations and these simulations are all discriminatory payment format auctions but they differ as regards weight given for environmental performance and bid. The results are expressed as average values for eight production systems with mean land productivity and erosion.

Table 6.11. Discriminatory payment auction: impact of weights for auction performance

Auction	N_env	CC_env	Bid	Environmental performance	B/C
Environment	78.7	27.3	190.8	0.452	0.0024
Environment 0.9, Cost 0.1	78.7	27.3	190.8	0.452	0.0024
Environment 0.8, Cost 0.2	78.7	27.3	190.8	0.452	0.0024
Environment 0.7, Cost 0.3	78.9	27.2	184.7	0.452	0.0024
Environment 0.6, Cost 0.4	79.3	26.9	170.2	0.451	0.0027
Environment 0.5, Cost 0.5	79.6	26.6	151.4	0.450	0.0030
Environment 0.4, Cost 0.6	79.8	26.5	137.2	0.450	0.0033
Environment 0.3, Cost 0.7	80.0	26.5	127.0	0.449	0.0035
Environment 0.2, Cost 0.8	80.1	26.5	119.3	0.449	0.0038
Environment 0.1, Cost 0.9	80.2	26.4	113.4	0.448	0.0040
Cost	80.2	26.4	109.0	0.448	0.0041

Source: Author's calculations.

Table 6.11 shows how assuming different weights affects auction markets and resulting environmental responses. Placing more weight to environmental performance naturally reduces nitrogen fertilizer application and increases adoption costs and thus also farmers' bids. Environmental performance slightly increases, however, it is dominated by the increase in costs and thus benefit-cost ratio worsens slightly. And when higher weight is assumed for cost/bid then opposite holds so that nitrogen fertilizer application slightly increases, adoption costs and bids decrease, and environmental performance slightly decreases while a benefit-cost ratio improves slightly.

Summary of the US case study

This chapter focuses on the economic and environmental performance of conservation auctions relative to more traditional agri-environmental policy measures. The economic and environmental effects are however not aggregated. In this application the sources of heterogeneity are both differential land productivity and environmental sensitivity of the land, more specifically differential propensity to erosion and thus nutrient and sediment runoff. The analysed policy instruments range from traditional regulatory and economic instruments, including fertilizer application limits and taxes to different types of conservation auctions including both uniform and discriminatory pricing types of auctions. Conservation auctions employ environmental benefit indices as environmental performance screens that help to target conservation effort to parcels that provide large environmental benefits.

As regards traditional policy instruments the regulation mandating the allocation of 2.5% of land along watercourses as vegetated buffers effectively reduces sediment and nutrient runoff with reasonably small adoption costs to farmers. The combination of a mandatory buffer with a fertilizer tax (25%) to reduce application intensity provides only small additional environmental gains over a mandatory buffer alone, while the combination of a nitrogen application standard and a buffer strip is much more effective. This result underscores the well known problem with fertilizer taxes – they need to be very high to have an impact on behaviour. Hence, the combination of a nitrogen application limit and a mandatory buffer provides the instrument combination that is superior to other traditional policy instruments.

As regards conservation auctions the application of a uniform pricing auction reveals farmers' estimated adoption costs and thus their information rent is reduced and budgetary cost-effectiveness is increased. On the other hand, a discriminatory payment format gives farmers an incentive to place their bids above their adoption costs: low adoption cost farmers have a greater incentive to do so than high adoption cost farmers. Changing the weight between environmental performance and cost/bid affects optimal fertilizer-use intensity, farmers' adoption costs, farmers' bids, and environmental performance for a given budget.

Notes

1. To avoid unnecessary notation we drop the superscript i and subscript j from the choice variables l and m.

2. It should be noted that changes in the width of buffers would not affect reductions in nutrient runoff in a linear fashion.

3. Note that buffer strips can naturally be voluntary as well, for example, through contracts.

Chapter 7
Japan: Optimal land-use allocation and nitrogen application

This case study investigates the optimal land-use allocation and nitrogen application under a representative Japanese farm that consists of a rice paddy field, upland field and land abandonment. Rice paddy fields, which are the typical land-use type in Asian monsoon conditions, are analysed in this study. Conducting a case study using Japanese data by incorporating characteristics of paddy field cultivation into the SAPIM framework provides interesting extension to the SAPIM studies.

Japan is currently undertaking market-oriented agricultural policy reforms and is accelerating the implementation of agri-environmental policies although they still constitute a very small part of the overall policy package. For this case study, data and some other information were provided by Japanese Ministry of Agriculture, Forestry and Fisheries (MAFF) and national research institutes. Direct payments for core farmers are linked to the application of these principles as a cross-compliance measure (MAFF, 2008a; OECD, 2009).[1]

Policy context and analytical framework

The transition to environmentally friendly farming is being encouraged in accordance with the *Principles of Environmental Policy in Agriculture, Forestry and Fisheries* (2003). *The Agricultural Environmental Code* agreed in 2005 requires the necessary production practices that farmers should adopt for environmental conservation. This code initiated a further movement to cross-compliance measures targeted to environmentally beneficial practices. New direct payments for core farmers are linked to the application of these principles as a cross compliance measure (MAFF, 2008a; OECD, 2009).

For environmentally-friendly farming practices which must go beyond the “reference level”, the government provides additional support for farmers as incentives. “Eco-farmers” who adopt sustainable agricultural practices are encouraged by concessionary loans. Moreover, direct payments for environmentally pioneering farming and the promotion of organic farming based on the *Law for Promoting Organic Farming,* enacted in 2006, are being implemented. Support and incentives for agri-environmental farming practice are summarised in Table 7.1. In addition to promoting agri-environmental farming, recent policies strengthen agri-environmental programmes, such as promoting the production of bio-energy derived from non-food materials, mitigation and adaptation to global warming and biodiversity conservation.[2] Based on the *MAFF’s Comprehensive Strategies for Global Warming,* which was enacted in 2008, and *The Role of Agricultural Soil in Preventing Global Warming* (MAFF, 2008c), counter measures for GHG mitigation from farmland, such as incentives for farmers, are under discussion.

Table 7.1 Current agri-environmental policy measures[3]

Policy aim	Details	Policy measures
Adoption and compliance with the Agricultural Environment Code	The adoption and compliance with the codes for Agricultural Practice in Harmony with the Environment (Agricultural Environment Code)	A condition of environmental cross compliance
Support for eco-farmers	Based on the *Law for Promoting the Introduction of Sustainable Agricultural Practices*, promoting certification of "Eco-farmers" who practice sustainable farming and support their activities by financial and technical support. The number of eco-farmers was 167 995 at the end of March 2008.	Concessionary loans
Support for pioneering farming	*Measures to Conserve and Improve Land, Water and Environment* was introduced in 2007, which support the progressive farming activity contributing to the conservation of local environments by reducing the use of chemical fertilizers and synthetic agri-cultural chemicals by more than 50% compared with conventional application.*	Direct payments (agri-environmental payment)
Support for organic farming	In accordance with the Law for Promoting the Organic Farming which was established 2006, no-chemical fertilizer and no-pesticide farming is promoted.	Concessionary loans; Tax relief; Direct payment (in the context of support for pioneering farmer)

* There is another scheme to support for maintaining and promoting the "natural-circulation" function of agricultural ecosystem (*e.g.* biodiversity, landscape).

Source: MAFF (2008b).

Before starting the modelling phase of SAPIM, it is useful to clarify the country specific agri-environmental issues and policy targets to appropriately reflect the actual situation. As shown in Table 7.2, the number of eco-farmers (step 1) has been used as a general indicator (see also OECD, 2008b). But, according to the discussion in MAFF (2008b), indicators such as the amount of organic matter applied, the amount of chemical fertilizer and pesticide use (step 2) which have important environmental effects, are set as policy targets (indicators) for the coming period.

It is difficult to define and measure actual environmental damage and benefits, such as water quality, GHG emissions and biodiversity richness (step 3), as policy targets. However, for effective policy design and evaluation, measurement is needed. In brief, step 1 is the easiest policy target (indicator) compare with step 2 (input-based) and step 3 (performance-based) targets.

Currently, the lack of monitoring data impairs the evaluation of agri-environmental performance at the national level. In addition, "little is known of the relative costs and benefits of using agricultural land to provide ecosystem services, especially rice paddy, compared to other land-use types" (OECD, 2008). Consequently, so as to capture cause-effect linkage, micro level policy analysis which integrates economic and biophysical modelling is necessary.

In Japan, farm nitrogen and phosphorus surpluses declined over the period 1990-92 to 2002-04, but absolute levels per hectare remain among the highest across OECD countries. On the other hand, agriculture can supply certain ecosystem services depending on their management, and rice paddies provide a higher level of ecosystem service (*e.g.* water-retaining capacity) than other types of agricultural land use. But due to the decrease in the farm area, provision of ecosystem services, wild species diversity and value of landscape have been impaired. These country-specific characteristics should be taken into account in the SAPIM case study.

Table 7.2. Agri-environmental policy objectives and indicators

	Indicators (example)		
Objectives	**Step 1**	**Step 2**	**Step 3**
Improve water quality	Number of farmers who conduct agri-environmental farming	Chemical fertilizer use per hectare	Water quality in public water body
Improve air quality (GHG)		Chemical fertilizer use	Level of greenhouse gases in the atmosphere
Improve soil conservation		Organic matter application	Soil fertility index (N, P and K)
Improve biodiversity		Pesticide use	Number of species or lives
Greater sustainable use of organic matter		Organic matter application	% of sustainable usage

Source: MAFF (2008b).

In this model, “nitrogen runoff” which might cause eutrophication[4] and “GHG emission (sequestration)”[5] are considered as environmental externalities. As indicated in Table 7.2, “water quality improvement” and “improve air (GHG)” are part of the key objectives in Japanese agri-environmental policy, and also are possible to capture quantitatively. Recently, improving biodiversity is under the spotlight in Japan,[6] but there is no clear evidence between agricultural inputs and the level of biodiversity, which makes it difficult to incorporate the relationship into the quantitative model.

Following the Finnish SAPIM study, land is divided into rectangular parcels which are of the same size and of homogeneous land quality (productivity: q), but heterogeneous as between parcels. It is assumed that each parcel is 10 ares and the total cultivated land is 6 ha (60 parcels). Policy support is concentrated on “core farmers”, who are defined as efficient and cultivating over 4 ha. Consequently, the farm size in the model needs to be set larger than 4 ha in order to reflect various support measures.

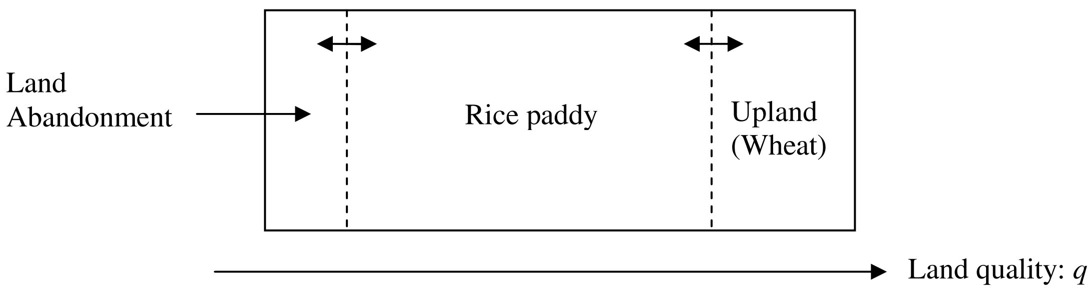
To estimate the profit function, national statistics data on 5-7 ha farm size is used, although the average Japanese farm size is approximately 1.36 ha. This assumption will not cause a big difference in social welfare estimation because the impact of the input-related externality (*e.g.* nitrogen runoff) is constant with farm size in this model.

The land-use classification is assumed as rice, upland crop production (or cultivation) and abandonment in this study (Figure 7.1). The upland crop area, which has been converted from rice paddy, is assumed to be increasing in the model. The area of temporarily converted upland field from paddy fields by draining water is 740 000 ha, accounting for 30% of cultivated paddy. Wheat is assumed here to be the upland crop.

There is some trade-off between paddy fields and upland crops with respect to environmental externalities. For example, the amount of methane emission from upland fields is zero, while those of N_2O emissions are higher (Nishimura *et al.,* 2004). Consequently, it is valuable to analyse both rice and the upland crop cultivation in a continuous analytical framework by formulating their main characteristics from both economic and environmental perspectives.

Suppose that the land reform in paddy fields (drainage canal and sub-surface drainage) has already been undertaken. This means that it is possible that a farmer can decide the land allocation only by reference to the profit from each parcel, and so it is not necessary to incorporate "the land conversion cost" exogenously.

Figure 7.1. Spatial characteristics used in the Japanese SAPIM



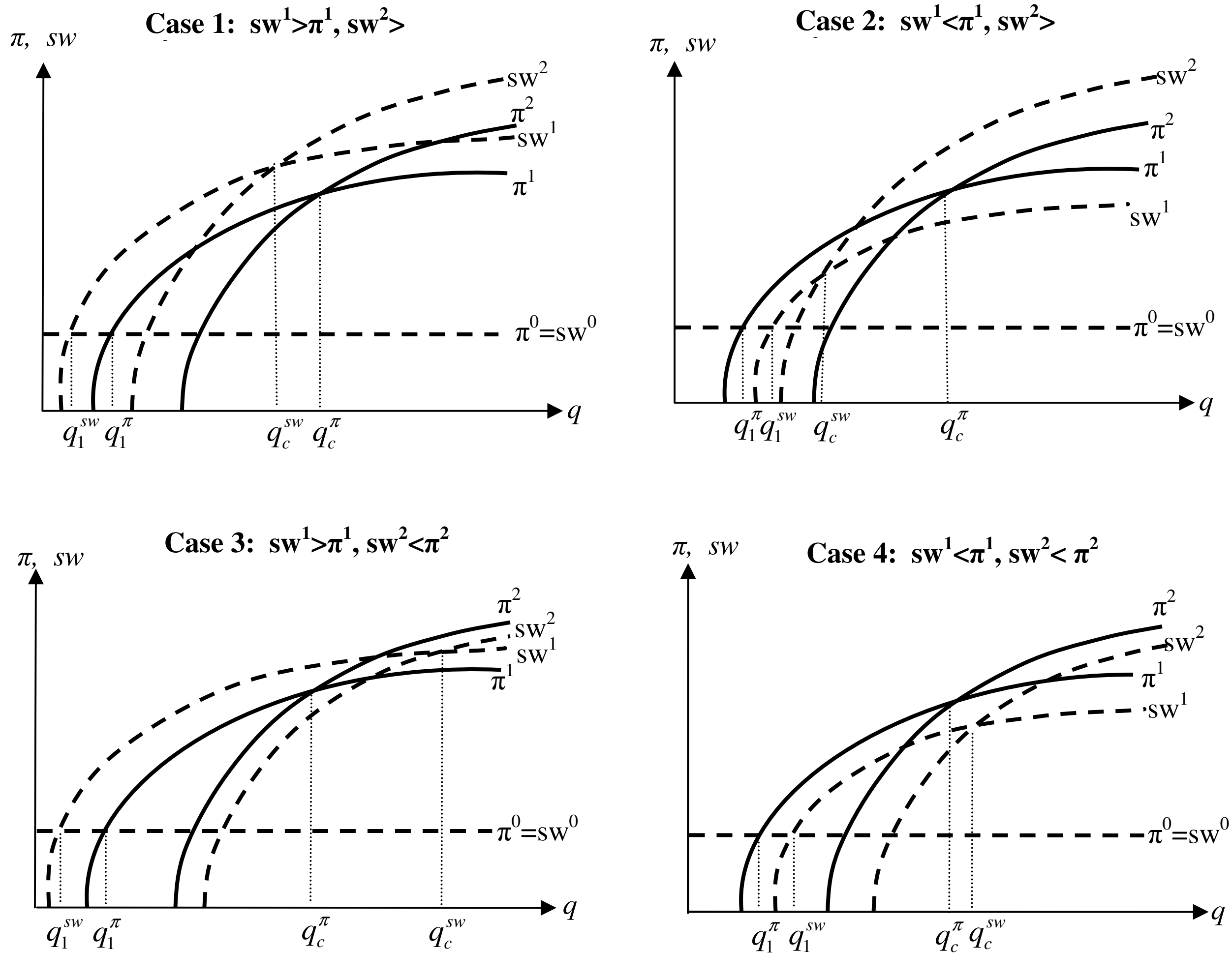
Source: Author.

Lichtenberg (1989; 2002), Lankoski and Ollikainen (2003), Lankoski *et al*. (2004), Ollikainen and Lankoski (2005) and Lankoski *et al*. (2006) have developed a framework for analysing the joint production of commodity and environmental outputs as well as negative externalities under heterogeneous land quality and this is the point of departure for this modelling exercise.

Estimating the absolute level of environmental effects is often difficult, and optimal land use might change depending on the relationship of absolute levels of externalities in the model. When π_i, sw_i and q_i refer to the private profits, the social welfare and land quality respectively, Figure 7.2 illustrates the land allocation between abandonment (π_0, sw_0), rice (π_1, sw_1) and wheat (π_2, sw_2) with respect to the private and social optima, depending on the magnitude of the relative negative and positive externalities. In Case 1 of Figure 7.2, positive externalities are greater than negative externalities for both crop 1 and 2, while social welfare in abandonment is considered as same in private optimum. The entry-exit margin of crop production has changed to q_1^{sw}, and the land switching point is q_c^{sw} in the social optimum. Alternatively, Case 4 illustrates the opposite case. As shown, the optimal land allocation generates very different degrees of environmental effects.

The conditions of the Japanese case correspond to Case 3, but further incorporation of alternative environmental externalities might bring different results for optimal land allocation. Consequently, it is important to place more emphasis on the *relative effect* of each agri-environmental policy on private and social welfare.

Figure 7.2. Private and social optimal land allocation under heterogeneous land productivity: Different cases



Source: Author, modified from Lichtenberg, 2002.

Theoretical framework

In the model, the optimal land-use allocation and nitrogen application (the combination of chemical fertilizer and organic fertilizer) were considered under a representative farm. The model consists of quadratic Nitrogen response functions for cultivated crops, exponential nitrogen runoff (purification) functions, and GHG emission functions which combine CH_4, N_2O and carbon sequestration functions. Producer profits are maximised under exogenous crop prices and input costs.

Following Lankoski and Ollikainen (2003) and the Finnish case study, let $G(q)$ denote the cumulative distribution of q (acreage having quality q, $0 \leq q \leq 1$), while $g(q)$ is its density, and assume that $g(q)$ is continuous and differentiable. The total amount of land in the region is

$$G = \int_0^1 g(q)dq. \tag{1}$$

It is assumed that only rice and wheat are cultivated in this region, $i = 1, 2$. Both crops are produced under constant returns to scale. Output of each crop per unit of land area is denoted by y_i, and yield is a function of land quality q and the fertilizer application x_i. The applied amount of fertilizer x_i is the combination of chemical fertilizer x_{ci} and organic fertilizer x_{oi}, $y_i = f^i(x_i; q)$. This production function is increasing and concave in fertilizer and land quality. Assume that the arable land can be allocated to either paddy rice or wheat. The share of each crop L_1 and L_2 is given by

$$L_1 = \int_0^{q_1} g(q)dq = G(q_1) \tag{2}$$

$$L_2 = \int_{q_2}^{1} g(q)dq = G(1) - G(q_1). \tag{3}$$

Land abandonment is not considered in the theoretical part in order to simplify the discussion. Two environmental effects are assumed: water quality impacts through chemical fertilizer runoff and GHG emissions through chemical and organic fertilizer application.

Crop production

The profit from agricultural production is expressed as,

$$\pi^i = p_i f^i(x_i; q) - cx_i, \ i = 1, 2 \tag{4}$$

Here p_i refer to the price of crops and c to the fertilizer price, which are both taken as given.

Organic fertilizer cause yield-increase effect, which effect depends on the amount of application: $\Phi^i(x_{oi})$, defined as $1 < \Phi^i(x_{oi})$ with $\Phi^i_x > 0$ and $\Phi^i_{xx} < 0$. At the same time, the additional cost of organic fertilizer collection, transportation and spreading are incorporated in the profit function. In the presence of yield-increase effect and additional cost of organic application, the profit function is modified as follows,

$$\pi^i = p_i f^i(x_{ci}, x_{oi}, q)\Phi^i(x_{oi}) - c^i(x_{ci}, x_{oi}), i = 1,2 \quad (5)$$

Nitrogen runoff and purification

Aggregate N runoff is a function of chemical fertilizer use. Suppose that the N content in organic fertilizer is not included in the N runoff function, because N in organic fertilizer could be serious problem only when the application amount is very large. In this model, the maximum application of organic is approximately 1.5 t/10 ares due to economic factors (high additional cost). The runoff of nutrients (kg) from each parcel is expressed as a function of chemical fertilizer applied x_{ci} as

$$z_i = v_i\left[x_{ci}(q)\right] \quad for\ i = 1,2 \quad (6)$$

with $v_x > 0, v_{xx} > 0$. Thus, the runoff function is convex in the fertilizer application. It is well known that paddy fields effectively improve water quality by removing nitrogen due to denitrification and absorption. When the total nitrogen inflow in the paddy field water exceeds the total outflow of nitrogen discharged out of the paddy field water, the paddy field works as a nitrogen removal site, which means z_1 is negative. Then total amount of runoff from the land area devoted to rice and wheat as

$$z = \int_0^1 \left\{v_1\left[x_{c1}(q)\right]L_1 + v_2\left[x_{c2}(q)\right](1-L_1)\right\}g(q)dq. \quad (7)$$

The monetary valuation of runoff damages (purification benefit), defines a valuation function, $D(z)$, which is assumed to be convex ($D(\cdot)_x > 0, D(\cdot)_{xx} < 0$)

GHG emission and sequestration

Regarding GHG emissions, agriculture is an important anthropogenic source of CH_4 and N_2O. In addition to GHG emissions, there is the role of agricultural soils as a carbon sink.

CH_4 emissions

The impact of organic fertilizer for CH_4 emissions is critical (Yan *et al*., 2005), and the amount of the applied material and CH_4 emission can be described by a response curve. Methane generation is not possible if soil is not maintained in an anaerobic state. Upland soils are normally oxidative and in aerobic condition, therefore CH_4 is not produced. CH_4 emission is denoted as

$$CH_4 = \int_0^1 \left\{m\left[x_{o1}(q)\right]L_1\right\} g(q)dq \quad (8)$$

with $m_x > 0, m_{xx} < 0$. Thus, the runoff function is concave in the organic fertilizer application (Yan *et al.;* 2005; IPCC; 2006).

N_2O emissions

Following the guideline of the Intergovernmental Panel on Climate Change (IPCC), if the N_2O emission is a combination of direct emissions (denitrification) and indirect emissions (associated with atmospheric deposition and nitrogen runoff),

$$N_2O = \int_0^1 \left\{ \left[n_1(x_{c1}(q), x_{o1}(q), z_1) \right] L_1 + \left[n_2(x_{c2}(q), x_{o2}(q), z_2) \right](1 - L_1) \right\} g(q)dq \quad (9)$$

with $n_x > 0, n_{xx} < 0$ then the emission function is concave in the fertilizer application.

Carbon sequestration

Soil carbon stock is affected heavily by fertilizer management as well as CH_4 and N_2O emissions from agricultural land. Appropriate amounts of organic fertilizer could increase the soil carbon content and stimulate the total GHG emission reduction. The carbon sequestration function is

$$Seq = \int_0^1 \left\{ s_1 \left[x_{o2}(q) \right] L_1 + s_2 \left[x_{o2}(q) \right](1 - L_1) \right\} g(q)dq \quad (10)$$

with $s_x > 0, s_{xx} < 0$. Thus, the sequestration function is concave in the organic fertilizer application.

Consequently, the net GHG emission is expressed as follows,

$$\begin{aligned} e = & \int_0^1 \left\{ m\left[x_{o1}(q) \right] L_1 \right\} g(q)dq \\ & + \int_0^1 \left\{ \left[n_1(x_{c1}(q), x_{o1}(q), z_1) \right] L_1 + \left[n_2(x_{c2}(q), x_{o2}(q), z_2) \right](1 - L_1) \right\} g(q)dq \quad (11) \\ & - \int_0^1 \left\{ s_1 \left[x_{o2}(q) \right] L_1 + s_2 \left[x_{o2}(q) \right](1 - L_1) \right\} g(q)dq. \end{aligned}$$

The monetary valuation of emission damages (sequestration benefit), defines the valuation function, $GW(e)$, which is assumed to be convex ($GW(\cdot)_x > 0, GW(\cdot)_{xx} < 0$).

Features of the first-best solution

Chemical fertilizer affects both yield and environmental externalities. Moreover, there is a trade-off in organic fertilizer applications, which means organic fertilizer can help maintain soil fertility (yield-increase effect) and increase carbon sequestration. On the other hand, it could increase CH_4 emissions and be a source of water quality problems. The social welfare maximisation problem can now be expressed as

$$SW = \int_0^1 \sum_{i=1}^2 \left[pf^i(x_i(q), q)\Phi_i(x_i(q)) - cx_i(q) \right] g(q)dq + D(z) + GW\left[e(m, n, s) \right]. \quad (12)$$

The social planner chooses the use of inputs (chemical and organic fertilizer) for each parcel under heterogeneous land productivity. The first-best optimum is solved for recursively.

$$SW^i_{x_{ci}} = p_i f^i_{x_c} - c^i_{x_c} + D'(z)\frac{\partial v_i}{\partial x_{ci}} + G'(e)\frac{\partial n_i}{\partial x_{ci}} = 0 \qquad (13)$$

$$SW^i_{x_{oi}} = p_i f^i_{x_o}\Phi^i_{x_o} - c^i_{x_{oi}} + D'(z)\frac{\partial v_i}{\partial x_{oi}} + G'(e)\left[\frac{\partial m_i}{\partial x_{oi}} + \frac{\partial n_i}{\partial x_{oi}} + \frac{\partial s_i}{\partial x_{oi}}\right] = 0 \qquad (14)$$

On the basis of the optimal use of inputs and thus profits for each crop from a given land quality the land is allocated to the highest social return use in each parcel. The unique value of switching land quality, q_1 is defined by,

$$\pi_1^* + D'(\cdot)v_1 + GW'(\cdot)e_1 = \pi_2^* + D'(\cdot)v_2 + GW'(\cdot)e_2 \qquad (15)$$

Consequently, land is allocated to crops by taking into account not only profits, but also the effect of land allocation on N runoff and GHG emission.

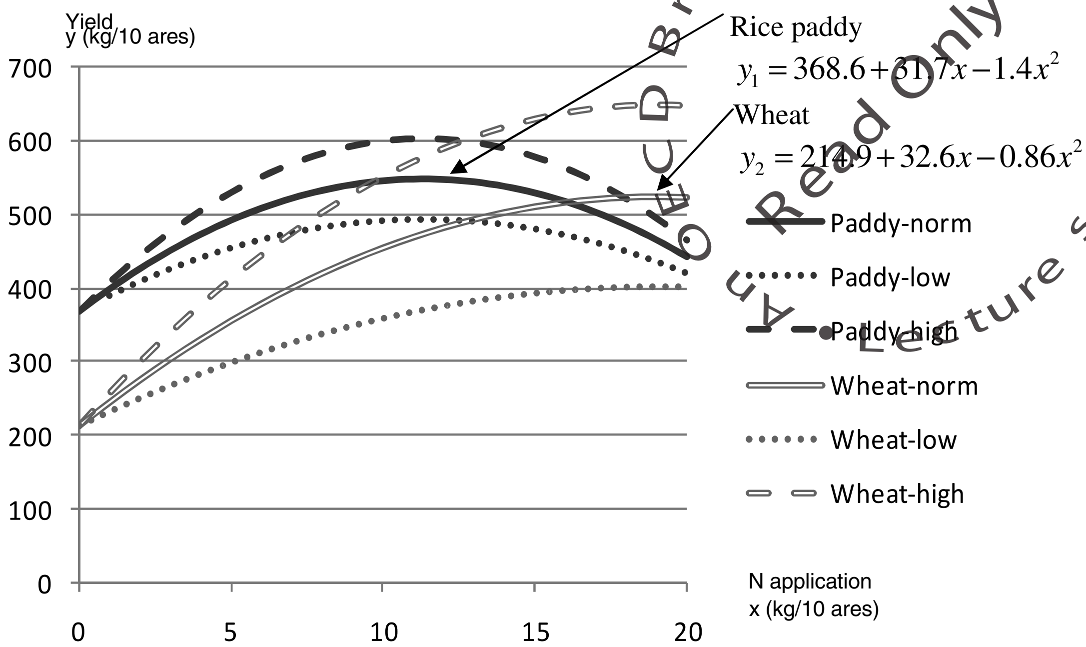
The private optimum can easily be extracted from equations (13) to (16). Under the private optimum, the farmer ignores the effects of environmental externalities. By setting the marginal damage (benefit) to zero $\pi_1^* = \pi_2^*$ is obtained.

Empirical framework

As shown in the theoretical framework, farmers consider not only the amount of total N, but also the combination rate of chemical fertilizer x_{ci} and organic fertilizer application x_{oi}. The total amount of N application to the agricultural field is the sum of N fertilizer and N content of organic matter. According to MAFF (2008b), despite the amount of organic matter applied that is recommended (*e.g.* 1.0-1.5t /10 ares for paddy field), this is not implemented (88 kg/10 ares) due to several difficulties. In this model, the trade-off for organic matter application, which has a positive effect on yield per parcel, and high costs of spreading and transportation costs, need to be considered. The estimated Nitrogen response functions and field survey data are shown in Figure 7.3. In general, wheat yield is more responsive than paddy rice to nitrogen applications. However, as shown in Annex C (Figure C.1), in the case of rice, the estimated nitrogen response is very difficult to assess.

Crop production also generates negative environmental externalities *via* nitrogen applications. But paddy fields could be N removal sites or pollution sites depending on agricultural activities and the nitrogen concentration of irrigation water. It is well known that paddy fields and wetlands effectively improve water quality by removing nitrogen due to denitrification and absorption, which is effective only when irrigation water has strong nitrogen concentration.

Figure 7.3. Nitrogen response function of rice and wheat



Source: Author's calculations.

The N removal (purification) function is expressed here by one numerical example. The basic nitrogen balance is calculated by following equation,

$$Nbalance\_paddy = F + ILi + ILp - OLs - OLp \qquad (16)$$

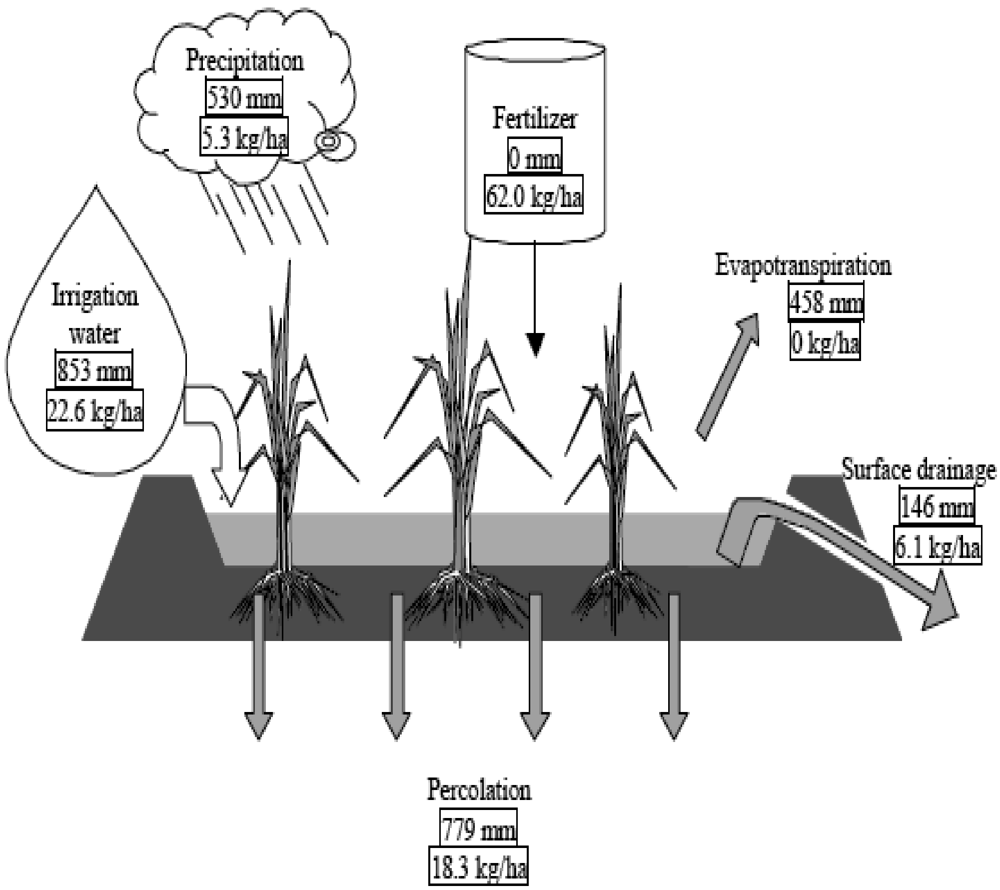
where F is load by fertilizer, ILi is inflow loads by irrigation, ILp is inflow loads by rain, OLs is outflow loads by surface discharged water, OLp is outflow loads by percolation. According to the field survey by Yoshinaga *et al*. (2003), the total nitrogen inflow in paddy field water was 27.9 kg/ha and total outflow nitrogen discharged out of the paddy field water was 24.4 kg/ha. The paddy field had a nitrogen removal capacity of 3.5 kg/ha during the observation period (Figure 7.4). So as to estimate the stylised runoff function, several data on the nitrogen application amount and runoff volume is needed. As explained in Annex C, data are, however, too scarce to build robust function estimates. Estimated exponential nitrogen runoff (purification) functions are described in Figure 7.5 and should be used with caution.

Agriculture does not account for a high percentage of total GHG emissions in Japan,[7] although agriculture is an important anthropogenic source of CH_4 and N_2O emissions. In SAPIM, since the same control variables must be used in the biophysical model and profit function (Nitrogen response function), it is important to consider the choice of control variable first. As summarised in Table 7.3, GHG emissions from agricultural land (IPCC's 4C category rice cultivation and 4D category agricultural soils) derived from chemical and organic fertilizer applications account for approximately 80% of total emissions (in italics in the table). In this analysis, therefore, fertilizer application amounts could be considered as control variables. Rice cultivation is a main anthropogenic source of CH_4 (methane) emissions. Fertilizer application and ploughing of organic soil cause ammonium ions inside the soil, and then N_2O is emitted. N_2O is also emitted *via* denitrification.

In addition to CH_4 and N_2O emissions, other research deals with the role of agricultural soils as a carbon sink. Agriculture could potentially achieve a significant reduction in the risk of climate change by taking CO_2 out of the atmosphere and storing it in the soil. Given the present circumstances, only four countries (Canada, Denmark, Portugal and Spain) have elected to include "Cropland Management and Grazing Land Management" (the key activities relevant to agricultural industries) in their accounts for the Kyoto protocol first commitment period (2008-12). No information is contained in the GHG Inventory of Japan on this category (MOE, 2008).

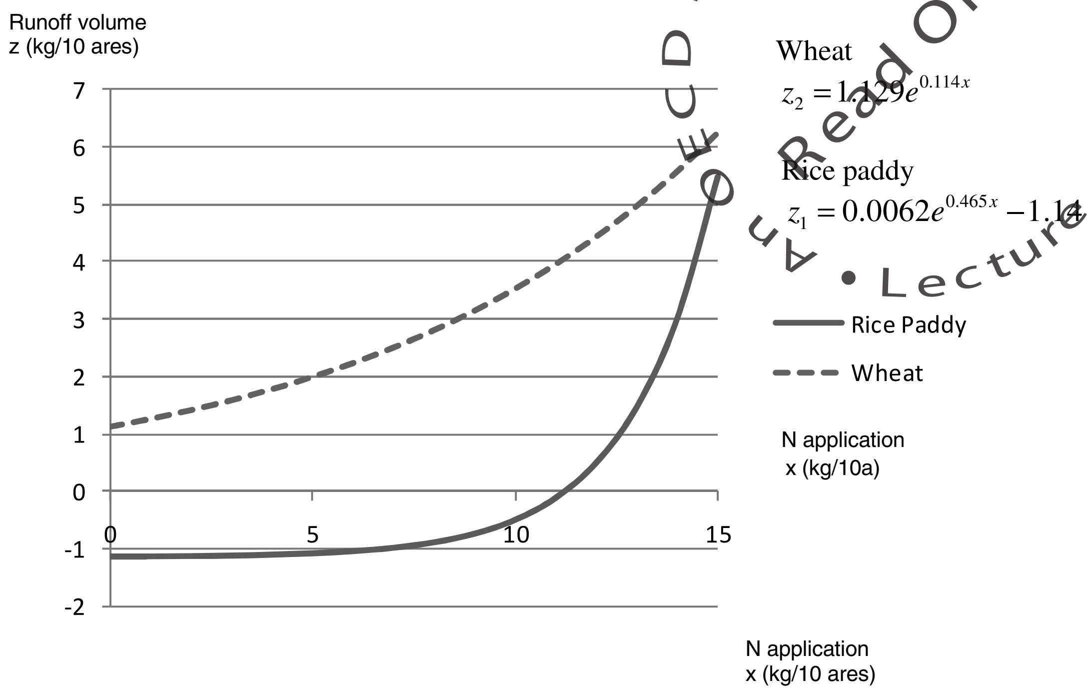
Figure 7.4. One example of water and nitrogen balance of paddy field during crop period

Top number: water (mm), bottom number: nitrogen (kg ha-1)



Source: Yoshinaga *et al.* (2003).

Figure 7.5. Shapes of estimated N runoff and purification function



Source: Author's calculations.

Table 7.3. CH_4 and N_2O emissions from rice cultivation and agricultural soils

IPCC Category			Gas	2006 emissions (Gg CO_2 equivalent)
Rice cultivation	Intermittently flooded	Straw left on field	CH_4	***3 775.78***
		Various compost materials left on field	CH_4	***981.60***
		No straw or compost left on field (baseline)	CH_4	***785.28***
Agricultural soils	Direct soil emissions	Synthetic fertilizers	N_2O	***1 522.70***
		Animal waste-applied soils	N_2O	***1 070.50***
		Crop residues	N_2O	913.28
		Organic soil	N_2O	721.10
	Indirect emissions	Atmospheric deposition	N_2O	***1 281.25***
		N Leaching and runoff	N_2O	***1 700.95***
		SUM		12 752.44

Source: MOE (2008).

The OECD Expert Meeting on "Soil Organic Carbon and Agriculture: Developing Indicators for Policy Analysis" recommended that OECD countries could develop the indicator: *Change in total organic carbon in agricultural land over time* (OECD, 2003). Regarding the survey on paddy and upland crop fields, precise and holistic soil surveys have been conducted in Japan since 1979.[8]

Adopting no-tillage, which is strongly recommended in the USA, according to the discussion in MAFF (2008b; 2008c), is unlikely to be a promising technique to suppress carbon release from arable soils, because of the high-humidity and high-temperature climate of Japan (vigorous weed growth is a serious bottleneck). The appropriate amount of organic input seems to be feasible to increase soil carbon content and stimulate reductions in total GHG emissions, given Japanese weather conditions.

Organic fertilizer applications help maintain soil fertility, and also cause environmental trade-off, which increase CH_4 emissions and soil carbon sequestration. Studies on comprehensive carbon dynamics are limited in paddy fields. However, Nishimura *et al*. (2008) studied the effects of land-use change from paddy rice cultivation to upland crop cultivation in the Soil Carbon Budget (SCB), which is estimated by integrating the amounts of net carbon supply and removal of CO_2 and CH_4 and drainage of paddy fields for upland crop cultivation causing significant carbon loss from the soil.

Since the amount of fertilizer application is the control variable in the profit function (Nitrogen response function), it is possible to incorporate the CH_4, N_2O and CO_2 emissions (sequestration) into the economic optimisation model. Then, the net GHG emission (CO_2 equivalent) is explained by the following equation.[9] The details of each gas emission formula, which are based on IPCC guidelines and field surveys in Japan, are explained in Annex C.

$$GHG(CO_2eq) = 21 \bullet CH_4 + 310 \bullet N_2O + CO_2. \tag{17}$$

Social welfare function

The monetary valuation of environmental effects is used to aggregate each environmental effect and then combined with the profit function. These valuation estimates are based on published valuation studies. Firstly, the monetary valuation of N runoff and purification (per kg) is considered. The stated preference method (contingent valuation method and choice experiment) is difficult to apply directly, because of the unfamiliarity of nitrogen runoff and purification, which might induce inappropriate valuation (Hanley *et al.,* 1997). Additionally, there is no precise calculation on this in Japan.

On the other hand, a couple of estimations were conducted using the replacement cost method. It is difficult to apply to water purification in cultivated land where the amounts of purification vary depending on natural conditions and farming practices. Shiratani *et al*. (2004; 2008) overcame this difficulty based on a newly developed method which replaces the N removal rate of paddy fields and the N runoff rate of upland fields by the sum of the maintenance and depreciation costs (in place of construction cost) of water quality improvement facilities. These facilities have same characteristics as paddy fields, which is that the amount of removal N increases in proportion to the N concentration. The related cost of water quality improvement facilities does not change even though the cost of removal is huge. Consequently, the cost per kg of removal becomes cheaper in proportion to the volume removed. Shiratani *et al*. (2004; 2008) estimated the monetary value of N purification (paddy) as 0.3 JPY/m^2/day based on the relationship between N removal rate (g/m^2/day) and N concentration of irrigation water (mg/L) and also N runoff (upland) as -0.08 JPY/m^2/day based on the relationship between N effusing rate (g/m^2/day) and applied N (g/m^2/day), respectively. Each value should be converted into per-parcel (10 ares) and per-year terms for SAPIM simulations, and after this operation 42 000 JPY/10 ares/y[10] and -29 200 JPY/10 ares/y were obtained. The difference between benefit and damage values comes from the specificity of replacement cost method.

Regarding paddy, N purification is effective only when the nitrogen concentration of irrigation water is above 2.5mg/l, and this kind of paddy field accounts for 10% of total paddy fields (Shiratani *et al*., 2004). In addition, the characteristics of SAPIM analysis, which capture the cause-effect linkages in a representative farm level model and extrapolate them to provide insights at more spatially aggregate levels, should be taken into consideration. In short, if the monetary value estimated above is used as it is, overestimation will occur to investigate the social welfare in the aggregated level. Consequently, we choose to abate the monetary valuation of N purification as 1/10 (10% of total paddy fields), with the result that 4 200 JPY/10 ares/y is obtained. This operation is follow as Shiratani *et al*., which tried to estimate the external economic value of Japan-wide cultivated land.

Concerning upland crops, the estimated runoff function is derived from surveying upland-catchment basins. But land use linkages will strongly affect the runoff amount from upland to rivers. According to the N outflow model developed by Tabuchi (1998a) and Tabuchi (1998b), approximately 65-75% of N is removed in the process of water flowing naturally from upland to paddy field (lowland) by denitrification under anaerobic condition and uptake by rice plants. In this analysis, the average figure 70% was quoted form the observations in Tabuchi (1998a, 1998b), and the monetary valuation obtained is -8 760 JPY/10 ares/y.

Since the average amounts of net purification and runoff are set at 0.64 kg/10 ares/y (under average N application: 8.9 kg/10ares/y [Nishio, 2001]) and 4.94 kg/10ares/y (under the average N application: 13 kg/10 ares/y estimated by the National Agriculture Research Centre), each monetary value (per kg) is given by 6 563 JPY/kg and 674 JPY/kg. Per-kg value of purification is much higher than runoff damage, because, again, it comes from the characteristics of replaced water quality improvement facilities and the relatively small purification amount compared with runoff volume.

In the next step, GHG valuations are considered. One of the choices amongst the various studies is to use the price of emission allowances as a proxy (CO_2-eq emissions). In emissions trading theory, the marginal abatement cost equals the allowance price. However, there are difficulties: The EU Emissions Trading Scheme (EU ETS) is the world's first large-scale GHG trading programme, covering around 12 000 installations in 25 countries and six major industrial sectors, but the marginal abatement cost of GHG in Japan is much higher than the EU average (Table 7.4).[11] In addition, Japan's ETS has been launched only recently, hence not enough data are available at present.

The social cost of carbon (SCC) is estimated as the economic value of the extra (or marginal) impact caused by the emission of one more tonnes of carbon (in the form of carbon dioxide) at any point in time. It can be interpreted as the marginal benefit of reducing carbon emissions by one tonne (Yohe *et al*., 2007). They conclude that the average cost is USD 12 in the *Contribution of Working Group II to the Fourth Assessment Report of the IPCC*. But this figure is also inappropriate to apply to Japan, because it is the overall average value in the world.

Table 7.4. Marginal abatement costs (in 1990 USD/tC; 2010 Kyoto target)

Model	**No trading**				**Annex I trading**	**Global trading**
	US	OECD-EU	***Japan***	CANZ		
ABARE-GTEM	322	665	***645***	425	106	23
AIM	153	198	***234***	147	65	38
CETA	168				46	26
Fund					14	10
G-Cubed	76	227	***97***	157	53	20
GRAPE		204	***304***		70	44
MERGE3	264	218	***500***	250	135	86
MIT-EPPA	193	276	***501***	247	76	
MS-MRT	236	179	***402***	213	77	27
Oxford	410	966	***1074***		224	123
RICE	132	159	***251***	145	62	18
SGM	188	407	***357***	201	84	22
WorldScan	85	20	***122***	46	20	5
Administration	154				43	18
EIA	251				110	57
POLES	135.8	135.3	***194.6***	131.4	52.9	18.4

Sources: Cited in Weyant, 1999; Council of Economic Advisors (1998); EIA (Energy Information Administration) (1998); Criqui *et al.* (1999).

Baker *et al*. (2007) (originally, in Viguier *et al.*, 2003) provide a comparison of four model estimates of the costs of meeting Kyoto targets with domestic emission trading and without international trading (Table 7.5). As for Japan, two of the four results are available, with the domestic carbon price estimated as 59.8 (2 000 USD/tCO_2) by EPPA model and 70.8 (2 000 USD/tCO_2) by POLES model, respectively (average, **65.3 USD/tCO_2**).

Table 7.5. A comparison of estimates of domestic carbon price

	Domestic carbon prices (2000 USD/tCO_2)			
Model	EPPA	GTEM	POLES	PRIMES
EU	47.3	46.1	55.9	40.1
US	68.1		52.6	
Japan	***59.8***		***70.8***	

Source: Barker *et al.* (2007).

Taking into consideration the pros and cons of each method (Table 7.6), it is feasible to use a modelling estimation of the domestic carbon price to aggregate the environmental effects in SAPIM. By using the average exchange rate in year 2000 – 1 USD equals 107.8 JPY – 7 039 JPY/Ct is obtained.

Table 7.6. A comparison of GHG monetary evaluation methods

Schemes	Merits	Demerits	Prices	Sources
Price of allowance in EU-ETS	Actual trading data with significant trading volumes	Differences in marginal abatement cost between EU and Japan	10-20 EUR	European Climate Exchange, ECX
Price of allowance in Japan-ETS (voluntary participation)	Actual trading data	Thin market	800-1 250 JPY	Government of Japan
Social cost of carbon	Several peer-review studies	MAC in Japan is higher than average	12 USD (43 USD/Ct)	Yohe *et al.* (2007)
Modelling estimation of domestic carbon price	**Domestic situation is reflected**	**Depends on the model (wide range of estimation value)**	**65.3 USD (average)**	**IPCC (2007)**

Source: Author.

Using the above monetary value estimation, the social welfare function can be expressed as

$$SW = \int_0^1 \sum \pi^i + \alpha z_1 + \beta z_2 + \gamma GHG \quad for\ i = 1, 2$$

$$where \quad \alpha = \begin{cases} -674 & if\ z_1 > 0 \\ -6563 & if\ z_1 < 0 \end{cases} \qquad (18)$$

$$\beta = -674$$

$$\gamma = 7039$$

where π^i refer to farmer's profit function, z to the amount of nitrogen runoff (purification), GHG to the total emission of global warming gases.

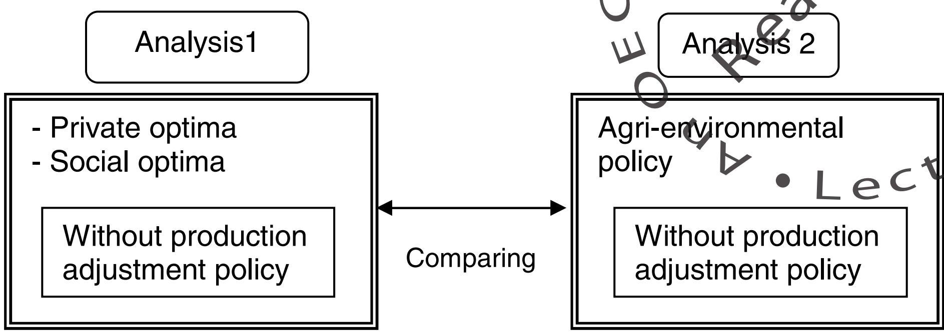
Policy simulations and results

The model estimated the government budget outlays and social welfare as well as crop production, nitrogen runoff and GHG emission under the various scenarios. There are two phases of the analysis. The first analysis compares the private and social optima.

- Private optimum: Producers maximise their profit *ignoring* both positive and negative externalities.
- Social optimum: Government planners maximise producers and society's profit through also *incorporating* both positive and negative externalities.

The second analysis compares different policy options for reducing nitrogen runoff and GHG emissions (Figure 7.6). The policy scenarios assume private profit maximisation by producers.

Figure 7.6. Analysis framework



Source: Author.

As prior conditions, three direct payments (national average) for core farmers, which were introduced in 2007 as part of the Farm Management Stabilisation Programme (MAFF, 2008a), were incorporated into the model:

- Payment based on historical area planted, aiming to correct for disadvantages in domestic agriculture caused by geographical handicaps as compared to other countries.
- Payment based on the commodity output, aiming to encourage quality improvement of domestic products by differentiating the payment rate according to the product quality.
- Payment to compensate for 90% of the loss of income compared with the average income of the preceding five years (excluding the extremes) to mitigate income instabilities caused by price fluctuations. Rice is included only in this payment.

Concerning the effect of the rice production adjustment policy, in which a rice production quota was allocated to each region based on the sales record of two preceding years, although the farm level analysis such as SAPIM could not reflect the effect of the quota (which prevents the rice price decrease *via* production control to meet decreasing demand), in this analysis the "without production adjustment" case was also calculated by assuming that the price decrease resulting from the relaxation of quota is 4.66 %, which has been estimated by OECD (2009).

Analysis 1: Private and social optimum

Firstly, the private and social optima are estimated as the benchmark. The private optimum reflects the rice price under the production adjustment policy (quota) and the subsidy. It means the profit maximisation of the farmer as it is at present. Consequently, the rice paddy area under the private optimum could be assumed as the amount of production quota for rice paddy. The results are compared to the private and social optimum without any corrective policies. The comparison between the private and social optima, that is to say, the impact of market failure on land use, is analysed in this model.

Estimated land allocation and fertilizer application per 10 ares (Table 7.7), total production and total fertilizer use (Table 7.8), N runoff and GHG emission (Table 7.9), and profit and social welfare (Table 7.10) are recorded in each table.

Table 7.7. Land allocation and fertilizer application

Policy	Land use		Fertilizer use per 10a			
			Rice		wheat	
	Rice	Wheat	Chemical (kg)	Organic (t)	Chemical (kg)	Organic (t)
Private optimum:under the production adj.	41	19	9.42-9.92	0.23-0.39	15.31-15.61	0.67-0.77
Social optimum:under the production adj.	41	19	6.69-6.76	0.77-0.86	13.24-13.29	0.92-0.99
Private optimum:without the production adj.	41	19	9.62-10.15	0.17-0.34	15.31-15.62	0.67-0.77
Social optimum: without the production adj.	60	0	6.89-6.98	0.69-0.83	-	-
Policy1: Chemical N -50% + Payment: under the production adj.	40	20	4.71-4.96	0.91-1.11	7.65-7.80	1.42-1.54
Policy1: Chemical N -50% + Payment: without the production adj.	52	8	4.74-5.07	0.86-1.13	7.71-7.65	1.42-1.53
Policy2-1: Chemical N tax (50%): under the production adj.	41	19	9.03-9.44	0.30-0.45	14.92-15.12	0.73-0.81
Policy2-1: Chemical N tax (50%): without the production adj.	43	17	9.22-9.64	0.24-0.40	15.13-15.72	0.38-0.72
Policy2-2: Chemical N tax (300%): under the production adj.	41	19	7.07-7.22	0.62-0.73	12.87-12.98	0.93-1.00
Policy2-2: Chemical N tax (300%): without the production adj.	52	8	7.16-7.31	0.58-0.72	12.95-12.98	0.97-1.00
Policy3: Minimum Organic + Payment: under the production adj.	41	19	6.71-6.86	1.00	12.15-12.25	1.50
Policy3: Minimum Organic + Payment: without the production adj.	46	14	6.69-6.86	1.00	12.18-12.25	1.50
Policy4: Organic payment (quantity payment): under the production adj.	41	19	4.27-4.47	1.57-1.58	13.08-13.09	1.27-1.30
Policy4: Organic payment (quantity payment): without the production adj.	49	11	4.22-4.48	1.57-1.59	13.09	1.29-1.30

Source: Author's calculations.

Private optimum

- Farmer's profit: The farmer maximises yield by relatively large amounts of chemical fertilizer application under profit maximisation behaviour. High-cost organic fertilizers which consist of input price, transportation and spreading cost are not used so much (0.23-0.39 t/10 ares in rice, 0.67- 0.77 t/10 ares in wheat). In private optimum, farmer's profit is the highest among all scenarios.
- Land use: 41 parcels are allocated to rice paddy and 19 parcels are wheat. No land abandonment. The reasons why rice was not cultivated on all of the parcels are the relatively high cost of rice production and the character of rice production, which has a small nitrogen response even in the high land quality field (see Figure 7.3). Since this analytical framework assumed the relatively large farm in the flat area, deficit (land abandonment) did not occur. Regarding this point, the primary reason of abandonment of cultivation is structural, such as lack of succession. In fact, the current land use of rice and upland crop is approximately 1.6 million ha (production quota) and 0.79 million ha, respectively. The simulation results could thus well express the reality of current land use.
- Environmental externalities: Nitrogen runoff via chemical fertilizer application. With regard to GHG, the impact of CH_4 emission and carbon sequestration *via* organic fertilizer application is small.

Table 7.8. Total production and total fertilizer use

Policy	Total production (kg)		Fertilizer use (total)					
			Rice		Wheat		Total	Total
	Rice	Wheat	Chemical	Organic	Chemical	Organic	Chemical	Organic
Private optimum: under the production adj.	22242	12555	396	13	294	14	690	27
Social optimum: under the producion adj.	22719	12671	276	34	252	18	528	52
Private optimum: without the production adj.	22164	12555	405	11	294	14	699	24
Social optimum: without the producion adj.	34248	0	412	48	0	0	418	46
Policy1: Chemical N -50% + Payment: under the production adj.	22055	13463	193	41	155	29	348	70
Policy1: Chemical N -50% + Payment: without production adj.	29250	5384	255	52	61	12	317	65
Policy2-1: Chemical N tax (50%): under the production adj.	23488	11340	397	16	255	13	653	29
Policy2-1: Chemical N tax (50%): without production adj.	23415	11340	406	14	255	13	661	27
Policy2-2: Chemical N tax (300%): under the production adj.	22570	12652	294	28	246	18	540	46
Policy2-2: Chemical N tax (300%): without production adj.	29135	5537	379	34	104	8	482	42
Policy3: Minimum Organic + Payment: under the production adj.	22889	12901	279	41	232	29	511	70
Policy3: Minimum Organic + Payment: without production adj.	25901	9669	312	46	171	21	483	67
Policy4: Organic payment (quantity payment): under the production adj.	23003	12877	180	65	249	24	429	89
Policy4: Organic payment (quantity payment): without production adj.	27868	7661	214	77	144	14	358	92

Source: Author's calculations.

Social optimum

- Farmer's profit: Social planners maximise social welfare. Consequently, the farmer's profit is lower than those of the private optimum.
- Land use: Maximum social welfare was obtained within the rice production adjustment policy (41 parcels are restricted as maximum allocation for rice cultivation).
- Environmental externalities: Substitutions from chemical fertilizer to organic fertilizer has occurred partially. The amount of nitrogen runoff decreased to approximately half volume. Regarding GHG emission, organic applications (0.77-0.86 t/10 ares in rice paddy and 0.92-0.99 t/10 ares in wheat) promote the carbon sequestration, and result in a significant decrease of net GHG compared with the emissions under the private optimum.

Without production adjustment policy

In the next simulation, private and social optima are estimated *without* the rice production adjustment policy (quota). It is assumed that the rice price drops by 4.66% and the diversion payment for wheat production[12] is kept. A direct payment to cover the income decrease due to the rice price drop is set as 6 000 JPY/10 ares (decoupling payment) to meet with same social welfare outcome as under the production adjustment scenario. The gross amount of the decoupled payment is not included in the social welfare calculation, because it is assumed that payments are allocated from the existing agricultural budget.

Table 7.9. N runoff and GHG emission

Policy	N runoff * (kg)	GHG emission and sequestration * (t)			
		CH_4	N_2O	CO_2	Total CO_2 eq.
Private optimum: under the production adj.	101.0	11.4	3.3	-3.4	11.3
Social optimum: under the production adj.	56.4	11.9	2.8	-8.5	6.2
Private optimum: without the production adj.	103.4	11.4	3.0	-3.0	11.4
Social optimum: without the production adj.	-58.9	17.3	1.3	-8.0	10.7
Policy1: Chemical N -50% + Payment: under the production adj.	11.3	11.8	2.3	-9.0	5.1
Policy1: Chemical N -50% + Payment: without production adj.	-34.4	15.3	1.7	-9.7	7.3
Policy2-1: Chemical N tax (50%): under the production adj.	76.9	12.1	2.8	-4.0	10.9
Policy2-1: Chemical N tax (50%): without production adj.	78.8	12.0	2.8	-3.6	11.2
Policy2-2: Chemical N tax (300%): under the production adj.	54.1	11.8	2.7	-6.3	8.2
Policy2-2: Chemical N tax (300%): without production adj.	-10.1	14.9	1.9	-6.6	10.2
Policy3: Minimum Organic + Payment: under the production adj.	45.5	12.1	2.9	-9.0	6.0
Policy3: Minimum Organic + Payment: without production adj.	17.9	13.5	2.5	-9.3	6.8
Policy4: Organic payment(quantity Payment): under the production adj.	50.7	12.5	3.0	-11.8	3.7
Policy4: Organic payment(quantity Payment): without production adj.	1.7	15.0	2.4	-13.0	4.4

* The minus represents the purification for nitrogen and the sequestration for carbon.

Source: Author's calculations.

Private optimum

- Similar results as under the production adjustment scenario are obtained as a corollary to setting the amount of the decoupled payment.

Social optimum

- Farmer's profit: The social planner maximises social welfare. Consequently, farmer's profit is lower than those of the private optimum.
- Land use: Social welfare was maximised by expanding the rice cultivation area, which has a bigger positive externality. Rice paddy was allocated to all parcels.
- Environmental externalities: Substituting organic fertilizer for chemical fertilizer has partially occurred. CH_4 emissions were increased due to the expansion of rice paddy land use, while nitrogen purification is fully achieved. Note that the increase amount of carbon sequestration is bigger than CH_4 emissions.

Table 7.10. Profit and social welfare

Policy	Profit	Profit +payment (tax)	Budget outlays	Runoff damage (Purification benefit)	GHG emission damage (Sequestration benefit)	Social welfare (1000JPY)	SW/SO
Private optimum:under the production adj.	1854	1854	-	73	-79	1847	0.94
Social optimum: under production adj.	1802	1802	-	203	-43	1962	1.00
Private optimum:without the production adj.	1873	1873	(246)	57	-80	1850	0.89
Social optimum: without production adj.	1765	1765	(360)	387	-75	2077	1.00
Policy1: Chemical N -50% + Payment: under the production adj.	1703	1871	168	247	-36	1914	0.98
Policy1: Chemical N -50% + Payment: without production adj.	1663	1827	164	354	-51	1966	0.95
Policy2-1: Chemical N tax (50%): under the production adj.	1853	1792	-61	121	-76	1898	0.97
Policy2-1: Chemical N tax (50%): without production adj.	1871	1811	-61	109	-79	1902	0.92
Policy2-2: Chemical N tax (300%): under the production adj.	1813	1515	-298	197	-58	1951	0.99
Policy2-2: Chemical N tax (300%): without production adj.	1814	1548	-266	300	-72	2041	0.98
Policy3: Minimum Organic + Payment: under the production adj.	1738	2218	480	209	-42	1906	0.97
Policy3: Minimum Organic + Payment: without production adj.	1755	2235	480	257	-48	1965	0.95
Policy4: Organic payment (quantity payment): under the production adj.	1619	2266	647	230	-26	1822	0.93
Policy4: Organic payment (quantity payment): without production adj.	1592	2287	695	314	-31	1876	0.90

Note: SW/SO represents the social welfare ratio of the scenario relative to the social optimum, and the denominators (SO) which are used in the each case (under or wothout production adj.) are those of social optimum, relatively.

Source: Author's calculations.

The comparison between private and social optimum, that is to say, the impact of market failure on land use is analysed in this model. As shown in Figure 7.7, results revealed that under the private optimum, rice is cultivated in 41 parcels and wheat is in 19 parcels, fertilizer use is higher than under the social optimum. Under the social optimum, more land is allocated to paddy fields due to their positive (*i.e.* reduced negative) environmental externalities only if the rice production adjustment policy (quota) is relaxed.

Analysis 2: The impact of agri-environmental policy

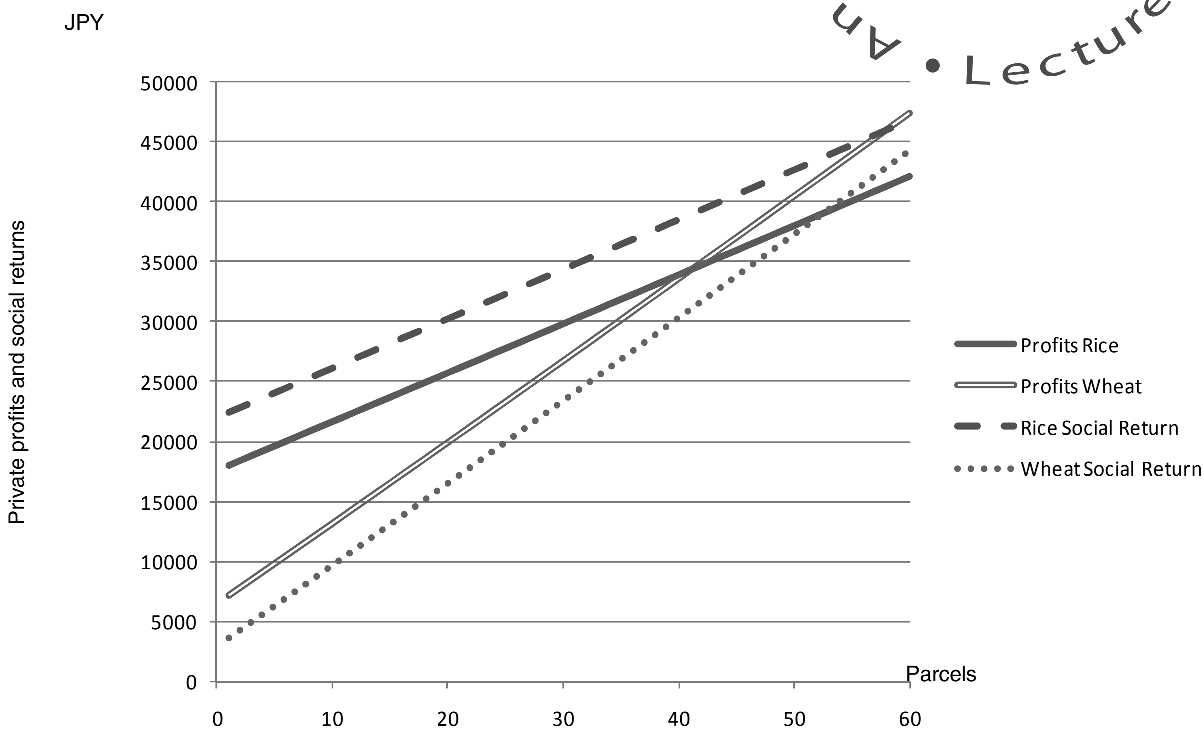
In Analysis 2, several agri-environmental policy simulations are conducted on the basis of results in Analysis 1. The policy scenarios assume private profit maximisation by producers and a comparison of different policy options for reducing nitrogen runoff and GHG emissions is undertaken. Policy scenarios are as follows:

- Policy 1: Imposing 50% reduction of chemical fertilizer use plus an environmental area payment, which compensate for the amount of profit lost compared with the private optimum.
- Policy 2-1: Tax on price of chemical fertilizer (tax rate is fixed arbitrarily at 50%).
- Policy 2-2: Tax on price of chemical fertilizer (tax rate is fixed arbitrarily at 300%).
- Policy 3: Imposing a minimum organic fertilizer (manure) application (1 t/10 ares for rice paddy and 1.5 t/10 ares for wheat) plus an environmental area payment

(8 000 JPY/10 ares for rice paddy and wheat), which compensates for the additional cost of organic matter use.

- Policy 4: Environmental payment based on the amount of organic fertilizer (manure) applied (8 000 JPY/1 t for rice paddy and 8 000 JPY/1.5 t for wheat), which compensates for the additional cost of organic matter use.

Figure 7.7. Private profits and social returns without production adjustment



Source: Author's calculations.

Amongst the policy scenarios, it could be said that policy 1 and policy 2 are targeted to control nutrient runoff and policy 3 and policy 4 are to enhance soil carbon sequestration. Note that policies aiming to decrease nutrient runoff through chemical fertilizer application control also affect net GHG emission simultaneously through the substitution from chemical to organic fertilizer as the result of farmer's profit maximisation behaviour. In addition, note that the policy mix with policy 1 (or policy 2) and policy 3 (or policy 4) have not been undertaken in this study.

Results

- Social welfare in policies 1-4 resembles the social optimum. Since social welfare is optimised under the rice production quota, there are no noticeable differences compared with the social optimum.

- Regarding the chemical fertilizer tax, as has already been analysed (*e.g.* Opschoor *et al.,* 1994), a low rate of environmental tax (policy 2-1) might not be enough incentive to change the farmer's behaviour and the impact for the farmer's profit is small. Quite high tax rates (policy 2-2) are required to achieve same level of environmental effect as other environmental policies, but this policy results dramatically decreases the farmer's profits. Consequently, in order to control the nitrogen runoff, reducing chemical fertilizer by promoting substitution of organic fertilizer use is effective. Agri-environmental payments subject to chemical nitrogen application standard are more effective than the unit tax or unit payment.

- In general, the increase in the amount of soil carbon sequestration is above the increase of CH_4 emission. Thus, net GHG emissions significantly decrease comparing with emission under the private optimum.
- Agri-environmental payments based on the amount of organic application (policy 4) result in significant amounts of carbon sequestration, although it increases the fiscal budget burden.

Without production adjustment policy

In the next step, the agri-environmental policy impact was considered *without* the production adjustment policy. In this case, it is assumed that the rice price drops by 4.66% and the diversion payment is kept. A direct payment to cover the income decrease due to the rice price drop is set as 6 000 JPY (decoupling payment), as in Analysis 1. The gross amount of the decoupled payment is not included in the social welfare calculation, because it is assumed that payments are allocated from the existing agricultural budget.

Results

- In the every policy scenario, higher social welfare is obtained than the under the production adjustment simulation.
- High efficiency was achieved though agri-environmental payments (policy 1 and policy 3), but care should be taken when interpreting the results because transfer efficiency and transaction costs are not considered.

Summary of the Japanese case study

In this case study a social benefit function is proposed and some environmental effects of agricultural production are modelised and then monetarised. However it should be kept in mind that these both steps entail important difficulties and uncertainties.

The results obtained with the SAPIM application to Japanese agriculture indicates that different agri-environmental policy instruments lead to very different outcomes in terms of land-use, production, and environmental externalities. A special feature of this case study is the integration of rice production with upland field crop production in the same analytical framework. In general, rice paddies could provide positive or negative environmental effects in this analysis, depending on farm management practices. Consequently, the incentives provided to farmers that encourage environmentally friendly production practices have a significant impact on the environmental effects of rice paddies.

Some caveats should be noted. First, it is not necessarily the case that the results would apply to every rural area, because national average data or available scientific data were used in this modelling exercise. For example, if the intermediate and mountainous areas are analysed then data for the profit function and externalities should be adapted accordingly. Second, it is assumed in this analysis that farmers change their land use within the multi-purpose paddy field to deal with the continuous land use of rice and upland crops. Finally, scientific data was limited to estimate robust relationship in the empirical model (Annex C) and also the following variables are not considered: CO_2 emissions from agricultural land, transaction costs, and transfer efficiency.

The main findings of Japanese case study are summarised as follows:

- In every scenario, more parcels are allocated to rice paddy than to wheat. Quantitatively, rice production predominated over wheat production in raising the farmer's profit and in improving environmental performance (externalities).
- In terms of social optimisation, social welfare is maximised when every parcel is allocated to rice production. In this case, the farmer's profit is lower than those of the private optimum.
- Agri-environmental policy could compensate for the reduction of social welfare of the private optimum via reducing negative and increasing positive externalities. But even in this case, not every parcel is allocated to rice production, due to the small nitrogen response of rice and the high production cost.
- In order to control nitrogen runoff, reducing chemical fertilizer through promoting substitution by organic fertilizer use leads to environmental improvements. Agri-environmental payments subject to a chemical nitrogen application constraint is more effective than a nitrogen tax.
- With regard to carbon sequestration, an agri-environmental payment depending on the level of application of organic matter (manure) is preferable. With regard to social welfare, a payment subject to the application of a minimum organic matter, which can avoid the increase of the fiscal budget burden (caused by the increase application of organic matter on paddy fields), has a higher social welfare than a unit payment depending on the level of application of organic matter.
- Without a production quota policy scenario the results show a positive impact for nitrogen runoff reduction, carbon sequestration and social welfare.

Notes

1. As the basis for this modelling exercise, the key agricultural policies in place as of December 2009 were assumed (*e.g.* direct payments for core farmers; also, see OECD, 2009). However, it has since been decided, under the new *Basic Plan for Food, Agriculture and Rural Areas* (Cabinet Council decision of March 2010), that a revised programme of income support (direct payments) for all farms will be established, and that payment schemes previously targeted only to core farmers will in future be discontinued. Prior to the formal implementation of this policy reform, in April 2010 pilot programmes were set up which made direct payments to rice-producing farmers conditional on their participation in rice-production adjustment schemes. Despite this recent reform, however, the key results of the present study continue to hold, given that its focus is a *relative* comparison of the social welfare outcomes brought about by the implementation of various agri-environmental policies, and not an *absolute* comparison.

2. For example, *The Biomass Nippon Strategy* (2002; 2006), which establishes a set of programmes aimed at recycling more than 80% of waste biomass and utilisation of more than 25% of unused biomass, the report on the *Remarkable Increase in Production of Domestic Biofuels* (2007) shows that an increase in biofuel production (6 million kilo-litres, MAFF estimation) is feasible by around 2030 if appropriate technical developments are achieved. The *Law Promoting Domestic Biofuel Production* (2008) aims to accelerate the use of non-food materials such as rice straw and wooden biomass for bioethanol production.

3 Although direct payments to farmers in hilly and mountainous areas aim to prevent abandonment of farming and maintain a range of ecosystem services, these payments are not included in this study.

4. Compared to the amount of nitrogen runoff, phosphorus runoff is quite small (Tabuchi and Takamura, 1985).

5. Methane, an important GHG, is generated from paddy field rice cultivation.

6. Japan is the host of the 10th Conference of the Parties to the Convention on Biological Diversity (COP 10) scheduled for October 2010, at Aichi-Nagoya.

7. Emissions of IPCC category 4C (rice cultivation) and 4D (agricultural soils) occupied only1% of total emissions.

8. The Korean government has also monitored the range and role of soil organic carbon in paddy and upland crop fields (OECD, 2003).

9. CO_2 emissions derived by agricultural machinery use are not included due to the lack of data.

10. Paddy cultivation period is assumed as140 days.

11. In the IPCC third assessment report, marginal abatement costs are compared for the United States, Japan, OECD-Europe, and the rest of the OECD (CANZ), calculated using 13 world models. "*Despite the wide discrepancies in results across models, the robust information is that, in most models, marginal abatement costs appear to be higher in Japan than in the OECD-Europe.*"

    (www.grida.no/publications/other/ipcc_tar/?src=/climate/ipcc_tar/wg3/341.htm)

12. MAFF provides subsidies to the producers' organisations to provide an economic incentive to participate in the production adjustment programme. The average amount is approximately 35 000 JPY/10 ares.

Chapter 8
Sensitivity analysis

Policy analysis, by testing the impact of different possible states of the world, provides insight into the impacts of different government instruments. These have been discussed in the various SAPIM case studies, and in the comparative analysis chapter. This chapter considers how the validity of these results can be assessed through sensitivity analysis.

Sensitivity analysis is important in the development of recommendations for policy makers. It permits an assessment of whether the analysis is credible, robust to different assumptions, etc. In principle, sensitivity analysis is a simple idea: change an assumption or a parameter in the model and observe how this affects behaviour. The more complex the model (*i.e.* the higher the number of assumptions and parameters), the more difficult it is to systematically test all possible combinations. Sensitivity analysis can be categorised into two categories: 1) sensitivity to parameter values, and 2) sensitivity to key assumptions.

Sensitivity analysis of model parameters

Pannell (1997) proposes various systematic strategies to assess uncertainty in model parameters. The most comprehensive strategy (*i.e.* testing all states of the world) is very resource intensive. On the other hand, a judiciously chosen simple strategy can still be reasonably systematic and yet provide the key insights that would have been obtained under the comprehensive strategy. Pannell describes this "simple" strategy as follows:

- Select parameters to be varied. Identify a range for each parameter which realistically reflects its possible range;
- Conduct sensitivity analyses for each parameter individually, using two parameter values (max. and min.);
- Identify the key parameter by sensitivity index

$$SI = (D_{\max} - D_{\min})$$

  where $D_{\max}$ is the result for the maximum parameter value and $D_{\min}$ is the result for the minimum parameter value.

- Summarise results. For each key decision variable, calculate the values of sensitivity index (SI) for all parameters and discrete scenario, and rank them by absolute value. These results can be reported directly or used to select which parameters will be examined in graphs and tables (*e.g.* spider diagrams). This permits the decision maker to focus on important parameters and relationships;

- Draw conclusions: (a) do X, (b) do either X or Y depending on the circumstances, (c) do either X or Y, depending on preferences, and (d) if in doubt, do X (safe-bet strategy).

Sensitivity analysis with respect to key assumptions

The SAPIM case studies are all based on a set of key assumptions. The models are partial-equilibrium, and therefore require assumptions about output prices and input prices – *i.e.* these are exogenous to the model. Values for these assumptions are specific to each case study, and therefore depend on the corresponding policy environment in each country. For example, a key assumption in two case studies (Japan and Switzerland) is government intervention in output prices such that they are above prices observed in the international market. In the Swiss case study, the "sensitivity" of this assumption is already analysed in the scenario in which the market regulation (dairy quotas) are eliminated. Similarly, a common scenario in all case studies has been the application of a tax on fertilizer. This is equivalent to testing the sensitivity to changing fertilizer price.

Therefore, there is some overlap between the definition of sensitivity analysis with respect to key assumptions and the policy simulations already conducted. Nevertheless, it is still useful to systematically assess the sensitivity of all case study models to a consistently defined set of changes in key exogenous assumptions.

Sensitivity analysis of SAPIM case studies

In summary, it is not necessary to test all combinations, as a well chosen subset comprising the critical assumptions and parameters will permit an assessment of what are the most likely sources of uncertainty in a model. In the context of this study, which ranks policy instruments according to different criteria (*e.g.* cost-effectiveness, environmental effectiveness, etc.), an important question is under what circumstances would the ranking of policies be changed?

The sensitivity analysis for the SAPIM case studies is therefore conducted as follows:

- Change by 10% and 30% output and input prices.
- Change by 25% the value of parameters that determine crop yield response to fertilizer.
- Compare the relative ranking of policies with respect to cost-effectiveness and environmental effectiveness.

Output and input price shocks

Table 8.1 reports the results for a 10% and 30% shock in output and input (fertilizer) prices respectively for each of the country case studies. The baseline situation is presented in absolute quantities, while the scenario results are presented as a percent change from the base. In the Finnish and Japanese case studies, the private optimum is reported here, while the social optimum is reported in Tables 8.2 and 8.3.

The output price shock is defined in each case study as a 10% and 30% change in the prices of agricultural commodities produced on that farm. Profit changes are in reasonable ranges for each case study – a 10% (30%) shock in output prices generates more than a 10% (30%) change in profits simply because costs have not been changed. There is some variation in the size of the profit change amongst the case studies, however comparisons across case studies is not meaningful because of the very different production systems studied (*e.g.* dairy in Switzerland *vs* corn in the US), and because productions costs have not been modelled in the same way in all studies (*e.g.* all farm costs included in the Swiss case while, in the US, labour and capital costs are not included).

Output price shocks also generate land-use change. In the Finnish and Swiss case studies, land-use changes were in some cases substantially different than the initial mix in the base. Differences in land-use change have an impact on nitrogen runoff. Therefore *a priori* it is not possible to predict whether an increase in output will lead to an increase in nitrogen runoff. In the US case study, the principle crop (corn) is planted in rotation with soybean and therefore land-use change is by definition constrained. In the Japanese case study, the increase in rice and wheat prices increases the incentive to expand rice paddy production area, because the base price of rice is higher than that of wheat. Indeed, the decrease in nitrogen runoff in the scenario without production adjustment (rice quota) scenario is bigger than the value under the rice production quota scenario where rice cultivation is constrained.

The fertilizer price shock shows consistently across the case studies that nitrogen runoff (or surplus) is relatively insensitive to changes in the price of fertilizer. This is essentially a consequence of the way nitrogen response is represented. It is a well known result that using a Mitscherlich nitrogen response function implies relatively price inelastic fertilizer application. Nevertheless, in the case study countries where alternative competing agricultural crops are available, some shifts in land allocation patterns are observed. In the Finnish case study, the reported change in nitrogen runoff is large in terms of the whole farm (-19% to +20%), but the changes reflect shifting land use into and away from forestry. Per hectare runoff for crop activities is very little affected. In the Swiss case study, no changes in nitrogen runoff occur by assumption, because within a wide range of situations chemical fertilizer is easily substitutable with the nitrogen equivalent in manure produced from the dairy operation.

Table 8.2 reports the results for a 10% shock in output and input (fertilizer) prices as well as 30% change in nitrogen runoff damage estimate in the social optimum of the Finnish case study.

The results show that an output price increase (decrease) of 10% increases (decreases) social welfare 56% (26%) due to increase (decrease) in the profitability of crop production. The corresponding effect of fertilizer price change has smaller impacts on social welfare under social optimum. A fertilizer price increase reduces social welfare since profits are decreased in absolute terms more than nitrogen runoff damage. In the case of nitrogen runoff damage estimate, the increase of that estimate by 30% results in a slight increase in social welfare due to decreased nitrogen runoff and thus runoff damage.

In the Japanese case study, the sensitivity of the results to the valuation of environmental externalities was tested by changing by 10% the monetary value of N runoff (purification) and GHG emission, to line up with other shocks reported earlier (output and fertilizer). In addition a 30% change was also imposed to account for the higher degree of uncertainty related to monetary valuation.

Table 8.1. Sensitivity analysis: 10% and 30% shocks to output and fertilizer prices

				Output price				Fertilizer price			
Country			**Base**	10%	-10%	30%	-30%	10%	-10%	30%	-30%
Finland	Private profit, EUR		2 869	50%	-30%	279%	-33%	-10%	13%	-24%	146%
	N runoff, kg total farm		367	56%	-71%	190%	-100%	-19%	22%	-54%	169%
Switzer-land	Private profit, CHF		82 083	29%	-26%	90%	-77%	-1%	1%	-4%	5%
	N balance, kg/ha		81.2	-19%	2%	-1%	-19%	0%	0%	4%	6%
US*	Private profit, USD/acre		375	21%	-21%	63%	-63%	-4%	4%	-11%	11%
	N runoff, lbs/acre		7.45	1%	-1%	2%	-3%	-1%	1%	-2%	2%
Japan	Private profit, 000 JPY	Production adjustment	1 854	37%	-35%	111%	-86%	-2%	1%	-2%	2%
	N runoff, kg total farm	Production adjustment	101	-7%	115%	-17%	158%	-2%	9%	-6%	14%
	Private profit, 000 JPY	w/o production adjustment	1 873	36%	-34%	110%	-86%	-1%	1%	-2%	2%
	N runoff, kg total farm	w/o production adjustment	103	-63%	91%	-146%	152%	-2%	3%	-6%	14%

*Simple average of eight production systems, mean productivity.
Source: Author's calculations.

Table 8.2. Finnish case study: The effects of output and input prices and nitrogen runoff damage estimate on social welfare (EUR); farmers' profits in social optimum (EUR); total nitrogen runoff (kg) and nitrogen runoff damage (kg)

		% Change from base					
	EUR	Output price		Fertilizer price		Runoff damage	
Social optimum	**Base**	**+10%**	**-10%**	**+10%**	**-10%**	**+30%**	**-30%**
Social welfare	3 019	56	-26	-3	19	2	12
Farmers' profits	2 498	62	-23	-3	22	7	7
Total runoff	170	54	-59	-18	18	-15	17
Runoff damage	607	54	-59	-18	18	-15	17

Source: Author's calculations.

Table 8.3 reports the Japanese results for a 10% and 30% shock in monetary valuation under the social optimum. The results show that the impact of changing monetary valuation is much smaller than the impact for output price changes reported in Table 8.1, even in the 30% scenario.

A key result from Japanese case study continues to hold within the range of this sensitivity analysis. That is to say, the results of *without* rice production quota scenario show a positive impact for nitrogen runoff reduction, carbon sequestration and social welfare, comparing *with* rice production quota scenario.

Table 8.3. Japanese case study: 10% and 30% shock in monetary valuation under the social optimum

			N Evaluation				GHG Evaluation			
		Base	+10%	-10%	+30%	-30%	+10%	-10%	+30%	-30%
Profit, (000 JPY)	Production adjustment	1 802	-0.2%	0.2%	-0.7%	0.7%	-0.1%	0.1%	-0.4%	0.3%
	w/o Production adjustment	1 765	-0.2%	0.2%	-0.6%	0.7%	-0.1%	0.1%	-0.3%	0.2%
Welfare, (000 JPY)	Production adjustment	1 850	7%	5%	9%	3%	6%	6%	5%	7%
	w/o Production adjustment	2 007	5%	2%	9%	-2%	3%	4%	2%	5%
N runoff (total kg)	Production adjustment	56.4	-3%	7%	-12%	23%	-	-	-	-
	w/o Production adjustment	-58.9	-1%	1%	-0.4%	3%	-	-	-	-
GHG emissions, (total CO2)	Production adjustment	6.2	-	-	-	-	-3%	2%	-8%	6%
	w/o Production adjustment	10.7	-	-	-	-	-1.0%	1.0%	-2.0%	2.0%

Source: Author's calculations.

Parameter shocks: Nitrogen response function

Each of the SAPIM case studies includes a module for crop activities that relies on non-linear functions to model crop yield response to nitrogen application. These functions comprise three parameters: an intercept parameter (a) that serves mainly to calibrate the model to the base yield, a slope parameter (α) and a curvature paremeter (β). The slope and curvature parameters together determine crop yield response to nitrogen application. The curvature parameter (β) is negative in order to reflect diminishing returns to nitrogen application.

The Finnish, US and Japanese case studies were modeled using quadratic functions, $y=a+\alpha x-\beta x^2$, while the Swiss cases study employed a Misterlich function, $y=\alpha(1-a)\exp(-\beta x)$, where y is the yield and x is the level of nitrogen application. The coefficients in the Nitrogen response function were shocked by 25%. Three types of shocks were conducted: only α, onlyβ and both αandβ. Table 8.4 reports the results.

Table 8.4. Sensitivity analysis: 25% shocks to parameters in the nitrogen response function

				Nitrogen response					
				α		β		α and β	
Country			**Base**	25%	-25%	25%	-25%	25%	-25%
Finland	Private profit, EUR		2 869	142%	-15%	273%	-29%	307%	-33%
	N runoff, kg total farm		367	134%	-23%	235%	-77%	235%	-100%
Switzer-land	Private profit, CHF		82 083	32%	-23%	3%	-5%	36%	-28%
	N balance, kg/ha		81.2	-98%	112%	-73%	69%	-142%	160%
United States	Private profit, USD/acre		375	154%	-118%	-52%	86%	71%	-71%
	N runoff, lbs/acre		7.45	57%	-35%	-28%	78%	2%	-3%
Japan	Private profit, 000 JPY	Production adjustment	1 854	50%	-41%	-19%	27%	23%	-23%
	N runoff, kg total farm	Production adjustment	101	81%	-34%	-34%	93%	3%	5%
	Private profit, 000 JPY	w/o Production adjustment	1 873	48%	-39%	-18%	25%	22%	-22%
	N runoff, kg total farm	w/o Production adjustment	103	99%	-71%	-52%	78%	17%	-16%

Source: Author's calculations.

Given a negative curvature parameter (β), shocking the slope parameter (α) alone should result in higher output response than shocking both α and β. This is true in the US and Japan case studies, where there is little (Japan) or no land-use change (US). For example, in the Japan case study, the 25% increase in parameter α induces a slight land-use change from rice to wheat simply because the slope of the wheat response curve is steeper than that for rice ($\partial y_{wheat}/\partial x_{wheat} > \partial y_{rice}/\partial x_{rice}$). Simultaneously, the intensification of nitrogen application increases. These integrated effects result in more than a 25% change. The 25% shock in both parameters generate less than a 25% change in profits and N runoff, because the effect of α change is partially offset by the effect of β.

However, in the Finnish and Swiss case studies, land-use changes also contribute importantly to profit and N-runoff. In these cases it is not possible *a priori* to predict whether or not shocking α and β individually will have a greater impact than shocking both α and β simultaneously.

Summary of the sensitivity analysis

The sensitivity analysis presented in Tables 8.1 to 8.4 reflect only a small subset of the range of possible alternatives that could be explored. Nevertheless, they cover important variables, and a range of uncertainly of 10% and 30%. While it is true that results for farmer's profit, nitrogen surplus, nitrogen response etc. are all crucially dependant on the assumptions used in the model, the analysis presented in this report does not critically hinge on the absolute level of the results obtained. The models are carefully calibrated to reproduce real world examples from each case study country. Therefore, *a priori*, the sensitivity analysis presented here (and any other reasonable combination of shocks) should not produce unexpected results. It is also true that some key assumptions drive model outcomes. This is well illustrated in the shocks on fertilizer price, where the functional form used for modelling nitrogen response implies very low price elasticity. This very closely approximates the actual situation in each of the case studies examined, and is therefore not only a crucial assumption of the framework but also reflects reality.

The key source of uncertainty is arguably related to the valuation estimates of social benefits in the case studies for Finland and Japan. While this analysis does not attempt to address the question of what is a reasonable range of uncertainty for the valuation estimates of social benefits, even a 30% shock does not change in a fundamental way the results obtained.

Chapter 9
Comparative analysis of results for all case studies

This chapter provides an overview of the main results obtained across the case studies for Finland, Switzerland, United States and Japan. In the previous chapters, the importance of the policy environment specific to each case study was emphasised. In particular, within each case study, the role of the "policy package" is crucial, as it defines the context, and therefore the assumptions that must be applied in order to have a realistic representation of the impact of agri-environmental policies. For example, the crop yield response to nitrogen application is location specific, as are the assumptions about exogenous prices for outputs and inputs.

The main contribution of this chapter would be to compare and contrast the results across countries, rather than to analyse the policy implications, because the individual context is crucially important in defining the policy mix. Nevertheless, the impact of different types of policies can still be compared – simply the focus will be more on relative effectiveness with respect to environmental goals.

Comparing environmental and economic impacts

Each of the case studies has been chosen to highlight the very different production system and policy context (Table 9.1). The common thread underlying all case studies is the impact of various environmental policies. Specifically, all case studies have an important crop production component, in which the impact of fertilizer application is assessed in terms of crop yield and nutrient runoff. In terms of policy instruments common to all case studies, fertilizer taxes and – with the exception of the Swiss case study – the impact of buffer strips on nutrient runoff can be compared.

Table 9.1. Choice variables, environmental issues and policy instruments covered in different case studies

		Finland	Switzer-land	US	Japan
	Land use	Forest, rape, wheat, buffer	Pasture, silage, barley	Land retirement, no-till, conventional tillage	Paddy, wheat, abandon-ment
Choice variables	Fertilizer	x	x	x	x
	Buffer strip width	x		x	
	Stocking rate		x		
	Manure application		x		x
Environmental issues	Nutrients	x	x	x	x
	Soil erosion			x	
	Ammonia		x		
	Greenhouse gases		x		x
	Biodiversity/wildlife	x			
Agri-environmental policy instruments	Regulatory	x	x	x	
	Indirect tax	x	x	x	x
	Direct payment	x	x	x	x
	Conservation auction	x		x	

Source: Author's classifications.

The environmental policy instruments to be compared:

Environmental policy	**Change in nitrogen application** (kg/ha)	**Change in nutrient runoff** (kg/ha)	**Change in GHG emissions**
Nitrogen tax	All countries	All countries	Japan, Switzerland
Buffer strip	Finland, United States	Finland, United States	

The cost-effectiveness of these policies in terms of profit foregone per unit of environmental impact:

Environmental policy	**Change in profit** (USD/ha)	**Change in profit per kg runoff** (USD/kg)	**Change in profit per kg GHG** (USD/kg)
Nitrogen tax	All countries	All countries	Japan, Switzerland
Buffer strip	Finland, United States	Finland, United States	

Comparative analysis results

Tables 9.2 and 9.3, below, provide summaries for comparative analysis of the results of case studies.

Table 9.2. Comparative analysis of nitrogen tax

			Finland	Switzerland*	US**	Japan***
Environmental impacts	Change in nitrogen application	kg/ha/%	1.10	2.65	0.22 - 0.44	0.13
	Change in nutrient (nitrogen) runoff	kg/ha/%	0.17	0.19	0.02 - 0.13	0.08
	Change in GHG emissions	CO_2t/ha/%		0.00		0.001
Economic efficiency	Change in profit	USD/ha	10.3	114.2	-0.25 - +0.5	2.89
	Change in profit per kg runoff	USD/kg	2.11	1.1	-2.2 - +1.5	0.70
	Change in profit per tonne GHG	USD/CO_2t				111.1

(Shaded cells indicate no data are available for a given case study.)
* Farm-gate nutrient balance, tax on fertilizer price.
** The diversion of each figure relates to each of the eight different combinations of crop/rotation/tillage/erodibility.
*** Without production adjustment programme.

Source: Author's calculations.

As can be seen from Table 9.2, nitrogen tax reduces nitrogen application by 1.1 kg/ha/tax rate in Finland and corresponding figure for Switzerland is 2.65 whereas application reduction per tax rate is more modest in the case of the US and Japan. As regards average abatement cost (change in profit per kg of runoff), the most expensive country is Finland (2.11 USD/kg) and the lowest abatement cost countries are the United States (-2.2 – 1.5USD/kg) and Japan (0.70 USD/kg). These differences mainly reflect the share of total production cost attributable to nitrogen fertilizer. However, changes in nitrogen application for both the US and Japan are quite small, and thus it is obvious that a nitrogen tax might not provide a very strong incentive for farmers to change their behaviour in those countries.

Table 9.3 provides cost estimates for reducing nitrogen runoff by establishing buffer strips. Huge variations for average abatement cost (change in profit per kg runoff) are observed across US cultivation systems.

The lowest average abatement cost in the US (HEL_NLL_Corn/soy: 1.4 and NonHEL_MLD_Corn: 2.9) is similar to the Finnish abatement cost estimate.

Table 9.3. Comparative analysis of buffer strips

			Finland*	Switzerland	US**	Japan
Environmental impacts	Change in nitrogen application	kg/ha/%	-0.03		2.21-8.83	
	Change in nutrient (nitrogen) runoff	kg/ha%	1.37		0.88-17.44	
	Change in GHG emissions	CO_2t/ha/%				
Economic efficiency	Change in profit	USD/ha	11.2		7.5-33	
	Change in profit per kg runoff	USD/kg	2.31		1.4-58.3	
	Change in profit per tonne GHG	USD/CO_2t				

(Shaded cells indicate no data are available for a given case study.)
* Buffer norm.
** The diversion of each figure relates to each of eight different combinations of crop/rotation/tillage/erodibility.

Source: Author's calculations.

Chapter 10
Conclusions

In this report, the conceptual and quantitative linkages between agricultural policies and environmental impacts have been analysed using the Stylised Agri-environmental Policy Impact Model. Developed by the OECD Secretariat, SAPIM was used to analyse the policy-environmental linkages in the cases of Finland, Japan, Switzerland and the United States. Overall, these cases cover a broad range of policy instruments, agricultural situations and environmental conditions.

Determining the environmental impact of agricultural policies is complicated because land resources are highly heterogeneous, biophysical processes are complex and the actions tied to a specific policy do not take place in isolation, but within a broad and evolving socio-economic and technological context. Moreover, weather variability and climate change can have a considerable impact on environmental impacts. Models cannot replicate the real world, but can provide indications of the expected environmental outcomes. They do not substitute for an *ex post* analysis of an impact of a specific policy. Nevertheless, in order to aid policy makers in the design and implementation of cost-effective policies, it is necessary to have a better understanding of the linkages between policy instruments and environmental impacts. The value of this type of modelling is that it isolates policy choices from other developments and influences (such as weather) – this is often difficult without an appropriate modelling framework.

The key challenges are to first identify the change in farmers' actions on choice of production and production methods that are due to specific policy interventions, and then to determine the extent to which those actions affect environmental quality. While the conceptual relationships are relatively well-established, quantitative modelling is complicated for at least four reasons:

- Biophysical processes are complex and the relationship between a given practice and its environmental outcome is not always clear.
- Many of the environmental effects are site-specific, reflecting heterogeneous agricultural and environmental conditions, and thus the impacts cannot be readily extrapolated to the aggregate level, through generalised policy-response coefficients.
- There are in practice a mix of policy instruments applied and multiple environmental impacts which make modelling particularly difficult.
- Many of the environmental impacts are not measured (or measurable) in monetary terms- limitation to accounting for these externalities in the objective function from the perspective of producers and society.

In brief, SAPIM uses a combination of economic and biophysical models of representative farms (or production units) in the countries analysed. The SAPIM approach is pragmatic, modeling a representative farmer's decision-making at the field parcel level, because this level of detail is necessary for policy analysis to capture the complex economic and biophysical interactions that are site-specific and generally characteristic of agricultural production. Therefore, considerable SAPIM results cannot be extrapolated directly to more aggregate spatial levels.

SAPIM is specifically designed to capture the environmental effects of different agricultural policies through their impacts at the *intensive* margin (input-use intensity and production practices), the *extensive* margin (land-use allocation between different agricultural activities) and the *entry-exit* margin (land entering or leaving agriculture) under heterogeneous conditions.

The ***Finnish study*** investigated how environmental regulations, environmental taxes and payments for voluntary improvements affect agri-environmental performance in the case of crop production with varying land productivity that implies different input-use intensities and adoption costs with regard to agri-environmental measures. The effects of alternative policy instruments on nitrogen runoff and biodiversity were taken into account through their impact on input-use and land allocation choices. The results indicate that different agri-environmental policy instruments lead to very different outcomes in terms of land-use, production, and environmental externalities. The policy context in which these agri-environmental policies are implemented influences the effectiveness of such policies so that the *e.g.* presence of arable crop area payments reduces both the environmental effectiveness and cost-effectiveness of targeted and tailored agri-environmental policy measures.

Targeted agri-environmental payments in the absence of crop area payments most closely replicate the social optimum that incorporates both farm profits and value of environmental externalities. However, the minimum buffer norm and the buffer payment are very close to the optimum and are likely to entail significantly lower transaction costs than targeted agri-environmental payments so they may be preferred. Mandatory buffer strips and nitrogen taxes both lead to under-utilisation of agricultural land in favour of forestry. As regards agri-environmental payment programmes, conservation auctions – in which farmers bid for a limited amount of conservation contracts with the possibility to bid on both environmental performance and cost/bid – slightly outperforms other agri-environmental payment programmes. The outcome of this type of auction is quite close to the social optimum. In contrast, the conservation auction ranking solely by environmental score performed worse, and was even less welfare-enhancing than the traditional flat-rate payment (which is uniform irrespective of heterogeneous adoption costs and environmental benefits). Thus, applying a benefit-cost targeting could help to deliver a more cost-effective outcome.

The ***Swiss study*** examined the dairy sector, with the environmental effects focused on ammonia emissions, GHGs, and nitrogen and phosphorus surpluses. The policy instruments ranged from general agricultural policy measures to more targeted agri-environmental policy instruments, including both regulations and economic instruments. The results show that abolishing the milk quotas would have a significant impact on the profitability of production, through a consequent decrease in the milk price.

Most of the policy scenarios do not affect the total amount of manure produced on the farms, but they have an impact on the amount of manure applied on the farms and therefore on manure available off-farm, that is to be exported outside of the dairy farms. Both a nitrogen standard and a tax on nitrogen application effectively reduce the amount of manure applied on the farm and therefore result in increased exports of manure off the farms. Chemical nitrogen and phosphorus application rates are also affected significantly by some of the policy instruments, notably a tax on nitrogen fertilizer and an upper limit on nitrogen application. When policy-induced excess amount of manure is produced, the farms' adaptation strategy depends on relative costs. If transporting is the cheapest way to manage the excess manure, milk production is not affected, fertilizer use is reduced and the transportation of the excess manure represents just an additional cost. Because nitrogen can be supplied to the plants either from chemical fertilizers or from manure, the nitrogen surplus is poorly addressed in most policy scenarios except with a quantitative standard on nitrogen application that combines both sources of nitrogen input, or when a tax is applied on both fertilizer and manure application. Thus, the well-established problem of substitution of unregulated activities is manifested in dairy production.

The ***United States study*** focuses on the economic and environmental performance of conservation auctions relative to conventional agri-environmental policy measures. Three alternative land-use practices are analysed in this case study – land retirement for environmental purposes (buffer strips) and two alternative tillage methods to produce cultivated crops (no-till and conventional tillage). No-till and conventional tillage represent key management choices under the working lands agri-environmental programmes. In this application the sources of heterogeneity are both differential land productivity and environmental sensitivity of the land, more specifically differential propensity to erosion and thus nutrient and sediment runoff. The policy instruments analysed range from traditional regulatory and economic instruments, including fertilizer application limits and taxes, to different types of conservation auctions, including both uniform and discriminatory pricing types of auctions. Conservation auctions employ environmental benefit indices as environmental performance screens that help target conservation effort to those parcels providing large environmental benefits.

Relative to conventional tillage, no-till farming entails much lower energy and labour costs; however chemical costs are higher, due to abundance of perennial weeds and thus greater need for chemical control. For almost all analysed production systems (or production units) that involve different crop/rotation/tillage/erodibility combinations, no-till entails higher yields and thus – because of cost savings and higher revenues – the profits for no-till are higher than for conventional tillage. As regards environmental effects, no-till provides benefits in terms of reduced soil erosion and nitrogen runoff. However, phosphorus runoff increases due to accumulation of phosphorus in soil surface. As regards traditional policy instruments, the regulation mandating the allocation of 2.5% of land along watercourses as vegetated buffers effectively reduces sediment and nutrient runoff with reasonably small adoption costs to farmers. The combination of a mandatory buffer with a fertilizer tax (25%) to reduce application intensity provides only small additional environmental gains over a mandatory buffer alone, while the combination of a nitrogen application standard and a buffer strip is much more effective. This result seems to confirm a well-known problem with fertilizer taxes – they need to be relatively high to have an impact on behaviour. Hence, the combination of a nitrogen application limit and a mandatory buffer provides the instrument combination that is superior to other traditional policy instruments. As regards conservation auctions, the application of a uniform pricing auction reveals farmers' estimated adoption costs, and thus their information rent is reduced and budgetary cost-effectiveness is increased. On the other hand, a discriminatory payment gives farmers an incentive to place their bids above their adoption costs: low adoption cost farmers have a greater incentive to do so than high adoption cost farmers.

The ***Japanese study*** investigates the optimal land-use allocation and nitrogen application for a representative Japanese farm that consists of rice paddies, upland fields and abandoned land. The results indicate that different agri-environmental policy instruments lead to very different outcomes in terms of land use, production and environmental externalities. A special feature of this case study was to integrate rice production with an upland field crop (wheat) in the same analytical framework. In general, paddy fields could provide either positive or negative environmental effects, depending on farm management practices. Consequently, the incentives provided to farmers that encourage environmentally friendly rice production practices have a significant impact on environmental effects.

In every scenario, the results show that more parcels of land were allocated to rice paddy than to wheat. Social welfare is maximised when every parcel of land is used for rice paddy production, due to the net positive externalities (nitrogen purification and carbon sequestration) generated by paddy rice production. Agri-environmental policy could provide incentives to farmers to reduce the negative and increase the positive environmental externalities. But even in these policy scenarios, not every parcel is allocated to rice production, due to the small nitrogen response of rice paddy and high production costs. In order to control nitrogen runoff, reducing chemical fertilizer use and increasing organic fertilizer use increases social welfare. If farmers reduce chemical fertilizer applications, and are provided with payments for that, then this increases social welfare more that the application of a nitrogen tax. With regard to carbon sequestration, an agri-environmental payment subject to a minimum organic fertilizer application provides a higher level of social welfare than a per-unit payment related to organic fertilizer application. In the case of the relaxation of a rice production quota, the result is an increase in social welfare. It should be stressed that in all the policy simulations, the basis of the calculations is the domestic price of Japanese rice.

In each of the four case studies, the importance of the specific policy environment is emphasised. In particular, the "policy package" is crucial, as it defines the context, and therefore the assumptions that must be applied in order to have a realistic representation of the impact of policies. Each of the case studies highlights different production systems, environmental issues and policy context. The common thread underlying all case studies is the impact of various policies under heterogeneous conditions. Specifically, all the case studies have an important crop production component, in which the impact of fertilizer application is assessed in terms of crop yield and nutrient runoff.

In each case, the analysis modelled alternative scenarios of policy options to determine the production choices and environmental outcomes that would be optimum from the perspective of producers and society (only in the Finnish and Japanese case studies). The results highlight the well-established observation that when positive or negative environmental externalities are not factored in to farmers' decisions, then the production choices and environmental outcomes will reflect the weighing-up of private costs and revenues by farmers. Policy intervention can potentially raise social welfare through bringing those externalities into the equation. One of the innovative elements in this study was to develop a framework in which a mix of policies and multiple environmental impacts can be analysed. Clearly, in practice, this is a formidable task given the requirements of data and model specification, as well as the need to take account of transaction costs incurred by farmers and the policy process. Another innovation was to model auction systems as one of the options – having the potential advantage to reflect the heterogeneity of productivity, preferences and agro-ecological situations among farms and farmers to achieve well-defined and targeted agri-environmental policy objectives.

The analysis thus highlights the trade-offs involved – among production choices, policy instruments, and economic and environmental outcomes. The value of the SAPIM approach is that a flexible framework has been developed that can be used by the policy and research communities to analyse their specific interests. The quantitative results in this study arising from the various scenarios modelled should be viewed and interpreted as illustrative. In that regard, given the heterogeneities involved, only policy options within a given country context – and not across countries – should be compared. There has often been a lack of robust and quantitative analysis of the linkages between policy drivers and environmental outcomes in the agricultural sector. Decisions have been taken that have relied heavily on "trial and error" approaches to establish "which policies work". The approach described here is intended to redress the balance so that observed changes, for example, in nutrient runoff or greenhouse gas emissions or biodiversity associated with farming, can be better explained as to their cause, and in particular their link to policy.

The quantitative approach employed in this analysis is subject to limitations. In particular, given the site-specific nature of agri-environmental linkages, the results obtained with SAPIM must be interpreted as depending on the specific conditions modelled in each case study country. Thus, it is not appropriate to generalise the results to more spatially aggregate areas, or to make quantitative comparisons between case studies. In addition, there is a range of uncertainty on key parameter estimates that then have consequences on the results obtained from modelling. This was explored, albeit only briefly in the sensitivity analysis. A key source of uncertainty is arguably related to the valuation estimates of social benefits in the case studies.These limitations are not unique to the SAPIM analysis – quantitative results obtained from any other model framework would be subject to similar limitations. Improving the reliability of the modelling is however an essential and challenging task.

Nevertheless, with these limitations in mind, some general policy lessons can be drawn from this analysis:

- Policies need to recognise the inherent heterogeneity of the agricultural sector. The environmental impacts and the adoption costs of a given policy instrument vary significantly depending on location-specific conditions, and therefore spatial targeting and tailoring of policy incentives should be considered.
- Unregulated polluting activities may be important, thus the need to control all relevant inputs, processes and technologies.
- The effectiveness of a given policy instrument is linked to the overall policy package – policies should be evaluated considering interaction with all other policies rather than in isolation.
- Environmental co-benefits and trade-offs need to be recognised as an integral part of policy design and implementation.

Modelling is but one element in the arsenal of information available to policy makers in designing and implementing policy. Tracking the environmental performance of agriculture through carefully designed and measured indicators provides comprehensive information on agri-environmental trends and can pinpoint where actions are especially needed. Understanding the range and characteristics of agricultural, agri-environmental and environmental policies can help identify the incentives and disincentives facing farmers in making production decisions that have environmental consequences. In this respect, SAPIM provides a valuable, pragmatic and flexible approach to help explain the link between policies and environmental outcomes.

Annex A
The Finnish case study

Table A.1. Estimation of the weights for biodiversity and runoff reduction

Biodiversity (B0) and runoff reduction (B1)

Non-linear Regression Summary Statistics

Dependent Variable NORMEBI

Source	DF	Sum of squares	Mean square
Regression	2	29.11133	14.55567
Residual	98	.58293	5.948284E-03
Uncorrected total	100	29.69427	
(Corrected total)	99	4.41821	

R squared = 1 - Residual SS / Corrected SS = .86806

			Asymptotic 95 % Asymptotic Confidence Interval	
Parameter	Estimate	Std. Error	Lower	Upper
B0	430403317	.023813339	.383146521	.477660113
B1	.569596683	.025683448	.518628717	.620564649

Asymptotic Correlation Matrix of the Parameter Estimates

B0	1.0000	-.8357
B1	-.8357	1.0000

Annex B
The Swiss case study: Background data

Table B.1. Characteristics differentiating dairy production systems analysed by FAT

	Categories	Available options
Forage provision	Winter feeding (with silo)	Grass silage Maize silage Hay (dried on ground)
	Winter feeding (without silo)	Hay (first-cut, barn-dried); Ventilated subsequent cuts
	Summer feeding (with silo)	Pasture-grazing only (pp) 50% harvested grass & 50% pasture grazing (hp) 50% silage & 50% pasture grazing (sp) Year-round silage (ss)
	Summer feeding (without silo)	Pasture-grazing only (pp) 50% harvested grass + 50% pasture grazing (hp)
Buildings	Forage stocking	Dry forage (without silo) Round bails with maize-silo Automated tower silo Horizontal silo
	Barn structure	Free-stall housing Cubicles Loose housing
	Milking equipment	Fixed milking unit (herringbone) Mobile milking unit (herringbone)
	Forage distribution	1:1 Animal/feeding post ratio, 2:1 Animal/feeding post ratio, *Ad-libitum* feeding with automated forage provision self-service from horizontal silo (combined with pasture-grazing)
Herd type	Reference cow	6 700 Kg ECM*/year
	Seasonal grazing cow	6 280 Kg ECM/year
	Medium productivity	7 740 Kg ECM/year
	High productivity	9 110 Kg ECM/year
	Small cow (Jersey)	6 130 Kg ECM/year
Herd size	30-100 cows	Medium mechanisation (m1) Higher mechanisation (m2) Outsourcing - minimal mechanisation (m3)

* ECM = Energy-corrected milk.

Source: Gazzarin and Schick (2004).

Table B.2. Emission factors for different combinations of housing system, manure storage and manure spreading

Housing system	Manure storage	Manure spreading	Ammonia emissions (% of N tot)
Tied stall	Open storage	Broadcast application	34
Tied stall	Open storage	Trail hose application	26
Tied stall	Covered storage	Broadcast application	29
Tied stall	Covered storage	Trail hose application	20
Cubicle house	Open storage	Broadcast application	35
Cubicle house	Open storage	Trail hose application	28
Cubicle house	Covered storage	Broadcast application	31
Cubicle house	Covered storage	Trail hose application	22

Source: Menzi (2006).

Annex C
The Japanese case study: Empirical specification

The detailed modelling specifications for Japanese case study are provided in this Annex.

Profit function

Farmer's profits from production in the absence of government intervention are

$$\pi^i = p_i y_i - cx_i - w_i n_i - o_i \quad for\ i = 1,2 \tag{19}$$

where p_i refer to the price of crops, y_i to the yield/10 ares, c to the fertilizer (nitrogen) price, w_i to wage rate per hour and o_i to other cost. The model employs a quadratic Nitrogen response function, $y_i = a_i + \alpha_i x_i + \beta_i x_i^2$ where x_i refer to the amount of N application (kg/10 ares). They are estimated for crop 1 (rice) and crop 2 (wheat).

When famers consider (to use) organic matter application x_{oi} in addition to (instead of) chemical N fertilizer x_{ci}, total amount of N application to the agricultural field is summation of N fertilizer and N content of organic matter. Despite recommend organic matter application amount (*e.g.* 1.0-1.5 t/10 ares for paddy field), the implementation is inactive (88 kg/10 ares) due to the following problems.

- Difficulties to realise of the manure application effects from farmers' viewpoints due to the diverseness of manure quality;
- Huge amount of application is needed comparing with chemical (high spreading cost);
- Lack of co-operation among crop and livestock farming (high transportation cost).

Several surveys have already revealed that the effect of organic matter application to the yield is statistically positive. According to Shibahara *et al*. (1999), continuous long-term application of organic matter retrench the total N application for the certain amount of the yield due to the high N absorption of organic matter. In fact, according to the answers for the mail survey by the Livestock Environmental Improvement Organisation in 2003, the reason for the organic matter applications for farmers were improvement of the quality of products or stabilisation of production *via* keeping fertility of soils, keeping the soil soft and activate soil microbe.

Average N content in organic fertilizer (cow manure) is set as 0.7% based on Okayama prefecture agricultural centre (2008, originally from MAFF), and then total amount of N application is expressed as

$$x_i = x_{ci} + 1000 \bullet x_{oi} \bullet 0.007 \quad (20)$$

1 000 means the conversion of unit from tonne to kg.

Generally, N requirement *substitution rate (%) = the amount of organic fertilizer (kg/10a)* N content rate (%)*Fertilizer efficiency (%), where fertilizer efficiency is 30% (Okayama prefecture agricultural centre, 2008, originally in Nishio, 2007).

Suppose that positive effect for yield is expressed as $\Phi_i(x_{oi})$, and that of paddy is supposed as 5% and wheat is as 10% under the 1t application, which is based on the several field survey data (*e.g.* Miyazaki prefecture, 1999; Shibahara *et al.*, 1999). Taking into consideration of additional cost for organic matter application, profits function is expressed as follows:

$$\pi^i = p_i(a_i + \alpha_i x_i + \beta_i x_i^2)\Phi_i(x_{oi}) - cx_{ci} - (c_{op} + c_{ot} + c_{os})x_{oi} - w_i n_i - o_i \quad for\ i = 1,2 \quad (21)$$

where c_{op} refer to the price of organic matter (JPY/tonne), c_{ot} to transportation cost (JPY/tonne) and c_{os} to the spreading cost (JPY/tonne).

Nitrogen response function

Rice paddy

Quadratic nitrogen response function of rice paddy was estimated by over 50 sample field surveys data which was collected by Toriyama (2000):

$$y_1 = 368.6 + 31.7x_1 - 1.4x_1^2 (R^2 = 0.61) \quad (22)$$

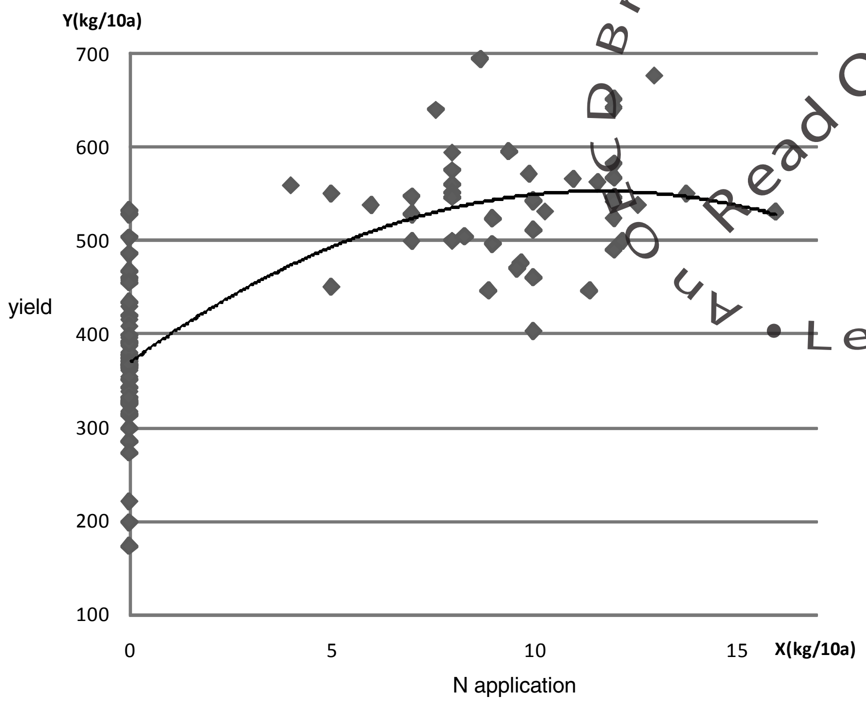
Even if without fertilizer, nutrition in the irrigation water affects to yield. It is generally said that the yield without fertilizer decrease only 1/5 due to the high fertility of paddy.[1] To reflect actual yield in paddy field, a_i is given by fixed value to exclude the effect of the irrigation water, and then the land quality *q* is incorporated into the response function. Response function is expressed as

$$y_1 = 368.6 + (e_0 + e_1 q)x_1 - (\mu_0 + \mu_1 q)x_1^2 \quad (23)$$

According to data in Toriyama (2000), spread of yield is about 30% under the average N application amount (see Figure C.1). Consequently, the ranges of parameters are set as $22.19 \le e_0 + e_1 q \le 41.21$ *and* $0.98 \le \mu_0 + \mu_1 q \le 1.82$. When q is distributed uniformly between 1 to 60, parameters e_0, e_1, μ_0 and μ_1 are estimated as follows:

$$e_0 = 22.868, e_1 = 0.322, \ \mu_0 = 0.994 \text{ and } \mu_1 = 0.014$$

Figure C.1. The relationship between nitrogen application and yield for rice



Source: Toriyama (2000).

Wheat

Quadratic nitrogen response function of wheat (converted from rice cultivation) was estimated by National Agricultural Centre data sets (1989):

$$y_2 = 214.9 + 45.6x_2 - 1.2x_2^{\,2} (R^2 = 0.99) \tag{24}$$

However, this survey was undertaken to collect the highest yield data. Therefore the function (24) could not be a representative average response function. Due to the lack of enough data to reflect land quality variety, average and lowest yield response function are estimated based on the assumption that spread of yields is about 40 %. This 40% is based on the variety of targeted yield under the average N application, which is determined in *The Nitrogen Application Standard* by each local government.

Wheat response function to nitrogen is expressed as

$$y_2 = 214.9 + (h_0 + h_1 q)x_2 + (\eta_0 + \eta_1 q)x_2^{\,2} \tag{25}$$

where $19.54 \le h_0 + h_1 q \le 45.6$ *and* $0.51 \le \eta_0 + \eta_1 q \le 1.2$.

Then, the following parameter s are obtained, $h_0 = 19.101$, $h_1 = 0.442$, $\eta_0 = 0.526$ and $\eta_1 = 0.012$

Nitrogen runoff and purification function

Rice paddy

It is difficult to formulate the relationship between the amount of N application and its impact by easy-to-use way, because N runoff from irrigation and meteoric water might affect to N balance in rice paddy. Generally, N runoff from paddy is explained as follows:

[N runoff (surface runoff＋subsurface flow)]＝[The effect of irrigation water-load]＋[The effect of meteoric water-load]+ [The effect of N application].

In this regard, Kunimatsu and Muraoka (1989) proposed that the polluting load L is given by $L = \alpha C_{i1} Q_{i1} + \beta C_{i2} Q_{i2} + \lambda X$, where C_{i1} and C_{i2} are concentration of irrigated water and meteoric water, Q_{i1} and Q_{i2} denote their volume, respectively. X is amount of fertilizer application. α, β and γ are each coefficients. They also said that, meanwhile, the amount of N into the agricultural land from fertilizer is fairly larger than those of irrigated water and meteoric water. Ignoring the effect of two terms $\alpha C_{i1} Q_{i1}$ and $\beta C_{i2} Q_{i2}$, the relational expression is $L = \lambda F$. Taking into consideration of the large effect of fertilizer application as stated in Kunimatsu and Muraoka (1989) and conveniences for economic optimization, the Secretariat tried to estimate the relationship between N application and runoff by exponential form (*e.g.* Tabuchi and Takamura, 1985) as:

$$z_i = \chi_i \exp(\delta_i x_i) \tag{26}$$

where z_i refer to the amount of N runoff (surface and subsurface) and x_i to the amount of N application.

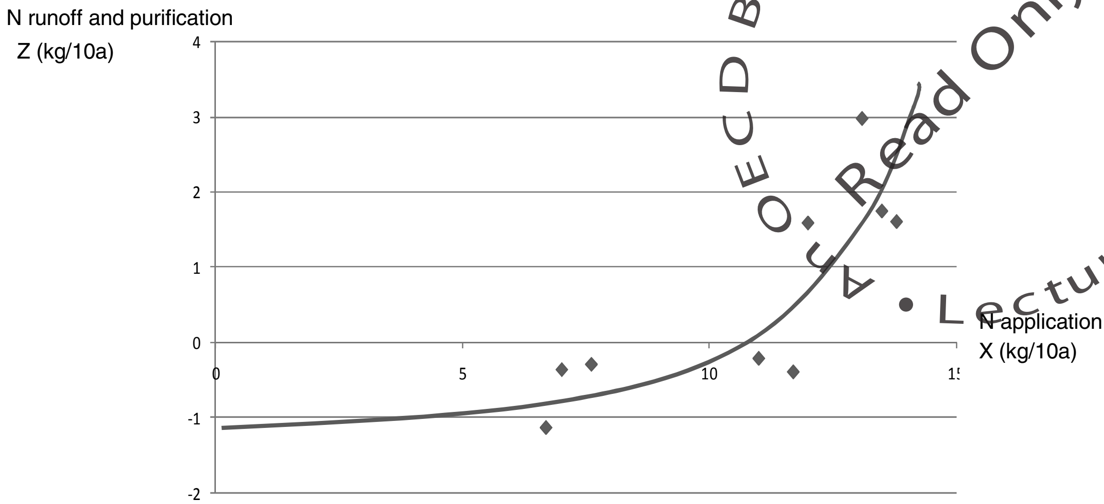
Paddy fields could be N removal sites or pollution sites depending on agricultural activities and nitrogen concentration of irrigation water. It is well known that paddy fields and wetlands effectively improve water quality by removing nitrogen due to denitrification and absorption, which is effective only when irrigation water has strong concentration. Although the nitrogen movement in paddy is not simple, relationship was estimated by using Kunimatsu and Muraoka (1989) and recent field survey data which were collected by Shiga prefecture during paddy cultivation period (Figure C.2). Exponential relation was found between the amount of N application and runoff.

$$z_1 = 0.0062 e^{0.465 x_1} - 1.14 (R^2 = 0.54) \tag{27}$$

where z_1 refer to the amount of N runoff from paddy and x_1 to the amount of N application in paddy.[2]

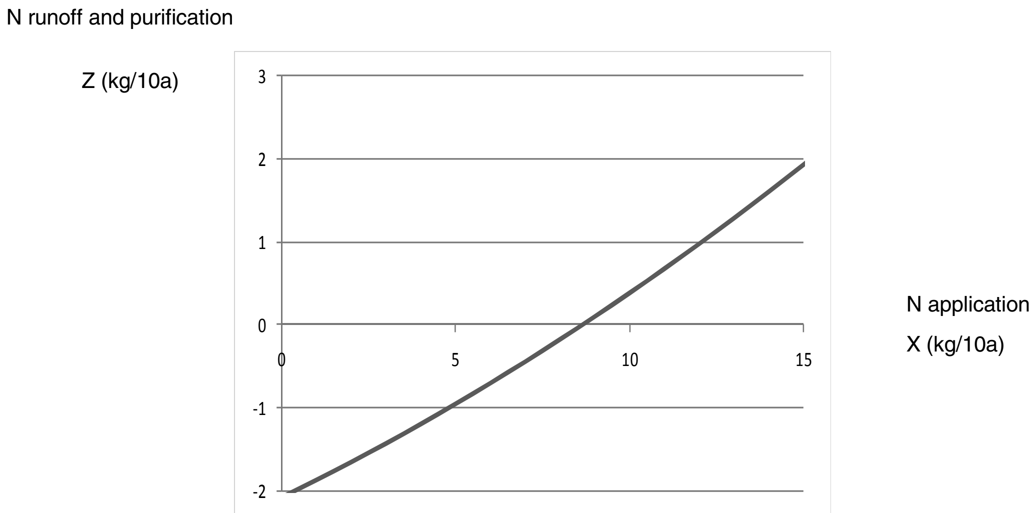
The number of observations is not enough to examine the validity and also R^2 is not so high to obtain the robust results. At this point, another curve was estimated by different approach based on N balance in paddy field: [Net N runoff (kg/10a)] = 0.0042 × [N applied (kg/10 a)]^2 + 0.2049 [N applied (kg/10 a)] – 2.0858,[3] which is shown in Figure C.3. As shown, the overall shape is not similar to Figure C.2. However, in the limited range for general N application in paddy field (5-10kg/10 a), the differences between those of two curve is not particularly large.

Figure C.2. Field data on N runoff and purification in paddy field



Sources: Kunimatsu and Muraoka (1989) and Shiga prefecture (2007).

Figure C.3. N runoff and purification curve alternative estimation



Source: Author's calculations.

Wheat

In a precise sense, soil condition, crops, cropping season and methodological condition could affect to N runoff, nevertheless approximately 30% of applied N could runoff as the average in Japanese condition (Kunimatusu, 1989; Takedam 1997; Shiratani, 2004).[4] But linear function is not appropriate for optimisation of the social welfare function. Consequently, exponential form was estimated on the basis of Japanese field data which was sorted out by National Institute for Ago-Environmental Science (NIAES),

$$z_2 = 1.129e^{0.114x_2} \; (R^2 = 0.19) \tag{28}$$

where z_2 refer to the amount of N runoff and x_2 to the N application.

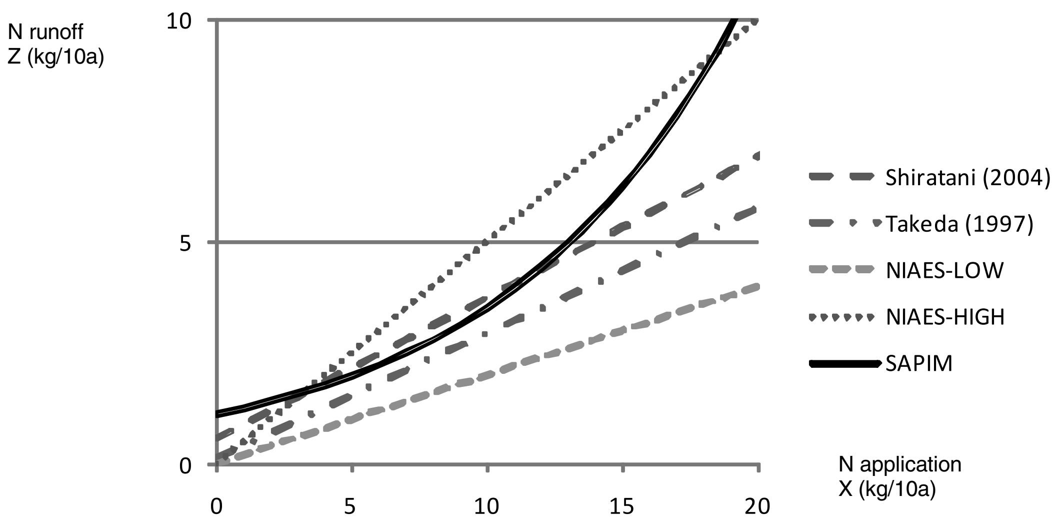
Due to the lack of enough observation (there is no information on slopes), R^2 is not sufficiently high. To verify the robustness of estimated exponential curve, the linear functions and general nitrogen runoff ratio (Table C.1) were compared in Figure C.4. Under the average amount of N application, say lower than 20 kg/10a, there is consistency with the other estimation results as shown in Figure C.4.

Table C.1. Nitrogen runoff ratio

Pollution source	Land use	N runoff ratio (%)
Fertilizer application	Paddy field	0-10
	Upland field	20-50
	Grassland	5-20
Manure	Livestock farm	60-100

Source: National Institute for Agro-environmental Science (1998).

Figure C.4. Estimated nitrogen runoff function form in upland field



Source: Author's calculations.

GHG emission and sequestration function[5]

Each category of emissions are considered here one by one based on the IPCC (2006), MOE (2008) and field survey data for country specific coefficients.

CH_4 emission

Rice paddy

It is well known that rice cultivation is a main anthropogenic source of CH_4 emissions. According to the IPCC (2006), several rice cultivation characteristics should be considered in calculating CH_4 emissions: regional differences in rice cropping practices, multiple crops, water regime, ecosystem type, flooding pattern etc. In addition to these factors, the impact of organic amendments on CH_4 emissions is huge, and amount of the applied material and CH_4 emission can be described by a response curve. Yan *et al*. (2005) conclude that organic amendment and water regime in the rice-growing season were top two control variables, and climate was the least critical variable.

The water regime in the rice growing season was classified as: continuous flooding, single drainage, multiple drainage, wet season rain fed, dry season rain fed, deepwater, or unknown. In Japan, most of paddy fields (98%) are intermittently flooded. There is scaling factor for water regimes during the cultivation period relative to continuous flooded field, however, intermittently flooded (multi aeration) in the IPCC category is different in nature from the intermittently flooded paddy field (single aeration) concept in the IPCC Guideline.[6]

IPCC (2006) set a default seasonal CH_4 emission factor for rice *under continuous flooding conditions and without organic matters*. Scaling factors (SF) are used to estimate CH_4 emissions from rice fields to reflect each countries situation such as water regimes or organic matters. But IPCC (2006) said that country-specific emission factors and scaling factors should only be used to reflect appropriate condition if they are based on well-researched and documented measurement data (IPCC, 2006). A default emission factor is 1.30 kg CH_4/ha/day (23.4 kg/10a/180 days).

The basic equation to estimate CH_4 emission from rice cultivation per 10a is defined in equation (29), which is converted form IPCC (2006).

$$CH_4 = EF_c \bullet SF_w \bullet SF_p \bullet SF_o. \tag{29}$$

where, CH_4 (t CH_4/10a/yr) is annual CH_4 emissions from rice cultivation, EFc is the baseline emission factor for continuously flooded fields without organic amendments, SFw is the scaling factor to account for the differences in water regime during the cultivation period, SFp is the scaling factor to account for the differences in water regime in the pre-season before the cultivation period and SFo is the scaling factor to account for the differences in both type and amount of organic fertilizer applied.

As for emission factors, Japan has country-specific emission factors for intermittently flooded paddy (single aeration), which has estimated as 12.96 $gCH_4/m^2/yr$ (0.001296 tCH_4/10a/yr) in MOE (2008).[7] This data reflects both of Japanese specific emission factors and water regimes.

The scaling factor of organic fertilizer is defined as follows (IPCC, 2006):

$$SF_0 = \left(1 + \sum_j x_{oj} \bullet CF_j \bullet 10\right)^{0.59} .^{8} \tag{30}$$

where x_{oj} (t /10a) is application amount of organic fertilizer j in dry weight for straw and fresh weight for others, CF_j is conversion factor for organic fertilizer j (in terms of its relative effect with respect to straw applied shortly before cultivation) as shown in Table C.2.

As shown in Table C.2 and Figure C.5, the impacts of organic fertilizer are much differing in their types and application amount. On present showing that rice straw is applied in 60% of agricultural land, the other manure is in 20% and no application is 20% (MOE, 2008) in Japan, otherwise MAFF is strongly promoting the manure application from the perspective of (net) GHG reduction and keeping fertility of the soil. The conversion factor of farm yard manure is, therefore, going to be used in this modelling. This choice of control variable is also important at the policy simulation stage, because manure application takes further effort for manure collection and spreading (Japan Soil Association, 2009).

By using country-specific data, CH_4 emission (t CH_4/10a/yr) equations (15) are re-written as follows:

$$CH_4 = 0.001296 \bullet (1 + x_o \bullet 0.14)^{0.59} . \tag{31}$$

The Guidelines for Enhancement Fertility of Soil recommend that normal manure application amount is 1.0-1.5t/10a in paddy, but the actual application is decreasing from 451 kg/10a (y1970) to only 88 kg/10a (y2005) due to decoupling of crop and livestock farming and aging of farm labour forces.

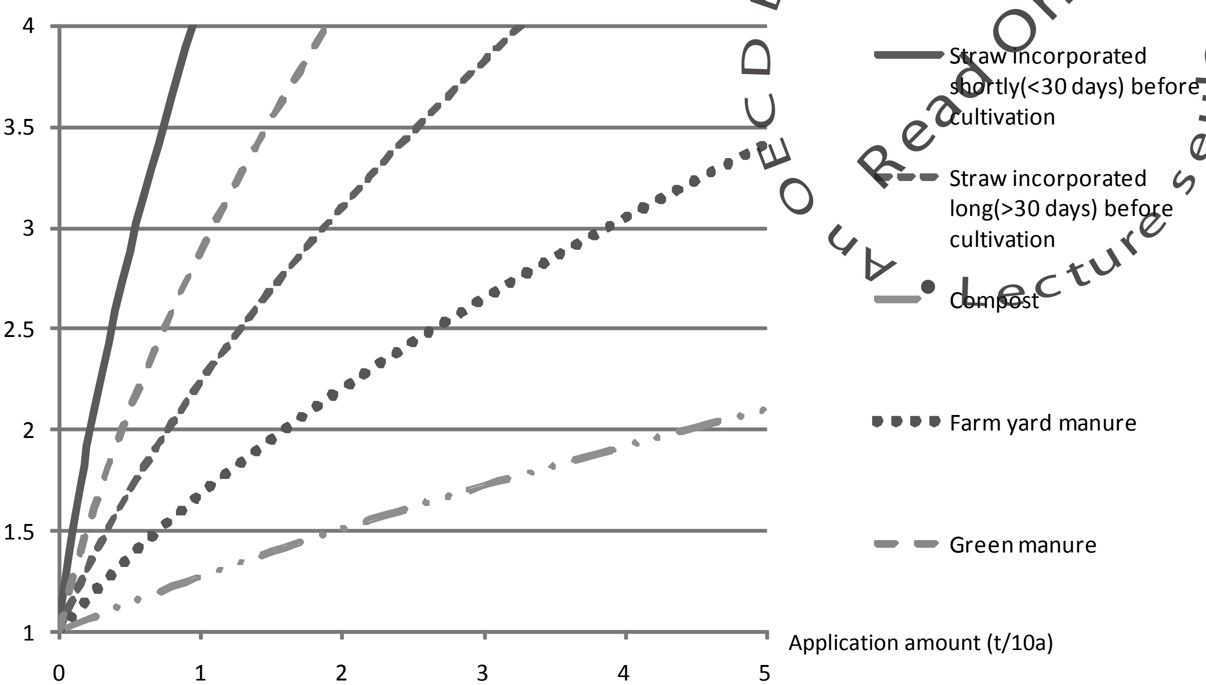
Table C.2. Default conversion factor for different types of organic amendment*

Organic amendment	Conversion factor (CF_i)	Error range
Straw incorporated shortly (<30 days) before cultivation	1	0.97-1.04
Straw incorporated long (>30 days) before cultivation	0.29	0.20-0.40
Compost	0.05	0.01-0.08
Farmyard manure	**0.14**	**0.07-0.20**
Green manure	0.50	0.30-0.60

* Straw is in dry weight, but others are in fresh weight.

Sources: Yan *et al.* (2005); IPCC (2006).

Figure C.5. The relationship between the amount of organic amendment application and the size of conversion factor



Sources: Yan *et al.* (2005); IPCC (2006).

Upland

Methane generation is not possible, if soil is not maintained in an anaerobic state. Upland soils are normally oxidative and in aerobic condition, therefore CH_4 is not produced.

N_2O emission

Direct emission

The fertilizer application and ploughing of organic soil cause ammonium ions inside the soil, and then N_2O is emitted in the process of oxidizing the ammonium ions into nitrate-nitrogen under aerobic conditions. In addition, N_2O is emitted *via* denitrification. EFs for N_2O associated with the application of synthetic fertilizers to farmland soil were set based on actual data conducted in Japan, and same emission factors are also used for those of organic fertilizer. Because there was no the significant differences between EFs of synthetic fertilizers and organic fertilizers, analysing data on N_2O emissions from Japanese agricultural fields. Akiyama *et al*. (2006) estimated EFs of Japanese rice paddies and upland fields as **0.31**% (±0.31%) and **0.62%** (±0.48%), respectively. Uncertainties still remain, but these EFs are used in MOE (2008),[9] as shown in Table C.3.

Fertilizer application derived N_2O emission is,

$$N_2O_{direct\_i} = \frac{1}{1000} \bullet EF_{di} \bullet (x_{ci} + x_{oi} \bullet 0.007 \bullet 1000) \bullet \frac{44}{28}. \tag{32}$$

where $N_2O_{direct\_i}$ refer to direct N_2O emissions derived from fertilizer application in land use i (t N_2O), EF_{di} to emission factors (kgN_2O-N/kgN) (for paddy: 0.0031 and for upland crop: 0.0062), x_{ci} to the amount of chemical fertilizer application (kgN), x_{oi} to application amount of organic fertilizer (tonnes/10a) and 44/28 means the conversion of N_2O-N emission to N_2O emission.

Indirect emission

In the next step, the estimation methods of indirect emission are going to be considered. When E_{adi} is N_2O emissions associated with atmospheric deposition (kgN_2O) and E_{li} is emissions associated nitrogen leaching and runoff (kgN_2O), indirect emission $N_2O_{indirect\_i}$ is expressed as follows:

$$N_2O_{indirect\_i} = E_{adi} + E_{li}. \tag{33}$$

Emissions from atmospheric deposition can be expressed as,

$$E_{adi} = EF_{ad} \bullet (x_i \bullet Frac_{GASF} + N_D \bullet Frac_{GASM}) \bullet \frac{44}{28}. \tag{34}$$

where E_{ad} refer to N_2O emissions from atmospheric deposition, EF_{ad} to emission factors (kgN_2O-N/kgN) , x_{ci} to the amount of nitrogen fertilizer, $Frac_{GASC}$ (0.1) to the rate of deposition chemical fertilizer (kgNH_3-N+NOx-Nkg), N_D to the amount of N in applied organic fertilizer, $Frac_{GASO}$ (0.2) is the rate of deposition from organic fertilizer (kgNH_3-N+NOx-Nkg). Therefore,

$$E_{adi} = \frac{1}{1000} \bullet 0.01 \bullet (x_{oi} \bullet 0.1 + x_{ci} \bullet 0.007 \bullet 1000 \bullet 0.2) \bullet \frac{44}{28}. \tag{35}$$

Emissions from nitrogen leaching and runoff (E_{li}) are defined by,

$$E_{li} = \frac{1}{1000} \bullet EF_l \bullet z_i \bullet \frac{44}{28} \tag{36}$$

where EF_l refer to the N_2O emission factor from nitrogen leaching and runoff (kgN_2O) and z_i to the runoff amount (kgN). Although the proportion of N runoff against application is set as 30% in the MOE (2008), equations (13) and (14) which are estimated in this SAPIM analysis are used for the leaching and runoff amount.

$$E_{li} = \frac{1}{1000} \bullet 0.0124 \bullet Z_i(x_i) \bullet \frac{44}{28} \tag{37}$$

All of the emission factors used this section are summarised in Table C.3.

Table C.3. N_2O emission factors for fertilizer in agricultural soils

	Crop species	Emission factor (kgN$_2$O-N/kgN)	Uncertainties (kgN$_2$O-N/kgN)
Synthetic and organic fertilizer	Paddy rice	0.31%	±0.31%
	Upland crop	0.62%	±0.48%
Indirect emission (atmospheric deposition)		1.00%	±0.5%
Indirect emission (nitrogen runoff and leaching)		1.24%	±0.6-2.5%

Source: MOE (2008).

CO_2 emissions and sequestration

As already mentioned, only four countries are elected to include “Cropland Management and Grazing Land Management (the key activities relevant to agricultural industries)” in their accounts for the Kyoto protocol first commitment period, however the relationship between farm management and SOC could be considerable in anticipation of post-Kyoto discussion. Japan has country-specific continuous survey data, which had been undertaken in 52 areas for paddy and 26 are for upland crops. Overall average data reveals that organic matter applications increase the amount of carbon sequestration: 1t /10 ares manure application cause 40.6-77.4 kgC/10 ares sequestration in paddy field and 1.5 t/10 ares manure results 37.3-170.9 kgC/10 ares sequestration in upland.

The amounts of carbon sequestration *via* organic matters application differ from soil type to soil type. In this analysis, gray lowland soils and gley soils for rice paddy and andosols for upland crop are used for curve estimation respectively, because these soil types are one of the representative soils which are widely distributed in Japan, as shown Table C.4. In addition, the use of dominate type soil could permit extrapolation to more spatially aggregate level.

The amount of carbon sequestration is expressed as follows,

$$CO_{2i} = \sum Seq_i \bullet \frac{44}{12} \qquad (38)$$

Regarding specification of function form, since there is upper bound for carbon sequestration capacity, polynominal functions are estimated by using data from MAFF which include the amount of application per year, the increased amount of soil carbon in each soil types. And then (39) and (40) are estimated for paddy and upland field, respectively (Figure C.6).

$$Seq_1 = -0.0062x_o^{\,2} + 0.052x_o \ (R^2 = 0.80) \qquad (39)$$

$$Seq_2 = -0.0013x_o^{\,2} + 0.022x_o . \ (R^2 = 0.69) \qquad (40)$$

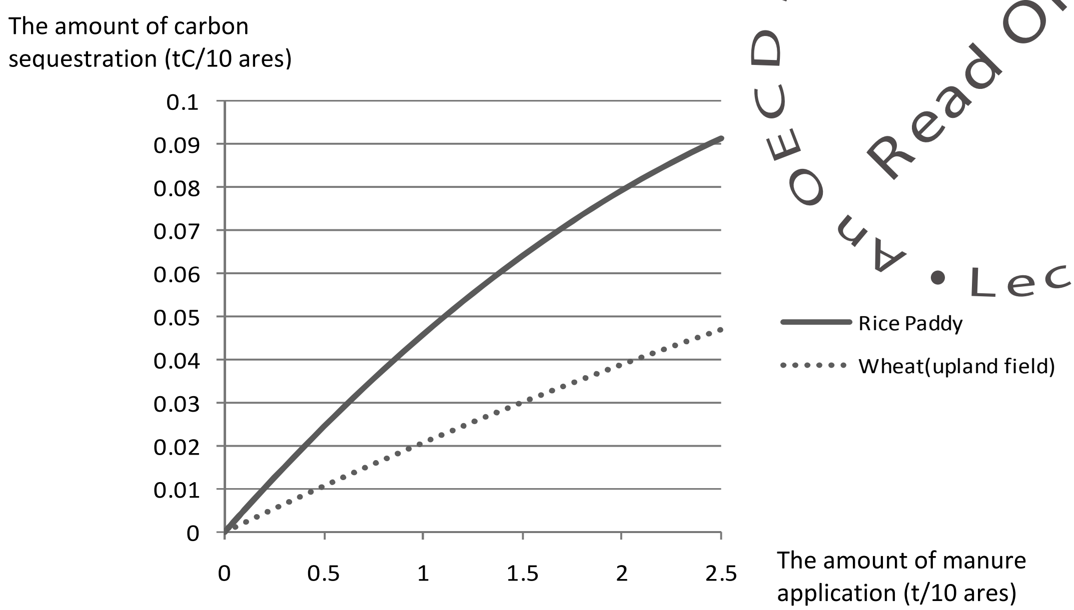
Table C.4. The amount of carbon sequestration in the case of manure application

(1.0 t/10 ares for paddy and 1.5 t/10 ares for upland)

		The amount of carbon sequestration (tC/10 ares/yr) **(A)**	**Area** (1 000 ha) **(B)**	**The amount of carbon sequestration** (1 000 tC/yr) **(A*B)**
Paddy field	***Grey lowland soil***	***0.0472***	***718***	***339***
	Gley soils	***0.0406***	***604***	***245***
	Wet andosoils	0.0774	186	144
	Yellow soils	0.0515	98	50
	Brown lowland soils	0.0752	96	72
	SUM		1 702	850
Upland crop field	***Andosols***	***0.0373***	***1 584***	***591***
	Brown forest soils	0.0644	450	299
	Yellow soils	0.0696	308	214
	Grey lowland soil	0.1709	144	246
	SUM		2 486	1 350
SUM				2 200

Source: MAFF (2008b; 2008c).

Figure C.6. The relationship between the amount of manure application and the amount of carbon sequestration



Source: Author's calculations.

The other parameters for the model are reported in Table C.5.

Table C.5. Parameter values in the numerical application

Parameter	Symbol	Value	Unit	Source
Price of crop:			JPY/kg	MAFF stat. (2008)
Rice	P_1	219		13 130 JPY/60kg
Wheat	P_2	152		9 144 JPY/60kg
Price of nitrogen fertilizer	C	183.8	JPY/kg	MAFF stat. (2008)
Labour cost:			JPY/10a	MAFF stat. (2008)
Rice	W_1n_1	26 087		W_1 18.5h/10a, n_1: 1 410 JPY/h
Wheat	W_2n_2	6 699		W_2: 4.4h/10a, n_2: 1 523 JPY/h
Organic matter:				MAFF (2008)
Price of organic matter	C_{op}	5 000	JPY/t	
Transportation cost	C_{ot}	1 000	JPY/t	
Spreading cost	C_{os}	2 000	JPY/t	
Other cost:			JPY/10a	MAFF stat. (2008)
Rice	O_1	62 267		
Wheat	0_2	43 972		
Monetary valuation:				
N removal benefit		6 563	JPY/kg	Shiratani *et al.* (2004)
N runoff damage		650	JPY/kg	Shiratani *et al.* (2004)
GHG damage		7 039	JPY/t	Baker *et al.* (2007)

Source: Author's compilation.

Notes

1. The following are the main functions of irrigation water in paddy: 1) natural supply of nutrient, 2) nitrogen fixation, 3) accumulation of organic and easily-absorbed and 4) less soil erosion.

2. Suppose that the N content in organic fertilizer is not included in this equation, because N in organic fertilizer could be serious problem only when the application amount is enormous. In this model, the maximum of organic is approximately 1.5 t/10 ares due to the economic reason (high additional cost).

3. This function is estimated by Dr Shiratani at National Institute for Rural Engineering as one example.

4. For example: Shiratani (2004) estimates as [N runoff(kg/10a)] ＝**0.317*** [the amount of N application]+0.5887, and Takeda (1997) estimates as [N runoff (kg/10 ares)] ＝**0.281*** [the amount of N application]+1.33

5. Calculations based on *2006 IPCC Guidelines for National Greenhouse Gas Inventories: Volume 4 Agriculture, Forestry and Other Land Use* (IPCC, 2006) and "National Greenhouse Gas Inventory Report of Japan" (MOE, 2008), as far as possible.

6. See also Ministry of the Environment Greenhouse Gas Inventory Office of Japan [GIO], CGER, NIES (2008) for detailed information (www-gio.nies.go.jp/aboutghg/nir/2008/NIR_JPN_2008_v4.0_E.pdf).

7. General emission factors which are used here are estimated by the Secretariat from CH_4 emission factors on each soil type and the proportion of Japan's surface area by soil types.

8. The exponent in this equation is provided with uncertainty range of 0.54-0.64.

9. The emission factor of Japan is lower than that of default value in the IPCC (2006). It is the reason that the volcanic ash soil that is widely distributed in Japan releases little N_2O emissions.

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Linkages between Agricultural Policies and Environmental Effects

USING THE OECD STYLISED AGRI-ENVIRONMENTAL POLICY IMPACT MODEL

Improving the environmental performance of agriculture is a high priority for OECD countries. But measuring and evaluating the impact of agri-environmental policies on the environment can be challenging, as it requires linking economic and biophysical models in country-specific contexts.

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This report notes that, overall, the diversity of conditions across sectors and countries makes it difficult to generalise the impact of agri-environmental policies beyond the situations that are modelled. Nevertheless, some wider policy messages emerge. Drawing on the four case studies examined, this report recommends that, polluting activites that are not regulated should be included in policy design; the existing overall policy environment needs to be taken into account in evaluating agri-environmental policies; and environmental co-benefits and trade-offs need to be recognised.

Green growth policies can stimulate economic growth while preventing environmental degradation, biodiversity loss and unsustainable natural resource use. The results from this publication contribute to the Green Growth Strategy being developed by the OECD as a practical policy package for governments to harness the potential of greener growth.

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