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The Environmental Dimension of Multifunctionality: Economic Analysis and Implications for Policy Design

Doctoral Dissertation

Jussi Lankoski

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The Environmental Dimension of Multifunctionality: Economic Analysis and Implications for Policy Design

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Abstract

Multifunctional agriculture refers to the fact that agriculture produces jointly a number of commodity and non-commodity outputs, and some of these non-commodity outputs exhibit the characteristics of externalities and public goods. Thus, multifunctionality provides an integrated framework for the simultaneous consideration of multiple commodity and non-commodity outputs.

Multifunctionality constitutes a complex problem from the perspective of policy design and implementation. Finding out the socially optimal bundle of multiple commodity and non-commodity outputs involves the identification of the important outputs as well as their relative significance, and policies conducive to multifunctional agriculture must simultaneously address several outputs, commodity and non-commodity ones. Moreover, the heterogeneous conditions under which agriculture operates create a spatial dimension in the supply of commodity and non-commodity outputs. That is, there are spatial differences in productivity and, hence, in the production costs of commodity and non-commodity outputs. Finally, there are trade-offs between the precision of the policy instruments and their information requirements and related administrative costs.

The main objective of the present study was to contribute to the understanding of the implications of multifunctionality for effective agri-environmental policy design. The main research question addressed was the performance of various types of policy interventions in achieving the optimal bundle of multifunctional outputs under heterogeneous conditions.

The scope of the present study was restricted to the environmental dimension of multifunctionality. Two commodity outputs and three environmental non-commodity outputs (nutrient runoffs, landscape diversity, and agrobiodiversity) were analysed, taking into account jointness and heterogeneity in their supply and the externality and public good aspects in their demand.

In this study an analytical model was developed, and then empirical results were obtained by calibrating the model to Finnish data. First, the farmer's private optimum was compared to the social optimum where nutrient runoffs, landscape diversity, and agrobiodiversity were valued at their social marginal values. Next, solutions were developed for the first-best differentiated policy instruments and the second-best uniform and semi-uniform policy instruments.

Finally, farm income support measures and environmental cross-compliance schemes were analysed.

The study brings out how the design of agri-environmental policies against the background of multifunctionality differs from the individual treatment of the various environmental effects of agriculture. Because of the joint production process, the levels of different multifunctional outputs are linked to each other. Hence, the regulation of one environmental effect necessarily influences the other environmental effects and agricultural production, as well as other dimensions of multifunctionality. These interactions need to be accounted for when designing policies inducive to multifunctionality.

It was shown that the optimal policy with respect to multifunctional agriculture under heterogeneous land quality is to use the combination of a differentiated fertilizer tax and a differentiated buffer strip subsidy. The requirement for the use of differentiated instruments arises from the fact that the non-commodity outputs indirectly depend on the heterogeneous land quality through the size of the buffer strips and the amount of fertilizer used. Thus, the first-best solution requires that policy instruments vary over land quality and crop because non-commodity outputs do so. The social welfare difference between the first-best differentiated instruments and the second-best uniform instruments is FIM 64 (10.8 €) per hectare in the case of semi-uniform instruments (crop-specific but uniform with respect to land quality) and FIM 116 (19.5 €) per hectare in the case of fully uniform instruments. Regarding farm income support measures, the results show that pure acreage subsidy and pure producer price support perform poorly in promoting the environmental elements of multifunctional agriculture. However, the performance of these income support measures could be greatly improved by incorporating some environmental cross-compliance mechanisms into them.

To sum up, the combination of differentiated policy instruments is needed to secure the production of the optimal bundle of multifunctional outputs under heterogeneous conditions.

Index words: Agrobiodiversity, buffer strip, heterogeneity, landscape mosaic, land quality, nutrient runoffs

Ympäristöllinen monivaikutteisuus: Taloudellinen analyysi ja merkitys politiikan suunnittelulle

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Tiivistelmä

Käsitteellä monivaikutteinen maatalous viitataan siihen, että maatalous ruoan- ja kuiduntuotannon lisäksi tuottaa muitakin yhteiskunnan hyvinvointiin vaikuttavia maaseutu- ja ympäristöhyödykkeitä. Tärkeimpiä monivaikutteisuuden ulottuvuuksia ovat ympäristön laatu, elintarvikkeiden huoltovarmuus ja maaseudun sosioekonominen elinvoimaisuus. Määritelmällisesti monivaikutteisuustuotosten tulisi syntyä yhteistuotosprosessissa varsinaisen tuotannon yhteydessä ja olla luonteeltaan selvästi ulkoisvaikutuksia ja julkishyödykkeitä.

Monivaikutteisuus tarjoaa haastavan ongelman politiikan suunnitteluun ja toimeenpanoon. Etsittäessä optimaalista monivaikutteisuustuotosten kokonaisuutta on pystyttävä tarkastelemaan samanaikaisesti sekä varsinaista maataloustuotantoa että monivaikutteisuuden muita ulottuvuuksia. Tarkastelua vaikeuttavat vielä ulottuvuuksien riippuvuus toisistaan yhteistuotosprosessin kautta, maataloudelle tyypilliset heterogeeniset tuotanto-olosuhteet ja toimivien markkinoiden puuttuminen useilta monivaikutteisuustuotoksilta.

Tämän tutkimuksen tavoitteena on lisätä tietoa monivaikutteisuuden merkityksestä maatalouspolitiikan ja maatalouden ympäristöpolitiikan suunnittelussa. Keskeisenä tutkimusongelmana on erilaisten politiikkatoimenpiteiden kyky saavuttaa yhteiskunnallisesti optimaalinen maatalouden monivaikutteisuus heterogeenisissä olosuhteissa. Tutkimus on rajattu monivaikutteisuuden ympäristölliseen ulottuvuuteen ja siinä analysoidaan kahden viljelykasvin ja kolmen ympäristöhyödykkeen yhteistuotosprosessia peltoekosysteemissä. Analysoitavat ympäristöhyödykkeet ovat agrobiodiversiteetti, maaseutumaiseman monimuotoisuus ja ravinnepäästöt.

Tutkimuksessa on ensimmäisen kerran kehitetty sekä analyyttinen että empiirinen malli, joka tarjoaa integroidun viitekehyksen monivaikutteisuuden ulottuvuuksien tarkastelulle. Aluksi johdetaan edustavan viljelijän yksityinen optimi ja verrataan sitä yhteiskunnalliseen optimiin, jossa ympäristöhyödykkeitä arvostetaan niiden yhteiskunnallisten rajahaittojen ja rajahyötyjen mukaisesti. Seuraavaksi analysoidaan kolmentyyppisiä politiikkatoimenpiteitä: (i) first-best eli lohko- ja kasvikohtaisesti erilaistetut lannoitevero ja suojakaistatuki, (ii) second-best eli osittain (vain kasvikohtaisesti) erilaistetut tai kokonaan erilaistamattomat politiikkatoimenpiteet ja (iii) käytännön politiikkatoimenpiteet eli perinteiset maatalouspolitiikan tulotukimuodot kuten hintatuki ja hehtaarituki sekä tulotuet, joihin on liitetty ympäristökriteerejä tulotuen saannin

ehdoiksi. Empiiriset tulokset saadaan kalibroimalla analyyttinen malli suomalaiseen havaintoaineistoon.

Tutkimus tuo esiin sen, kuinka maatalouden ympäristöpolitiikan suunnittelu monivaikutteisuuden viitekehyksestä käsin poikkeaa maatalouden ympäristöasioiden yksittäisestä tarkastelusta. Yhteistuotosprosessin vuoksi eri monivaikutteisuustuotosten tuotannon taso on kytköksissä toisiinsa. Tämän seurauksena yhden ympäristöhyödyn tai -haitan sääntely aiheuttaa väistämättä muutoksia niin muissa ympäristöhyödykkeissä ja itse maataloustuotteiden tuotannossa kuin muissa monivaikutteisuuden ulottuvuuksissakin. Yhteiskunnallisesti optimaalisen monivaikutteisuuspolitiikan lähtökohdaksi tulevatkin räätälöidyt politiikkatoimenpideyhdistelmät, joissa toimenpiteiden taso asetetaan koordinoidusti.

Tutkimuksen tuloksena on myös, että heterogeeniset olosuhteet vaativat heterogeenisen sääntelyn. Kun viljelysmaan laatu vaihtelee siten, että se vaikuttaa sekä viljelykasvien että ympäristöhyödykkeiden tuotantoon, yhteiskunnan näkökulmasta optimaalinen monivaikutteisuuspolitiikka on erilaistaa politiikkatoimenpiteiden taso heijastamaan näitä heterogeenisiä olosuhteita. Käytännössä tämä merkitsee lohko- ja kasvikohtaisesti erilaistettuja toimenpiteitä. Mikäli politiikkatoimenpiteet on erilaistettu vain osittain tai ei lainkaan, yhteiskunnallisesti optimaalista monivaikutteisuustuotosten määrää ei saavuteta. Tutkimuksen empiirisessä aineistossa tämä yhteiskunnan hyvinvointitappio epätarkempien ohjauskeinojen käyttämisestä oli 64 mk/ha (10,76 €/ha) kasvikohtaisesti erilaistetuilla toimenpiteillä ja 116 mk/ha (19,51€/ha) kokonaan erilaistamattomilla toimenpiteillä. Epätarkemmista toimenpiteistä koituva hyvinvointitappio on kuitenkin suhteutettava erilaistettujen politiikkatoimien vaatimiin suuriin hallinnointikustannuksiin.

Lisäksi tutkimus osoittaa, että sekä puhdas hintatuki että hehtaarituki kuten CAP-kompensaatiotuki eivät edistä monivaikutteisuuden ympäristöllistä ulottuvuutta. Näiden toimenpiteiden kykyä ottaa monivaikutteisuus huomioon ja edistää sitä voidaan kuitenkin merkittävästi parantaa liittämällä niihin ympäristöehtoja.

Asiasanat: Biologinen monimuotoisuus, heterogeenisyys, maaseutumaisema, ravinnepäästöt, suojakaista, viljelysmaan laatu

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1 Introduction

1.1 Background

For some time it has been recognised that, besides the production of food and fibre, agriculture has other functions through which it contributes to social welfare. Jointly with the production of agricultural commodities, so-called non-commodity outputs arise. These include rural and environmental amenities, rural settlement and employment, and national food security. Sometimes also cultural and historic heritage values, food safety, and farm animal welfare are mentioned in this context. A new slogan, multifunctionality, has emerged in the international policy debate to capture these features of agricultural production.

While the economic significance of agriculture has decreased for some time in a number of countries, income growth has resulted in a growing demand for many of the non-commodity outputs. Through domestic agricultural policies governments try to ensure that the provision of these outputs corresponds to that demanded by society. Thus, even though multifunctionality as a policy term is new, as an implicit concept in the context of domestic agricultural policy it is not entirely new since some countries have already taken into account selected non-commodity outputs of agriculture in their policy-making.

Multifunctionality is important from the domestic policy perspective, but it is the implications of further liberalised agricultural trade on the multifunctional character of agriculture that have raised this issue to the forefront in the international debate. Some countries fear that further reductions in and constraints on domestic support would reduce the ability of governments to pursue their domestic non-commodity objectives, whereas other countries consider that multifunctionality is being used as a pretext for maintaining high levels of production-related support. Hence, the concept of multifunctionality and its use as a basis for concrete policy interventions has raised conflicting views among the WTO members.

The most commonly cited elements of multifunctionality – environment, food security, and the viability of rural areas – were listed as legitimate non-trade concerns in the draft text on agriculture during the WTO Ministerial Meeting in Seattle in 1999. The non-trade concerns related to agriculture can be defined as domestic policy objectives that countries perceive to be threatened by the further liberalisation of agricultural trade (Burrell 2001). However, due to its controversial nature the term multifunctionality itself was not mentioned in the draft text of the WTO Ministerial Meeting, and thus it did not reach a formal status.

It could be argued that the elements of multifunctionality are already covered through the agreed list of non-trade concerns and that the policy relevance of multifunctionality is in that sense vague. However, although multifunctionality and non-trade concerns overlap, there is a fundamental difference between these concepts that has important implications for policy design. Whereas multifunctionality provides an integrated framework for the simultaneous consideration of multiple commodity and non-commodity outputs, non-trade concerns are dealt with as separate issues. That is, each non-trade concern is separately linked to commodity production, but tradeoffs and complementarities between alternative non-commodity outputs are not explicitly recognised.

Clearly, multifunctionality constitutes a complex problem from the perspective of policy design and implementation. Finding out the socially optimal bundle of multiple commodity and non-commodity outputs involves the identification of the important outputs as well as their relative significance, which in itself is a challenging task. Moreover, policies promoting multifunctional agriculture must address simultaneously several outputs, commodity and non-commodity, which have tradeoffs and complementarities in their supply. Even if some non-commodity outputs of agriculture have in cases been taken into account in national policy-making, they have been addressed indirectly through commodity-related interventions. Now they are to be addressed directly. All this is further complicated by the fact that the heterogeneous conditions under which agriculture operates bring into being a spatial dimension in both the supply of and demand for non-commodity outputs. There are spatial differences in productivity and, hence, in production costs of commodity and non-commodity outputs on the supply side, and spatial valuation differences on the demand side. Finally, there is the practical problem that the information requirements and related transaction costs for designing and implementing spatially differentiated interventions in order to maximise social welfare from optimal bundles of commodity and non-commodity outputs may be quite extensive, wherefore governments may be obligated to look for less effective solutions which are less information-intensive but which distort production decisions and thus trade.

So far, the economic analysis of policy design for multifunctionality has mainly been conceptual. OECD (2001a) provides some preliminary policy guidance that is based on the working definition of multifunctionality and on the conceptual framework developed by the OECD. However, owing to the conceptual nature of the analysis, the guidance for policy design remains quite general; for example, whether policy intervention is warranted or not and what kind of policy interventions (coupled with or decoupled from commodity production) would most likely be efficient. Thus, it could be argued that due to the conceptual nature of the existing studies, the economic analysis of policy design for multifunctionality has not yet been rigorously conducted. A major problem in the previous literature is that the commodity and non-commodity outputs have not been rooted in an integrated analytical and empirical framework in which the multiple commodity and non-commodity outputs are considered accurately and simultaneously.

1.2 Objectives and outline of the study

The objective of this study is to contribute to the understanding of the implications of multifunctionality for effective agri-environmental policy design. The main research question addressed is how various types of policy interventions perform in achieving the optimal bundle of multifunctional outputs under heterogeneous conditions.

The rigorous treatment of jointness between commodity and non-commodity outputs and the related implications for policy design under heterogeneous conditions requires that input use and land use are endogenous in the analysis so that farmers' responses to policy interventions in both intensive and extensive margins can be determined. Lichtenberg (1989, 2000) provides an excellent core for examining farmers' input use and land allocation choices under heterogeneous conditions (heterogeneous land quality). However, his model does not have a spatial structure, which plays an important role in the analysis of such non-commodity outputs as landscape diversity and agrobiodiversity. The present study covers all three aspects: endogenous input use and land allocation, heterogeneity, and spatiality.

Figure 1 illustrates the contents of the study. The study focuses on two commodity outputs and three environmental non-commodity outputs of multifunctional agriculture. These are considered simultaneously, taking into account the special features of jointness and heterogeneity in the production, and externality and public good aspects in demand. In order to achieve the optimal production of these multifunctional outputs, various policy solutions are examined.

The starting point for the analysis is the conceptual framework on multifunctionality developed by the OECD (2001a). However, the present study attempts to move from concepts to a formal analysis and from there further to empirical applications. This is done by developing an integrated analytical and empirical framework that provides a sound basis for policy design and evaluation.

Of the non-commodity outputs of multifunctional agriculture, the scope of the present study is restricted to the environmental dimension of multifunctionality. There are several reasons for this. First, the environmental dimension is the

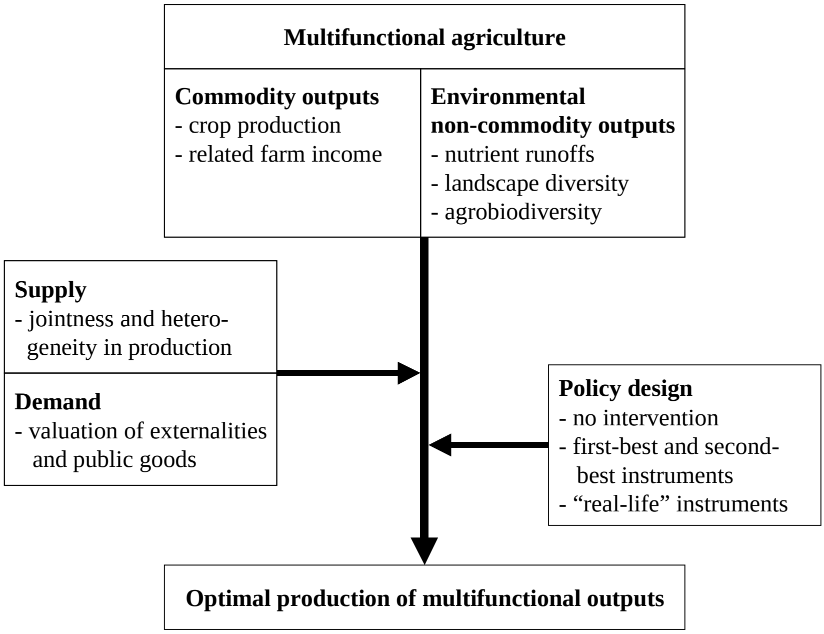

Figure 1. Components of the study.

least controversial one in the international debate. Second, the joint production process is apparent in this dimension. Third, the environmental non-commodity outputs clearly possess the characteristics of externalities or public goods, some of them being in fact pure public goods. The analytical sections of the present study are generally applicable, but the empirical parameters of the parametric model apply to Finland only.

The study is structured as follows. Chapter 1 of the study (introduction) continues with a short presentation of the key concepts used in the study. It also describes the context for multifunctional agriculture in Finland. Chapter 2 (literature review) discusses the supply, demand and policy aspects that are specific to multifunctional agriculture. Chapters 3, 4, and 5 constitute the core of the study. They examine the optimal provision of multifunctional outputs without government intervention (Chapter 3), the use of differentiated (first-best) and uniform (second-best) policy instruments for promoting multifunctional agriculture (Chapter 4), as well as some "real-life" policy instruments (Chapter 5). All these are investigated both analytically and by means of an empirical application with Finnish data. Chapter 6 concludes the study and discusses its main findings, policy implications, and limitations.

1.3 Core concepts

The core concepts and terms used in the present study are defined in this section. The aim is to provide a quick overview rather than a comprehensive discussion, since some of the concepts presented here will be further elaborated and discussed in the literature review in Chapter 2.

In the present study the terms ***multifunctionality, multifunctional agriculture*** and ***the multifunctional character of agriculture*** are used interchangeably. As there is no universally accepted definition for the concept of multifunctionality, a "working definition" provided by the OECD (2001a) is adopted. According to this definition, the fundamentals of multifunctionality are:

(i) the existence of multiple commodity and non-commodity outputs that are jointly produced

(ii) the fact that some of the non-commodity outputs exhibit the characteristics of externalities or public goods

Hence, multifunctional agriculture may be defined as an economic activity which, besides its primary function of producing agricultural commodities, affects social welfare by producing multiple positive or negative non-commodity outputs jointly with the commodity production. Thus, in economic terms, multifunctional agriculture produces jointly private goods, public goods, and positive or negative externalities.

It is worth noting that agriculture is by no means the only economic activity with multifunctional characteristics. For example, forestry provides several non-commodity outputs jointly with timber production. However, in the present study the term multifunctionality refers only to the joint production of non-commodity outputs with agricultural commodities. The significance of forestry to Finnish farms and the significance of farmers as forest owners in Finland nevertheless attach an interesting feature to the multifunctional character of Finnish agriculture. Finnish farmers provide non-commodity outputs such as landscape diversity, biodiversity, and viability of rural areas both through multifunctional agriculture and multiple-use forestry. Farmers' management decisions with respect to agricultural land and production practices, but equally those with respect to forested land and timber harvesting, play a crucial role in the supply of several non-commodity outputs in rural areas. Consequently, the distinction between the non-agricultural provision of non-commodity outputs and that of their agricultural provision is somewhat vague because the same management unit shapes the provision of non-commodity outputs from both agriculture and forestry.

Commodity outputs refer to agricultural commodities that are private goods. These include crops, farm animal products, fibres, energy plants, and so on. ***Non-commodity outputs***, in turn, refer to non-market goods that arise as a side effect of the commodity production, such as landscape and environmental amenities.

Joint production or ***jointness*** refers to a situation where two or more outputs are produced interdependently so that a change in the supply of one output affects the levels of the other outputs. According to the OECD (2001a), three frequently distinguished causes for jointness are

(i) technical and biological interdependencies in the production process
(ii) non-allocable inputs
(iii) allocable inputs that are fixed at the firm level

One example of a technical interdependence between commodity and non-commodity outputs is fertilizer use, which results in both increased yields and increased nutrient runoffs. The joint production of milk and manure, in turn, provides an example of a non-allocable input (cow). An example of jointness due to allocable inputs that are fixed at the firm level in the short run is the allocation of agricultural land between commodity production and wildlife habitat, such as conservation headlands.

According to Baumol and Oates (1988: 17-18), there are two conditions for an ***externality***. First, "an externality is present whenever some individual's utility or production relationships include real variables, whose values are chosen by others without particular attention to the effects on this individual's welfare". Second, "the decision maker whose activity affects others' utility levels or enters their production function, does not receive (pay) in compensation for this activity an amount equal in value to the resulting benefits (or costs) to others". Thus, in brief, an externality can be defined as an uncompensated effect on a utility function or production set. The eutrophication of surface waters due to nutrient runoffs is an example of a negative externality produced by agriculture.

A ***pure public good*** possesses the following characteristics: it is non-rival in consumption and yields benefits that are non-excludable (Callan and Thomas 1996). Non-rivalry means that one agent's consumption of the good does not preclude that of the others. In other words, there is a zero marginal cost for an additional consumer of the good (Stiglitz 1988). Non-excludability means that it is impossible or prohibitively costly to exclude agents from consuming the good. Thus, because of non-rivalry it is not desirable and because of non-excludability it is not feasible to ration the use of the public good (Stiglitz 1988).

The non-use values of landscape and agrobiodiversity can be regarded as examples of pure public goods.

The ***environmental dimension of multifunctionality*** or ***environmental multifunctionality*** refers to the joint production of commodities with environmental non-commodity outputs. The latter include positive non-commodity outputs, such as landscape diversity and agrobiodiversity, but also negative ones, such as impairment of the groundwater and surface water quality due to nutrient and pesticide leaching and runoffs, as well as loss of wildlife due to the use of chemicals and fragmentation and loss of habitats.

The environmental non-commodity outputs, however, may also have indirect effects on the other dimensions of multifunctionality. For example, the attractiveness of rural areas for both the rural and urban population is affected by environmental quality and by landscape amenities (OECD 2001a). Through the natural resource base and the productive capacity of agriculture, environmental outputs, such as erosion and agrobiodiversity, may also affect food security as long as domestic production is regarded as an important part of this.

According to the Convention on Biological Diversity (UNEP 1992), ***biological diversity*** or ***biodiversity*** is the variability among all living organisms from all sources, including terrestrial, marine and other aquatic ecosystems, and the ecological complexes of which they are a part. ***Agrobiodiversity*** is that part of biodiversity which relates to agriculture and agro-ecosystems. Like biodiversity, this can also be described at three fundamental levels: the diversity of ecosystems, species, and genes. Qualset et al. (1995) define agrobiodiversity to include all crops and livestock and their wild relatives, as well as all the interacting species of pollinators, symbionts, parasites, predators, and competitors. Beyond its role in the production of food and fibre, agrobiodiversity has multiple functions in agro-ecosystems. These include the recycling of nutrients, the regulation of hydrological processes, the control of microclimate, the regulation of undesirable organisms, and the detoxification of noxious chemicals (Altieri and Nicholls 1999). Swift and Anderson (1994) divide the biotic components of agro-ecosystems into three types: productive biota such as crops and livestock, resource biota that increase the productivity of the agro-ecosystem, such as pollinators and soil biota, and, finally, destructive biota such as weeds, pests, and pathogens.

Gliessman (2000) defines an ***ecosystem*** to be a functional system of complementary relations between living organisms and their environment, delimited by arbitrarily chosen boundaries, which in space and time appear to maintain a steady, yet dynamic equilibrium. Whereas the structure of an ecosystem refers to its parts and their relationships, its function refers to the dynamic processes occurring within the ecosystem. An ***agroecosystem*** can be defined as a site of agricultural production that is understood as an ecosystem, for example, a group of farms in the context of a watershed, an individual farm, or a farm field (Gliessman 2000). A fundamental feature of agroecosystems, compared to natural ecosystems, is the human intervention that usually aims to reduce species diversity in order to obtain the largest possible yield of the cultivated crops (Swift and Anderson 1994). Swift and Anderson (1994) subdivide a field ecosystem into the following components: (i) the plant subsystem, including the cultivated crop, weeds, and legumes and their obligatory pathogens and symbionts, (ii) the herbivore subsystem, including farm animals, predators, parasites, and parasitoids, and (iii) the soil or decomposer subsystem, including soil organic matter, soil micro-organisms such as bacteria, fungi, and algae, and micro-, meso-, macro- and megafauna (earthworms). The functions of a field ecosystem refer to processes such as the flow of energy and the cycling of nutrients (biogeochemical cycles), such as carbon and nitrogen (Swift and Anderson 1994).

According to the OECD (2001b), ***agricultural landscapes*** are the visible outcomes that result from the interaction between commodity production, natural resources, and the environment. They include amenity, heritage, cultural, aesthetic, and other societal values. Three essentials of agricultural landscapes are their (i) structure or appearance, for instance, flora, fauna, habitats, ecosystems, crops, hedges, and farm buildings, (ii) cultural, environmental, and economic functions, and (iii) value, that is, society's valuation of landscape.

The structure of the spatial mosaic of a landscape and its effects on ecological systems, patterns, and processes is the focus of ***landscape ecology*** (Wiens 1995). The ***landscape mosaic*** can be described with the help of three types of spatial elements: patches, corridors, and background matrices (Forman 1995). In general terms, the spatial structure of the landscape is associated with the composition (number and occurrence) and configuration (distribution and spatial character) of different landscape elements (Eiden et al. 2001).

Buffer strips and ***field boundaries*** are semi-natural habitats adjacent to the crop. In agricultural landscapes they constitute linear elements which form a network of corridors through which organisms can move between larger habitat patches. A field boundary can be defined as a strip of semi-natural vegetation bordering an arable field. Field boundaries are important as they comprise the largest area of semi-natural vegetation in modern arable landscapes and provide food, shelter, nesting, and overwintering sites for most of farmland wildlife (Kleijn 1997). A buffer strip is a managed, uncultivated area that is covered by perennial vegetation and located between the arable land and a water body.

Buffer strips serve both ecological and environmental purposes as they promote agrobiodiversity and protect surface waters from nutrient and pesticide runoffs. Important factors affecting the botanical diversity of field boundaries and buffer strips include the nutrient and herbicide load from the adjacent cropland, disturbance by farming operations, mowing and removing of the cuttings, and the width of the boundary or the buffer strip. Low disturbance levels, low agro-chemical load, removing of the cuttings, and sufficient width maximise botanical diversity (see e.g. Kleijn 1997, Kleijn and Snoeijing 1997, Ma et al. 2002, Schippers and Joenje 2002).

According to Gilliam et al. (1997), buffer strips are very effective in the removal of sediment-associated nitrogen from surface runoff and nitrate from subsurface flows, and removals of 50–90% have been common. However, the effectiveness of buffer strips in removing nutrients from surface water and groundwater depends highly on hydrology. For example, surface flows should occur as a sheet flow rather than as focused flows, and groundwater should move at a slow speed through the buffer in order for nitrates to be effectively removed (Correll 1997). According to Hill (1996), vegetation uptake and microbial denitrification are the two major mechanisms in buffer strips for removing nitrates from subsurface water, but the relative importance of these two processes is uncertain. Moreover, as pointed out by Gilliam et al. (1997), the increased denitrification in buffer strip areas may trade water pollution for atmospheric pollution due to the increased generation of N_2O. It is also important to note that buffer strips only reduce the surface runoff of nutrients but not runoff through drainage pipes. In Finnish experiments, 50–75% of the total nitrogen loss from cultivated fields occurred through drainage pipes (see e.g. Turtola and Jaakkola 1987, Turtola and Puustinen 1998).

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. ***Runoff*** refers to nutrient transportation over the soil surface by rainwater and melting snow, whereas ***leaching*** refers to the transportation of nutrients through the soil by percolating rain and melting snow (Ribaudo et al. 1999). Nitrogen, in the form of nitrate, is easily soluble and very mobile in the environment, but phosphorus is relatively immobile and may build up in the soil over time. Nitrogen is transported from fields to water bodies through both surface runoff and leaching, or through drainage. Phosphorus is transported from fields to water bodies in particulate form and in dissolved form through surface runoff (Hanley 1990, NRC 1993, Ribaudo et al. 1999).

Point source pollution refers to discharges at a specific location through a pipe, out-fall, or ditch. ***Nonpoint source pollution*** (NPSP), or dispersed or diffused pollution refers to pollution that affects waters in a more diffuse way and is difficult to trace back to a precise source. Nutrient runoff from agriculture is typical nonpoint source pollution, since the runoff does not emanate from a single point except in the case of drainage but leaves the field in so many places that an accurate monitoring of each source would be prohibitively expensive (Ribaudo et al. 1999).

1.4 The context for multifunctionality in Finland

When discussing a topic such as multifunctionality which is composed of several elements, it is important to keep in mind the topic in its entirety to maintain an appropriate perspective on individual issues. Therefore, this section outlines some major socio-economic and environmental features of multifunctional agriculture in Finland[1]. The aim of the section is to provide some background on the substance for the analysis and to illustrate the relevance of the selected commodity and non-commodity outputs in a wider context. Thus, this description draws from the specific case of Finland to provide a context for the empirical applications in Chapters 3, 4, and 5. The data concern mainly the year 1999, as do the empirical data in Chapters 3, 4, and 5.

Finland is one of the world's northernmost agricultural countries. Agricultural land covers only 8% of the surface area of Finland, but Finland is the EU Member State with the largest percentage (98.5%) of rural areas. Depending on the definition of countryside, between 1.2 and 1.6 million Finns, or 23 to 32% of the population, live in rural areas. (MAF 2001a).

Finnish agriculture is mainly based on relatively small, privately owned family farms. A notable feature of Finnish farms is that 95% of all active farms own forests; in fact, 34% of Finnish forests are owned by farmers. (MAF 2001a).

In 1999 the share of agriculture in GDP was 1.2%, but the importance of the total food chain in the national economy is much higher. The share of agricultural support in the gross return on agriculture was about 40% in 1999. The EU contributed a little over a third of this, while the rest was covered by national financing. In 1999 the share of agriculture in the employed labour force was about 5%. (MTTL 2000).

The significance of agriculture in the Finnish economy has been decreasing and production growth has been much slower than in the other sectors of the economy (MTTL 2000). The number of people living in rural areas and gaining their livelihood from agriculture has been shrinking fast (MAF 2001a). The number of active farms has fallen from 129,000 in 1990 to 82,000 in 1999. At the same time, the average farm size has been on the increase. During the EU membership, that is, since 1995, the production structure has changed rapidly as the share of animal husbandry farms has fallen, while the share of crop farms has increased. In 1999, 43% of active farms cultivated arable crops; barley accounted for about 30% and wheat about 5% of the total cultivated area. (MTTL 2000).

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[1] For the assessment of the social costs and benefits of multifunctional agriculture in Finland see Yrjölä and Kola (2001).

All in all, agriculture remains the most important economic activity in rural areas, even though both the number of farms and the number of people they employ are declining (MAF 2001a). However, the contribution of agriculture to regional economies and employment varies between regions. In 1999 the share of agriculture in GDP and employment was the highest in South Ostrobothnia (7–8% in GDP and 12% in employment) and the lowest in Uusimaa (0.2% in GDP and 0.8% in employment). (MTTL 2001).

Nutrient runoffs and the resulting eutrophication of surface waters can be regarded as a major negative externality of Finnish agriculture. Agriculture is the main source of both nitrogen (43%) and phosphorus (62%) runoffs into surface waters (Valpasvuo-Jaatinen et al. 1997). Hence, one of the major objectives of Finland in the application of the European Union's agri-environmental regulation EEC 2078/92 has been the reduction of nutrient runoffs. The long-term effects of the agri-environmental programme for 1995–99 (The General Agricultural Environment Protection Scheme) have been expected to amount to a 20% to 40% reduction of both nitrogen and phosphorus runoffs. In addition to this agri-environmental programme, the Finnish Government has issued a Resolution on water protection targets to 2005. The main goals of the Resolution are the reduction and prevention of eutrophication. Nutrient runoffs from agriculture should be reduced by 50% (nitrogen from 30,000 to 15,000 tons per year and phosphorus from 3,000 to 1,500 tons per year) from 1993 levels by the year 2005. Moreover, in the new agri-environmental programme for 2000–2006 improvement in the quality of surface water through reductions in nutrient runoffs is still considered one of the most important policy objectives.

Although soil erosion is not a significant problem, the mechanisation of agriculture and the use of heavy field machinery on wet soils has resulted in soil compaction and related soil erosion in some areas of Finland, which has increased phosphorus and sediment runoffs into surface waters.

Due to the climatic conditions, that is, the hostility of the cold climate to many pests, pesticide use per hectare is very low in Finland. Pesticide runoffs in Finland have been estimated to vary between 0.1% to 1% of total use, depending on the pesticide used and on weather conditions (Laitinen et al. 1996). Pesticide contents in water bodies occasionally exceed the limits set for drinking water. Pesticide residues in foodstuffs have not been a problem in Finland, since the amount of residues has consistently been lower than the limits set for residues (Miettinen et al. 1997).

Between 1990 and 1999, greenhouse gas emissions from agriculture decreased from 10.2 to 7.6 million tons in carbon dioxide equivalent. In 1999 agriculture accounted for 10% of total emissions in Finland. The share of agriculture has been estimated to be 2% of carbon dioxide, 40% of methane, and 50% of nitrous oxide emissions (MAF 2001b). Of ammonia emissions in Finland agriculture accounts for 90%, wherefore reducing the amount of ammonia released from manure storing and spreading has been one of the goals of the agri-environmental programme (Grönroos et al. 1998).

Over the past 50 years agricultural landscapes in Finland have become increasingly homogeneous due to the structural change and the mechanisation, rationalisation, and intensification of production. Rationalisation through subsurface drainage and the removal of small-scale elements (trees, ponds, hedges) and forest islands has resulted in more geometric field parcels with inevitably less value for landscape diversity and agrobiodiversity. The decline in the number of linear landscape elements (ditch banks and arable field borders adjacent to non-arable land) is largely explained by the replacement of open field ditches by subsurface drainage. At the national level, subsurface drainage has replaced, on the average, 500 m/ha of open ditches (Ruuska and Helenius 1996).

The maintenance of diverse agricultural landscapes is of particular concern in Finland, since only 8% of the land area is used for agriculture. Although forestry also provides several non-commodity outputs in rural areas, not all of them are substitutes for those of agriculture. This is especially the case with respect to biodiversity and landscape diversity. Most of the threatened species in Finland live either in forest habitats or in agricultural habitats, and neither of them are substitutes for each other. Also in terms of landscape diversity, forestry may be a poor substitute for the landscape provision of agriculture. For example, the results of a survey by Hietala-Koivu et al. (1999) show that afforestation (refers to the establishment of forest cover to land that was not previously forested, e.g. agricultural land) and land abandonment are considered the most important factors that decrease the scenic value of landscapes. This is confirmed by the study of Tahvanainen et al. (1996), who conducted an interview survey on the effect of gradual afforestation on the scenic beauty of 32 different rural landscapes in Finland. The scenic beauty was considered to decrease along with the increasing intensity of afforestation. However, moderate afforestation could have a positive effect on scenic beauty. The more attractive the original landscape was the greater the negative effect of afforestation was found to be.

Traditional biotopes, such as dry meadows and pastures, have the greatest number of species diversity in Finnish landscapes. The botanical diversity of these habitats has benefited from grazing and mowing (Pitkänen and Tiainen 2001). Specialisation in crop farming and the associated decrease in animal husbandry and grazing animals has resulted in such a dramatic decline in the total area of meadows and pastures that less than 20,000 hectares of valuable agricultural heritage environments remain (Heritage Landscapes Working Group 2000). According to Pykälä and Alanen (1996), species living in heritage landscapes constitute 75% of all threatened species in agricultural landscapes. Because of the decline in the area of traditional biotopes, the importance of field boundaries and buffer strips for overall agrobiodiversity has increased.

According to Rassi et al. (1991), one out of 4 bird, 21 insect and 14 vascular species that are endangered have suffered from the indirect effects of herbicides, the disappearance of boundary habitats, and autumn ploughing.

Although the species diversity of agricultural environments has been on the decrease, there have also been some changes in agricultural practices that promote agrobiodiversity. Most of these positive changes have been introduced by the Finnish agri-environmental programme, like buffer strips, the management of field boundaries, limits on fertilizer use, as well as requirements relating to pesticide use and plant cover during winter. In addition to these so-called basic measures of the agri-environmental programme, there are special contracts relating to traditional biotopes, the promotion of biodiversity, and the management of landscapes.

Figure 2 summarises some key data on the environmental and socio-economic significance of Finnish agriculture.

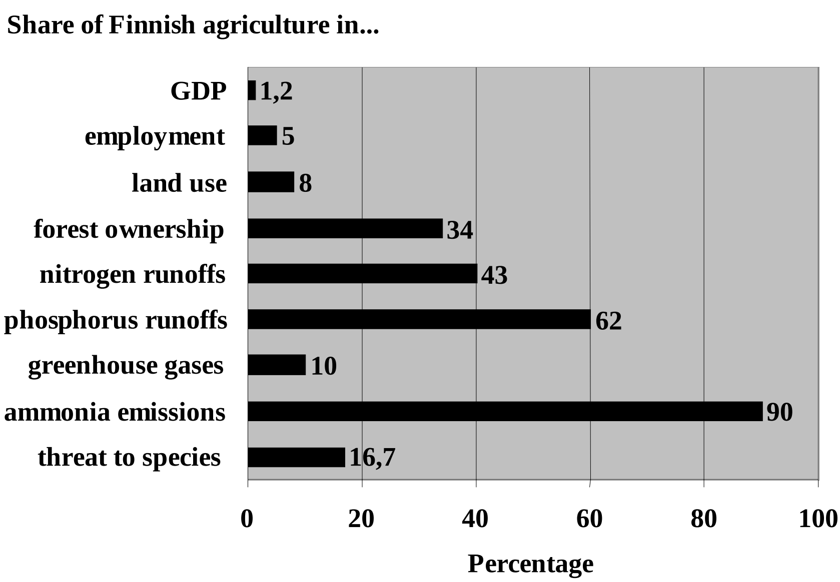

Figure 2. Selected indicators for the significance of Finnish agriculture in 1999 (MTTL 2000, MAF 2001a, Valpasvuo-Jaatinen et al. 1997, Grönroos et al. 1998, and Rassi et al. 1991).

To conclude, the economic significance of agriculture is small from the national perspective, although its local socio-economic role in rural areas is crucial. At the same time, the environmental significance of agriculture is great in many respects. Moreover, since Finnish farmers are also an important group of forest owners in Finland, they have a twin role as custodians of multifunctional agriculture and multiple-use forestry. Therefore, the agricultural and silvicultural decisions and production practices of Finnish farmers are important in shaping the occurrence of several externalities and the provision of many public goods in Finland.

2 Environmental multifunctionality – Literature review

This chapter reviews the literature relating to the environmental dimension of multifunctionality. It comprises three main sections: special features in the supply of, demand for, and policy design for environmental multifunctionality. This structure arises from the conceptual framework provided by the OECD (2001a), which this study makes use of. In this framework, a series of questions related to jointness on the supply side, public good characteristics on the demand side, and the possibility of the non-governmental provision of the non-commodity outputs are posed in order to arrive at appropriate policy guidance for multifunctionality.

As noted by the OECD (2001a), the particularities in the supply and demand of multifunctional outputs are crucial for any discussion on policy implications. On the one hand, if there were no jointness in the production of multifunctional outputs, the non-commodity outputs could be supplied independently of the commodity production. On the other hand, if there were functioning markets for the non-commodity outputs, supply and demand would meet through those markets. In both cases, environmental multifunctionality becomes a non-issue from the policy perspective[2].

The literature review is geared to serve the analysis presented in Chapters 3 to 5. As such, it focuses on aspects that are specific to the environmental dimension of multifunctional agriculture. The examples presented are also selected so as to relate to the non-commodity outputs analysed in Chapters 3 to 5, that is, nutrient runoffs, landscape diversity, and agrobiodiversity.

It should be noted that as an explicit topic of investigation, policy design for multifunctionality has been the subject of very little formal economic analysis. Notable contributions include Peterson et al. (1999), Romstad et al. (2000), Boisvert (2001), Guyomard and Levert (2001), OECD (2001a), and Vatn (2002). There are, nevertheless, a number of studies that shed light on the various individual aspects of multifunctionality. For example, there is a growing economic literature on the policy design for controlling nutrient runoffs.

[2] Naturally, public policy may be needed also in the case of non-joint non-commodity outputs. However, in this case there may be no policy link between promoting the non-commodity outputs and international trade flows (OECD 2001a).

2.1 Special features in supply

As noted in Chapter 1, the existence of multiple commodity and non-commodity outputs that are jointly produced is a fundamental feature of multifunctionality (OECD 2001a). Joint production or jointness is a key feature in the supply side of multifunctionality and it implies that there are interdependencies in production so that a change in the supply of one output affects the levels of the other outputs. Hence, the supply of commodity and non-commodity outputs needs to be analysed within the joint production framework. It is noteworthy that joint production is adopted as the starting point in almost all economic analysis which explicitly examines multifunctionality, perhaps owing to the framework and working definition provided by the OECD (2001a).

Another important feature on the supply side of environmental multifunctionality has to do with spatial differences in supply. That is, the quantity, quality, and composition of multifunctional outputs differ between and within countries due to heterogeneous conditions. These heterogeneous conditions which affect the nature of jointness and the optimal bundle of commodity and non-commodity outputs are termed "site productivity" by the OECD (2001a).

2.1.1 Joint production

According to Shumway et al. (1984), even though technical interdependence is generally regarded as the primary cause of jointness, also allocable fixed (or quasi-fixed) inputs, such as land, may cause jointness. In fact, Shumway et al. argue that jointness caused by allocable inputs is especially typical for agriculture as many farms produce more than one output, the amount of land devoted to each crop can easily be distinguished, and the amount of land is usually fixed in the short run (Shumway et al. 1984).

Following Lau (1972), Shumway et al. (1984) make a distinction between jointness and nonjointness as follows. For technology to be nonjoint in inputs requires that the profit function is additively separable in output prices: $\pi = \sum_{i=1}^{m} p_i \mathbf{G}_i(\mathbf{r} / p_i)$, where $\mathbf{G}_i$ is the individual profit function for the i:th output, p_i is the i:th product price, and $\mathbf{r}$ is the vector of input prices. Now the distinction between jointness and nonjointness is given by the supply response of the i^{th} output to the price of the j^{th} output, that is, for nonjointness $\partial y_i^* / \partial p_j = 0$ and for jointness $\partial y_i^* / \partial p_j \neq 0$ where $(i \neq j)$. Thus, for instance, there is jointness (nonjointness) in the production of barley and wheat if the supply of barley responds (does not respond) to the price of wheat.

Lynne (1988), in his comment to Shumway et al. (1984), proposes a distinction between "jointness in technology" and "jointness in supply" so that jointness in technology refers to technical interdependence and jointness in supply to behavioural interdependence. According to Lynne (1988), jointness, as traditionally represented, occurs only with non-allocable inputs and is synonymous with technical interdependence. However, fixed but allocable inputs may cause behavioural jointness in supply even if outputs are technically independent (nonjoint). In their reply to Lynne (1988), Shumway et al. (1988) refer to Lynne's argument that production functions underlying joint production do not contain allocable inputs. They provide an example of the allocation of inputs for two crops where pesticide, which is a fully allocable input, applied to one crop in one field affects the yield of the other crop in an adjoining field. Thus, Shumway et al. argue that the production functions of these crops are not technically independent. In the subsequent contributions to the role of fixed but allocable inputs as a cause of joint production, Moschini (1989) and Leathers (1991) clarify the discussion through the notion of normal inputs and the cost function approach, respectively.

Beattie and Taylor (1985) define the technical interdependence of two outputs as follows. Technical interdependence between two outputs produced from one allocable input can be viewed as the change of the marginal productivity of an input in the production of one output when the level of the other output changes. Thus, according to Beattie and Taylor (1985), if the multioutput production function is given by $x = g(y_1, y_2)$, y_1 and y_2 are technically complementary if $\partial^2 x / \partial y_1 \partial y_2 \equiv g_{12} < 0$, technically competing if $\partial^2 x / \partial y_1 \partial y_2 \equiv g_{12} > 0$, and technically independent if $\partial^2 x / \partial y_1 \partial y_2 \equiv g_{12} = 0$. That is, technical interdependence is present if one output is increased and that results in the change of the inverse marginal productivity of the input use for another output (Beattie and Taylor 1985).

Beattie and Taylor (1985) further define the economic interdependence of two outputs as follows. Two outputs are economically interdependent if a change in the price of one output affects the supply of the other output. In other words, two outputs y_i and y_j are economically competing if $\partial y_i^* / \partial p_j < 0$, economically complementary if $\partial y_i^* / \partial p_j > 0$, and economically independent if $\partial y_i^* / \partial p_j = 0$ for i, j =1,2 and $(i \neq j)$.

According to Boisvert (2001), the three commonly distinguished causes for joint production (technical interdependencies in the production process, non-allocable inputs, and allocable inputs that are fixed in the short run at the firm level) are also representative for jointness between most commodity and non-commodity outputs. However, these three sources of output interdependence may arise in various combinations and proportions, and it is unlikely for jointness to occur in fixed proportions (Boisvert 2001).

Fertilizer or pesticide use that results in the joint production of commodities and nutrient or pesticide runoffs is one example of a technical interdependence between commodity and non-commodity outputs. Technical interdependencies are the source of many negative environmental externalities of commodity production, such as the nutrient and pesticide runoffs mentioned above, leaching, soil erosion, and greenhouse gas emissions. However, changes in farming technologies and practices may modify the composition of the commodity and non-commodity output bundle. (OECD 2001a).

The joint production of meat and landscape by grazing cattle provides an example of a non-allocable input. In this case, multiple outputs arise from the same input, but they are rarely produced in fixed proportions, and so using different production methods may change the proportions (OECD 2001a).

Land allocation between commodity production and abatement activities such as buffer zones represents an example of jointness which is due to allocable inputs that are fixed at the firm level in the short run. Boisvert 2001 provides an example where agricultural land is simultaneously an allocable and non-allocable input: it is allocable between two commodities but non-allocable between these commodities and landscape amenities. By taxing or subsidising the farmer, Boisvert demonstrates the economic significance of joint production of commodity and non-commodity outputs, regardless of the cause of the jointness. This makes it possible to compare policies aimed specifically at the non-commodity outputs directly with commodity policy.

It has been argued that, in spite of its pervasiveness in agriculture, jointness due to allocable inputs such as land may not be as important as the two other sources of jointness in analysing multifunctionality. The argument rests on the notion that the option to allocate the production of commodities and non-commodities to different parcels of land implies a high degree of output separation and a low degree of jointness. (OECD 2001a). However, even in this case jointness is present in the sense that some of the non-commodity outputs compete with the commodity outputs for a fixed amount of land, and hence land allocated for non-commodity production reduces the land available for commodity production. In other words, there is jointness in supply.

2.1.2 Heterogeneity

Both agricultural productivity and the site productivity of non-commodity outputs show significant heterogeneity due to spatial variation in the natural resource base and natural conditions. Consequently, the same agricultural production practices may produce drastically different bundles of commodity and non-commodity outputs in different areas. Hence, the nature and degree of jointness between commodity outputs and non-commodity outputs vary spatially.

There is spatial variation in the environmental, ecological, and economic attributes of agroecosystems. According to Wossink et al. (2001), spatial variation in the abiotic environment arises from climatic and soil factors and their interaction, and spatial variation in the biotic environment is caused by pests, weeds, diseases, and beneficial organisms. Moreover, there is heterogeneity with regard to a keystone species of agroecosystems – the farmer. Human capital and behavioural characteristics differ between farmers according to factors such as age, education, experience, risk preferences, wealth, debt structure, productive capital, and farm size (Antle and Just 1992). Thus, because of heterogeneity among farmers, the non-commodity output bundle may be different even in two adjoining field parcels which share the same environmental and ecological characteristics. In sum, through the inherent spatial variation of environmental, ecological, and economic characteristics, heterogeneity plays a fundamental role in determining site-specific bundles of commodity and non-commodity outputs.

The farmer (and, through him, the policy-maker) may have some control over certain site-specific environmental and ecological characteristics while other characteristics escape control. De Koijer et al. (1999) classify abiotic and biotic factors based on their influence on crop growth as follows. The potential for crop growth is determined by growth determining factors that are beyond the farmer's control, including site-specific environmental factors, such as light and temperature, and plant intrinsic characteristics. Growth limiting factors, in turn, are abiotic factors such as nutrients and water, which may, if short in supply, reduce crop yield below the potential yield. The farmer has control over these growth limiting factors which may also have detrimental environmental effects. Finally, growth reducing factors, such as pests, diseases, and weeds, reduce the attainable crop yield to the actual yield, but can be controlled through crop protection measures, which may, again, have detrimental environmental effects.

According to Wossink et al. (2001), spatial variation and spatial relations are treated quite superficially in economics. For example, land is typically assumed to be homogeneous in all physical characteristics through regions. This assumption of homogeneous land quality is very restrictive in the analysis of relationships between commodity and non-commodity outputs, as heterogeneous land quality is a pervasive feature of agriculture. Soil quality is part of overall land quality and refers to the capacity of the soil to perform crop production, environmental, and ecological functions. Important soil quality attributes which are influenced by management include soil-depth, organic matter, respiration, texture, bulk density, infiltration, nutrient availability, and retention capacity (Arshad and Martin 2002).

Antle and Just (1992) provide a conceptual framework for the analysis of interactions between agricultural commodity production and the environment. In this framework the fundamental role of the physical and economic heterogeneity of farms is recognised in the determination of the environmental outcomes of commodity production. By modelling the joint distribution of production and pollution, Antle and Just are able to analyse production-pollution trade-offs and the effects of alternative policy instruments on both the intensive and extensive margins. Their analysis clearly demonstrates the need to account for farm heterogeneity in informed policy design and implementation.

2.2 Special features in demand

There are two issues to consider when analysing the demand for multifunctional outputs (OECD 2001a). One issue is that the non-commodity outputs exhibit the characteristics of externalities and public goods. Hence, their demand cannot be directly observed from the markets, and demand and supply may not meet through market transactions. Another issue is that multiple non-commodity outputs are demanded simultaneously. As the outputs may be substitutes for or complements to one another, their simultaneous aggregate demand may differ from the sum of the demands for the individual outputs.

2.2.1 Externalities and public goods

OECD (2001a) notes that there are differences between various externalities and public goods that lead to different policy conclusions. From a policy perspective it is important to consider the characteristics of each non-commodity output, asking the following questions:

(i) Does the non-commodity output exhibit the characteristics of an externality?
 (ii) if yes, does it constitute a market failure?
  (iii) if yes, what kind of public good is affected?
   (iv) thus, what is the scope for government intervention?

In Chapter 1, an externality was defined as an uncompensated effect on a utility function or production set. If commodity production generates effects (costs or benefits) that are outside the market transaction, that is, external to the market, the market fails in the sense that the price of the commodity does not capture these effects. As a result, the market price undervalues (overvalues) commodity production which generates external benefits (costs), and there is a tendency for the supply to fall short for a commodity that generates benefits, whereas a commodity that generates costs is oversupplied. Hence, an environmental market failure occurs when the market fails to reflect the true social costs and benefits of using environmental resources, and market prices of exchanged commodities fail to capture all the environmental costs and benefits associated with a market transaction (Callan and Thomas 1996). Due to such market failures, the price signals from commodity markets are highly unlikely to ensure the provision of the optimal bundle of commodity and non-commodity outputs. Thus, the private and social optima for multifunctional outputs diverge. The objective of the internalisation of externalities is to incorporate the external costs and benefits into the optimisation calculus of economic agents through appropriate policy instruments so that the gap between private and social costs of multifunctional production is bridged.

OECD (2001a) has argued that in the case of multifunctional outputs the relationship between externalities and market failures becomes more complicated. They propose three situations where an externality does not lead to market failure. The first one is jointness in the production of the commodity output and the non-commodity output. It is possible, according to the OECD, that a non-commodity output which creates a positive externality is produced in sufficient amounts to meet the demand of society, in which case there is no market failure. That is, if the social and private costs of producing the positive externality coincide at the market price, there is no market failure despite the presence of an externality, even if social costs may be lower than private costs when the commodity output is below market equilibrium. The second situation relates to jointness in the production of two non-commodity outputs. A decrease in the supply of a positive externality may be associated with a decrease in the supply of a negative externality, which reduces or offsets the market failure. An example would be falling agricultural production, resulting in reduced landscape amenities but also in less eutrophication. The third situation put forward by the OECD is triggered by consumption relationships between the non-commodity outputs: the presence of a negative externality may reduce the demand for the positive externality and, thus, again reduce the market failure. For example, the valuation of the scenic amenities provided by a flowering rape field could be reduced by the knowledge of a loss of species in an adjacent wildlife habitat caused by the agro-chemical load from the rape field. It should be noted that these arguments proposed by OECD (2001a) concerning situations where an externality does not lead to market failure have not won unanimous support.

The example of agro-chemical load resulting in a loss of species in a wildlife habitat illustrates how an environmental externality may affect an environmental resource with public good characteristics. As noted earlier, a pure public good is non-rival in consumption and yields benefits that are non-excludable.

Table 1. Types of public goods (based on OECD 2001a).

Type	Description	Examples	Scope for government intervention
Pure public goods	Non-excludable, non-rival	Non-use value of landscape, wildlife habitat, agrobiodiversity	Important role
Local pure public goods	Non-excludable, non-rival, benefits restricted to small jurisdictions	Use value of landscape by residents	Important/limited role (if sufficient local voluntary provision)
Open access resources	Non-excludable, non-rival, congestible	Use value of landscape by visitors	Important role
Common property resources	Excludable to outsiders, rival	Use value of wildlife habitat and agrobiodiversity	Limited role (if community establishes rules)
Excludable and non-rival goods	Excludable, non-rival	Non-use value of wildlife habitat and agrobiodiversity if some institutional arrangements like environmental trusts could be established	Important/limited role
Club goods	Excludable, congested	Non-use value of natural habitat and biodiversity if some institutional arrangements like environmental trusts could be established	Limited role

However, according to the OECD (2001a), a more detailed classification of public goods (see Table 1) is required in order to arrive at best policy choices. Pure public goods and open access resources are difficult to provide optimally without government intervention, but for other types of public goods the scope for government intervention may be more limited.

In the light of the above discussion, let us briefly examine the non-commodity outputs that are selected for analysis in Chapters 3 to 5. Nutrient runoffs from arable lands represent a negative externality which constitutes a market failure as the private and social costs of commodity production diverge and, due to impairments in surface water quality, the value of the public good decreases. Land allocation between alternative crops and linear landscape elements, such as field boundaries and buffer strips, determine the landscape mosaic, and thus the aesthetic and ecological values of landscape, which are public goods. The effects of land allocation on these values are external to the commodity markets, resulting in a divergence between the private and social optima, and thus they constitute a market failure. The agrochemical load from farming affects wildlife habitats adjacent to arable lands and thus agrobiodiversity. Again, an externality leading to market failure can be identified, and the public good in question is agrobiodiversity. Hence, each non-commodity output in the ensuing analysis is relevant from the government intervention perspective.

When discussing the degree of excludability in the consumption of non-commodity outputs, such as landscape, wildlife habitats, and agrobiodiversity, it is important to note that in Finland there is a common right of access ("everyman's right") to all natural (undeveloped) areas. This practice gives everyone the right to roam freely in the countryside without obtaining permission, no matter who owns or occupies the land. Hence, anyone may walk, ski, cycle or ride freely in the countryside, provided that no harm is caused to property or nature. Moreover, one may pick wild berries, mushrooms, and flowers that do not belong to any protected species. The common right of access is, however, limited in cultivated fields so that in the summer, hikers must go around fields or cross them using only tracks or ditches, but during the winter, fields may be crossed freely by skiers. (Ministry of the Environment 2002). Hence, the right basically reduces the degree of excludability so that, for example, scenic landscapes, wildlife habitats, and agrobiodiversity within agricultural landscapes are non-excludable, and thus the creation of markets or quasi-markets for these non-commodity outputs may be difficult.

2.2.2 Valuation

When government intervention is required to address the externality and public good characteristics of multifunctional outputs, these outputs need to be explicitly valued. This is done in order to estimate the social demand for them and to ensure that the intervention does not shift the bundle of non-commodity outputs in the wrong direction or too far beyond the optimum (Santos 2000).

If the welfare-economic perspective is adopted, the concept of total economic value captures all the values related to non-commodity outputs, that is, all use and non-use motivated values (see e.g. Pearce and Turner 1990, Randall 1991, Holstein 1998). The total economic value of a non-commodity output may thus consist of several components. Table 2 lists types of economic value that may be derived from non-commodity outputs.

The OECD (2001a) suggests that use values may dominate non-use values of landscapes, whereas for wildlife habitats and agrobiodiversity, the opposite may hold.

The assessment of potential multiple trade-offs between alternative non-commodity outputs requires that a basic system of weights is developed (Santos 2000). Economic valuation involves assigning monetary values for non-commodity outputs for which market values do not exist. It can help policy-makers

Table 2. Types of economic value (OECD 1999).

Type of value		Examples
Use values	Use value	E.g. value obtained from consuming (looking at) a scenic landscape
	Option value and quasi-option value	E.g. value obtained from having the possibility to consume a scenic landscape in the future
Non-use values	Existence value	E.g. value obtained from knowing that the scenic landscape exists
	Bequest value	E.g. value obtained from knowing that a scenic landscape is maintained for future generations

to design appropriate policy interventions to achieve a welfare-increasing or even welfare-maximising bundle of commodity and non-commodity outputs (Santos 2000). It is therefore necessary to turn to valuation methods that have been developed to elicit economic value in cases where it cannot be observed from the markets. Such valuation methods can be based either on preferences that are revealed by actual market behaviour in connection with a marketed good that is related to the non-commodity output of interest, or on preferences that are stated in surveys or experiments. Table 3 provides a classification of different valuation methods.

According to Navrud (2000), the direct revealed preference methods – simulated markets, market prices, and replacement costs – are simple to use but ignore behavioural responses to changes in non-commodity outputs. While the indirect revealed preference methods – travel costs, averting expenditure, and hedonic price analysis – are limited to use values, stated preference methods are able to measure the total economic value, including non-use values. The stated preference methods are either direct, such as contingent valuation, or indirect, such as contingent ranking and choice experiments.

Table 3. Types of valuation methods (Navrud 2000).

	Indirect	Direct
Revealed preferences	Travel costs Averting expenditure Hedonic price analysis	Simulated markets Market prices Replacement costs
Stated preferences	Contingent ranking Choice experiments	Contingent valuation

The contingent valuation method has been the most widely used of the stated preference methods, but in recent years, also contingent ranking and choice experiments have become popular (Navrud 2000). There are problems associated with each of the individual valuation methods, but a general discussion on these problems is beyond the scope of the present study. Hence, let us turn to some approaches proposed for the empirical valuation of multiple non-commodity outputs.

A particular valuation problem stems from the nature of environmental multifunctionality. There are multiple outputs that are consumed simultaneously, and these outputs may be substitutes for or complements to one another in consumption. As a result, the social demand for the totality of the multifunctional outputs may not be equal to the simple sum of the demand for the individual elements of multifunctionality. Santos (2000) and Randall (2002) discuss these consumption relationships between non-commodity outputs in their proposals for the empirical valuation of multifunctionality.

According to Santos (2000), the single most important valuation problem related to multifunctionality are the demand interactions between the non-commodity outputs, which may cause severe aggregation problems when summing up values across different non-commodity outputs. Thus, this valuation problem is not due to the jointness between commodity and non-commodity outputs on the supply side, but to demand side interactions. Santos (2000) proposes a rule for the sequential valuation of multiple commodity price levels and multiple non-commodity output levels in order to minimise the individual valuation and summation (IVS) bias, which may lead to wrong policy recommendations (see also Randall 1991). This aggregation bias can also be avoided when the non-commodity outputs are valued jointly, as this automatically takes into account any substitution effects between the non-commodity outputs. However, Santos (2000) also notes that, while a simultaneous valuation of all non-commodity outputs is theoretically preferable, it may be practically impossible, for instance, due to cognitive errors. These cognitive errors have to do with the capacity of the respondents to consider all non-commodity outputs and trade-offs between them in rapid valuation exercises.

According to Randall (2002), "…environmental economists have seldom if ever attempted a task so demanding as valuing the outputs of multifunctional agriculture". There are two reasons why this task should be so demanding. First, the potential "costs" of wrong valuation are high because of the inefficiencies in public goods provision and the possible distortions in the domestic and international commodity markets. Second, obtaining correct values is a considerable undertaking as these values are contextual and detailed and, while they must be estimated on a national or even continental scale (for example, for the European model of agriculture), the potential compensations for farmers are implemented on a farm-by-farm basis. Nevertheless, Randall (2002) proposes two valuation strategies which can be applied in the case of multifunctional agriculture. The first valuation strategy starts with the contingent valuation method to obtain a holistic WTP for multiple non-commodity outputs on a continental scale. This serves as an upper-bound estimate for all local and individual non-commodity outputs which would be estimated by the decomposition of contingent valuation procedures, subjected to convergent validity tests. The second valuation strategy uses choice experiments and the techniques of random utility modeling and conjoint analysis.

Choice experiments belong to the class of stated preference methods which are consistent with the random utility theory in economics and psychology. Consistency with the random utility theory requires that the elicitation method provides information about preference orderings for all the choice options or for their subset. (Louviere 2001) Choice experiments arise from conjoint analysis but are different in the sense that, instead of ranking or rating alternative bundles of attributes, the respondents are asked to choose the best one out of a set of three or more alternatives (Navrud 2000). For further discussion on the choice modelling approach to environmental valuation see, for example, Bennett and Blamey (2001).

2.3 Special features in policy design

Environmental multifunctionality constitutes a multifaceted problem for efficient policy design and implementation for at least the following reasons: (i) the policy instruments or instrument combinations should be able to address several commodity and non-commodity outputs simultaneously, (ii) heterogeneous conditions lead to spatial differences in agricultural productivity and the site productivity of non-commodity outputs, (iii) there are spatial valuation differences on the demand side, (iv) the spatial dimensions of the supply of and demand for non-commodity outputs do not always coincide, and (v) there are trade-offs between spatially differentiated and tailored policy instruments and their information requirements and related administrative costs.

Bearing these multiple problems in mind, the discussion in this section is structured as follows. As there are only few commonly known studies which specifically deal with the formal policy design for multifunctionality, the section starts with a review of these studies. This is followed by a brief review of economic studies on the policy design for selected non-commodity outputs which are central to the present study: nutrient runoffs, landscape diversity, and agrobiodiversity. Finally, the implications of heterogeneity and transaction costs for efficient policy design are discussed.

2.3.1 Policy design for multifunctionality

Peterson et al. (1999) develop a general equilibrium framework for analysing optimal environmental policies under multifunctionality and relate these optimal policies to international trade. Their model is based on aggregate multi-output technology: agricultural and non-agricultural commodities are produced by land and non-land inputs, and two non-commodity outputs (emissions and landscape amenities) are jointly produced by agricultural inputs. Peterson et al. show that the socially optimal policy is to use the combination of a tax on agricultural non-land input and a subsidy on agricultural land input and that the levels of these instruments have to be determined jointly. Moreover, since the emission function for each acre is convex, the optimal subsidy for an acre of land exceeds the amenity value of that acre as the marginal acre of agricultural land decreases the total pollution. In the empirical application Peterson et al. estimate that the optimal subsidy for agricultural land is 50% higher than the amenity value of that land.

Peterson et al. (1999) analyse the implications of optimal environmental policy under multifunctionality for international trade in the cases of a closed, small open, and large open economy. An important distinguishing feature in their analysis compared to traditional literature on trade and the environment (for a survey of this literature see e.g. Ulph 1998) is the combination of environmental policy instruments aimed for multiple externalities. However, their results are conventional in the sense that small economies have no incentive to distort their environmental policies from first-best levels, but large economies have an incentive to manipulate the terms of trade in their favour. In the empirical simulations Peterson et al. show that by using domestic environmental policy the U.S. could increase the world prices of agricultural products by 9% compared to the base case of no environmental policy intervention.

Boisvert (2001) analyses policy design for multifunctionality by using a farm level model. In his model two agricultural commodities are produced with a land input and a purchased input. Whereas landscape amenities (positive externality) are produced by land allocated to both commodities, the environmental residual (negative externality) is associated with the purchased input and land used for only one of the commodities. The production of the environmental residual is convex with respect to the application of the purchased input but decreases if the production of the commodity becomes more land intensive. Landscape amenities are concave with respect to land used in the production of the commodities. Boisvert analyses five alternative cases in order to illustrate the causes and implications of joint production in the multifunctionality context. Among other things, he demonstrates the economic significance of the joint production of commodity and non-commodity outputs, regardless of the cause of the jointness. That is, when the social costs and benefits of non-commodity outputs are explicitly accounted for in the farmer's decision making, joint products which are technically interdependent are revealed to be economically interdependent as well.

Guyomard and Levert (2001) develop a conceptual framework for analysing how traditional farm income support programmes meet the objectives of income support, the promotion of positive externalities, and the reduction of negative externalities, as well as the potential trade distortions of these support policies. Their analytical model contains three equilibrium conditions: (i) the agricultural commodity market where aggregate supply equals aggregate demand, which consists of domestic demand and exports, (ii) the land market where aggregate supply equals aggregate demand, with explicit land trades between farmers, and (iii) an entry/exit condition that permits the endogenous determination of the number of farms. Positive and negative externalities are modelled as follows. It is assumed that positive externalities are related to the number of producers. This assumption is based on Hueth (2000), where the existence of relatively high-cost producers is explained by the desire to sustain rural communities; in other words, society values the production of the high-cost producers beyond its market value. Negative externalities are linked to the intensity of fertilizer and pesticide use.

The farm income support programmes analysed are price support, production-linked direct payments, land-based direct payments, and decoupled direct payments with and without mandatory production. The results are reported with respect to farmers' income, the number of farms, the level of intensification, and commodity exports. The main results from the comparative static analysis are as follows. A decoupled direct payment without mandatory production affects only farmers' income. A decoupled direct payment with mandatory production has a positive effect on the number of farms while the other effects are indeterminate. A land-based direct payment has a positive effect on farmers' income and on exports and reduces intensity, while its effect on the number of farms is indeterminate. A production-linked direct payment increases farmers' income, exports, and intensification, while the effect on the number of farms is indeterminate.

Romstad et al. (2000) analyse a wide variety of policy measures for promoting multifunctionality in a Norwegian setting. The policy measures are evaluated with the help of production theory and resource allocation mechanisms (an extension of traditional principal-agent models), as well as alternative theories of farmers' behaviour, including profit maximisation, utility maximisation, and norms-driven choices. In the evaluation of the policies, consideration is given to trade-offs between the precision and the transaction costs of the policy instruments, and to the marginal costs of public funds. The analysis is conducted in a stepwise manner, starting from jointness and moving to the case where private and public goods are complementary or competing and, finally, to the case where public goods are relational. In the case of jointness and a poorly competitive agricultural sector internationally, Romstad et al. (2000) find that price support equal to the marginal value of the jointly produced public good is the most efficient policy alternative. The role of direct payments for the public goods increases when there are complementary or competing relationships. In the case of relational public goods, Romstad et al. find it very difficult to design a payment scheme that would produce the optimal bundle of public goods.

Latacz-Lohmann (2000) develops a conceptual framework for assessing trade-offs and synergies between domestic agri-environmental policies and trade policies in the context of the environmental dimension of multifunctionality. On the basis of the theory of joint production, Latacz-Lohmann first elaborates the concept of multifunctionality and shows that government intervention to internalise environmental externalities increases domestic social welfare even though it may, because of the joint production, affect the quantities produced and traded. Next, Latacz-Lohmann incorporates multifunctionality into a partial equilibrium trade model developed by Anderson (1992). The model is used to analyse the trade and welfare implications of agri-environmental policies in a large open economy. Alternative cases involving positive and negative externalities in either exporting or importing countries, or both, are assessed. Two of the cases analysed may be particularly problematic in the context of trade negotiations. One such case is the internalisation of a positive production externality in a large importer country, which reduces the quantity traded, and thus the world price. Another one is the internalisation of a negative production externality in a large exporter country, which may curb production and thus reduce trade flows and raise the world price. This may adversely affect low-income net-importers (Latacz-Lohmann 2000).

2.3.2 Policy design for selected non-commodity outputs

Nutrient runoffs. In the case of agricultural nonpoint source pollution (NPSP), 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). When effluents or runoffs cannot be addressed directly because of the nonpoint features of agricultural pollution, the regulator is forced to use indirect instruments, such as input and output taxes, to avoid costly monitoring and enforcement. NPSP instruments that have been proposed or implemented in practice include taxes on polluting inputs, subsidies for abating inputs or technology, liability for damages, taxes and subsidies on ambient concentrations, tradable permits (for example, point-nonpoint source trading), and regulations of inputs and technology (that is, the regulation of farm management practices). Of the proposed instruments for controlling NPSP, especially input and ambient taxes and subsidies have gained the most interest in economic literature (Shortle et al. 1998).

According to Segerson (1988), due to the stochastic nature of agricultural pollution, that is, the fact that ambient pollution levels resulting from agricultural production activities depend on a number of stochastic variables (like rainfall, temperature, or wind) in a manner that cannot be predicted with certainty, there will be a range of possible environmental outcomes associated with any given abatement practice or effluent level at any given time. In other words, because of the natural variability associated with the runoff process, a specific policy instrument or abatement practice will yield a distribution of outcomes rather than a single outcome. Thus, policy instruments for controlling NPSP should be evaluated in terms of their effect on the distribution of outcomes, as determined by distributions of the underlying random variables (Braden and Segerson 1993). In the case of NPSP, policy goals should be defined in terms of the probability of attaining, for example, a mean ambient water quality or a mean runoff with the least cost. Furthermore, due to the stochastic factors even policies that reduce ambient pollution with the least cost may unintentionally increase the variability of pollution levels and thus increase social damage (Shortle 1990, Ribaudo et al. 1999).

Monitoring nonpoint source pollution creates additional problems and uncertainty. Braden and Segerson (1993) divide these monitoring problems into three categories: (i) the inability to observe runoffs, (ii) the inability to infer runoffs from observable inputs, and (iii) the inability to infer runoffs from ambient environmental quality. In general, the inability to observe runoffs is the most problematic feature of NPSP, and the feature that most distinguishes it from point source pollution. Monitoring agricultural runoffs is impractical, as runoffs are by definition diffuse. Although many agricultural pollutants are closely associated with specific and observable production inputs like fertilizers and pesticides, the pollution resulting from a given quantity of application may depend not only on the total quantity applied but also on the weather, soil characteristics, crop uptake, topography, the timing of application, and so on. The inability to infer runoffs from observed ambient environmental quality is the result of both the influence of other polluting farmers and natural randomness. (Braden and Segerson 1993).

The site-specific nature of agricultural NPSP also has important implications for policy design. Since site-specific factors, such as soil quality, soil slope and hydrology, have a great impact on the runoff process, policy instruments that are flexible enough to provide cost-effective outcomes under variable conditions should be preferred (Braden and Segerson 1993, Ribaudo et al. 1999).

The economic literature on the design of nonpoint source pollution control started by an article by Griffin and Bromley (1982). Subsequent influential contributions were made by Shortle and Dunn (1986) and Segerson (1988).

Griffin and Bromley (1982) evaluate four alternative policies for controlling NPSP under certainty related to runoffs (non-stochastic runoff function). The instruments analysed are incentives and standards based on either runoffs or input use/farm management. Griffin and Bromley show that all these instruments exhibit least-cost properties of pollution control and are equally efficient when properly specified. However, the instruments vary in terms of the number of parameters to be specified by the controlling agency. If there are J pollution sources and N management practices/inputs, the number of instruments necessary for accomplishing the environmental goal at least cost is one for runoff incentives, J for runoff standards, and $J \times N$ for input/management incentives and standards.

Shortle and Dunn (1986) point out that the results of Griffin and Bromley (1982) on the equal efficiency of alternative instruments hold when there is a single polluting farm maximising expected profits, information is symmetric between the farmer and the controlling agency, and the damage function is linear. However, allowing asymmetric information and a non-linear damage function affects the relative efficiency of incentives and standards.

Shortle and Dunn (1986) incorporate uncertainty about the level of runoff and weather conditions when examining the relative expected efficiency of incentives and standards based on estimated runoffs and farm management practices/inputs. They show that under asymmetric information incentive-based policies are more efficient than standards since they allow the farm to utilise its informational advantage related to the operation of the farm. When there is both asymmetric information and a non-linear damage function, incentives on farm management practices/inputs are more efficient than runoff-based incentives. This is because incentives on management practices/inputs induce optimal behaviour, while runoff-based incentives do not since they cannot be specified to internalise damages in the case of a non-linear damage function.

Segerson (1988) proposes ambient-based incentives (taxes and subsidies) to overcome the monitoring problems of NPSP. In her model, the regulatory agency monitors ambient pollutant levels for a particular water body and pays (charges) firm-specific subsidies (taxes) when pollutant levels are below (exceed) target levels. This ambient tax or subsidy scheme can, under restrictive conditions, provide correct incentives for polluters to undertake socially efficient abatement, that is, the scheme provides each polluter the efficient marginal incentive to abate (Braden and Segerson 1993). Subsequent contributions to ambient-based instruments have focused mainly on the moral hazard problem of controlling NPSP when there is uncertainty about the relationship between input use and ambient concentrations of pollutants and when input use is costly to monitor (Shortle et al. 1998).

Landscape diversity. The economic analysis of policy design relating to landscape diversity is very scarce. Brunstad et al. (1999) analyse the optimal level of landscape preservation in Norway. They introduce a method for incorporating willingness to pay for landscape preservation in the objective function of a price-endogenous mathematical-programming sector model of Norwegian agriculture. The model is then used for calculating the optimal size of the agricultural sector and the optimal level of subsidies. Their results show that in the optimal solution the current levels of support, production, and employment would decrease significantly. Thus, they conclude that the high levels of agricultural support in Norway cannot be defended by the landscape preservation argument alone.

Agrobiodiversity. The economic literature on the policy design for agrobiodiversity is relatively recent and scarce. Notable contributions have been made by Wossink et al. (1999), Van Wenum et al. (1999), and Van Wenum et al. (2001).

Van Wenum et al. (1999) analyse the impact of farm heterogeneity on the production of wildlife. They present the functional form and an estimation technique for a wildlife production function at the farm level, and develop a random effects model to capture the relationship between wildlife output, management practices, non-observed farm-specific factors, and regional conditions. Species richness and a wildlife yardstick are used in the estimation of the production function. The results show that farm-specific conditions have a significant impact on wildlife production.

Van Wenum et al. (2001) use a location-specific model with the integer programming technique for optimising wildlife management on crop farms. The most important model outcome is the farm-level wildlife – cost frontier. The results also show that the rotation of wildlife activities within a farm is economically preferable to the fixed location of wildlife practices.

Wossink et al. (1999) use network design modelling and GIS for analysing the regional supply of wildlife conservation. They first show how network design modelling can be used for the optimal spatial selection of field margins for creating wildlife corridors in the landscape. Their empirical application is implemented using the GIS model. The results show that selective control has clear advantages in wildlife conservation: more wildlife is obtained at lower costs when the ecologically most important margins are identified.

2.3.3 Policy design under heterogeneous conditions

The simultaneous consideration of several commodity and non-commodity outputs under heterogeneous conditions calls for the spatial targeting of policy interventions. Spatial targeting increases the precision with which a policy meets its goals. However, the success of such spatial targeting depends, for example, on the relationship between the productivity of the commodity outputs and the site productivity of the non-commodity outputs (Latacz-Lohman 2001). Latacz-Lohman (2001) shows that, if agricultural productivity and environmental sensitivity are positively correlated, a trade-off must be made between environmental effectiveness (targeting environmentally sensitive land) and costs in terms of forgone output. If, by contrast, agricultural productivity and environmental sensitivity are negatively correlated, environmental improvements can be bought at relatively low costs as the targeted land has low productivity in commodity production.

Hochman and Zilberman (1978) introduce a microparameter model for analysing pollution – production trade-offs. This model integrates physical and economic models at a disaggregate level to capture the heterogeneity of site characteristics, and then statistically aggregates the microunits into the level needed for policy analysis. This basic model has later been used in several studies examining agriculture – environment relationships (for a brief review of different studies using this approach, see Lichtenberg 2000).

Lichtenberg (2000) analyses the use of differentiated and uniform fertilizer taxes under heterogeneous land quality and land allocation between two crops. He first assumes that environmental quality depends only on the total use of the polluting input (fertilizer) and is invariant with respect to land quality and crop. Under this assumption, Lichtenberg shows that a constant per unit tax (uniform tax) on fertilizer results in socially optimal fertilizer use as well as in the optimal land allocation between crops. Next, Lichtenberg assumes that environmental quality also depends on land quality and crop. Under this assumption he shows that the social optimum cannot be achieved by a uniform tax, but that the socially optimal fertilizer use and land allocation between crops require a tax whose rate varies according to land quality and crop produced. In his discussion concerning the implementation of such differentiated first-best instruments, Lichtenberg notes that differentiated fertilizer taxes would be difficult to implement, but in the case of scenic amenities and wildlife habitats their implementation would be more feasible.

Helfand and House (1995) analyse NPSP 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 individual 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. According to Shortle et al. (1998), 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.

2.3.4 Transaction costs

While spatially differentiated instruments could perform better in the provision of multiple non-commodity outputs, this better performance involves a higher administrative burden. Thus, there are inherent trade-offs between the precision (the degree of goal attainment) of a policy and its related transaction costs (Vatn 2001, 2002). According to Falconer et al. (2001), although there is widespread recognition of the importance of taking transaction costs into account when evaluating agri-environmental policies, there are only few studies that provide empirical estimates of these costs (notable exceptions are McCann and Easter 1998 and Whitby and Saunders 1996).

The transaction costs related to agri-environmental policy could be defined as administrative costs associated with the design, implementation, monitoring, and enforcement of the policy. According to Vatn (2001), transaction costs play a crucial role in the determination of optimal policy, and the increased costs of precision have to be weighed against the potential gains in achieving the objective. Moreover, as precision increases, its marginal utility is likely to decrease, while its marginal transaction costs are likely to increase (Vatn 2001). Thus, when all costs are considered, it is not reasonable to expect that a precise instrument is necessarily the optimal one (Vatn 2002). In the case of environmental multifunctionality, the trade-off between transaction costs and precision depends on the relationship between the commodity and non-commodity outputs. If the outputs are joint products from a non-allocable input, high precision can be achieved even with quite few measures, but in the case of complementary or competitive outputs, payments must be directed towards non-commodity outputs, which implies high transaction costs (Vatn 2001).

Vatn (2002) discusses the consequences of multifunctional agriculture for international trade regimes. He analyses the implications for trade policy when there is joint production between commodity and non-commodity outputs as well as positive transaction costs. He shows that the core issue is the intrinsic trade-off between the precision of a policy instrument and policy-specific transaction costs. If jointness or complementarity is involved, it may not be rational to use direct payments aimed for non-commodity outputs as a universal rule for multifunctional policy design, since it may be more reasonable to pay via the joint commodity output. Thus, it may not be rational to have free trade for commodity outputs while paying separately for non-commodity outputs. This is because the increased transaction costs of targeted policy measures may be higher than the gains obtained from improved precision. Vatn concludes analysis with two important observations concerning situations where there is joint production between commodity and non-commodity outputs as well as positive transaction costs. First, if countries are not equally competitive in commodity markets, free trade may not be the optimal solution. Second, because of positive transaction costs, policy measures linked to commodity prices may be used to obtain the efficient supply of non-commodity outputs. (Vatn 2002).

2.4 Conclusions

This chapter presented a review of the economic literature relating to specific supply, demand, and policy aspects of the environmental dimension of multi-functionality. This literature review provides the basis for the choice of the analytical and empirical approaches for environmental multifunctionality in the present study. The central components of this approach include joint production under heterogeneous conditions, the externality and public good characteristics of selected non-commodity outputs and the related valuation of these outputs, and the targeting and precision of policy instruments.

3 Socially optimal multifunctional agriculture

3.1 Introduction

In this chapter an analytical framework for characterising socially optimal multifunctional agriculture under heterogeneous conditions is developed. To this end, the joint production of two commodity and three environmental non-commodity outputs is analysed in a model of endogenous input use and land allocation. Lichtenberg's model of crop production (1989) is employed to examine farmers' input use and land allocation choices under heterogeneous land quality.[3] This model is augmented by a description of how landscape diversity, agrobiodiversity, and nutrient runoffs are determined by the input and land allocation choices. Lichtenberg's model does not have any spatial structure, but in this study an assumption is made concerning the spatial structure of the arable land as it plays an important role in the analysis of our non-commodity outputs (see Figure 3).

The diversity valuation function is described as a product of "landscape diversity" and "agrobiodiversity". The choice of crops and the land area devoted to them are linked to the landscape and agrobiodiversity valuation as follows. Land allocation between crops together with buffer strips and other field boundaries forms the landscape diversity as a field mosaic. The use of fertilizer as well as buffer strips affect the interaction of arable land with the surrounding ecosystems and the diversity of wild species and, hence, agrobiodiversity. It should be noted that, while the choice of the land area devoted to each crop is a land allocation problem, the choice of the buffer strip size in each parcel is an input use problem.

The privately optimal use of inputs and land allocation create a market solution for nutrient runoffs and, through the landscape mosaic, for agrobiodiversity as well. This private solution is compared to the socially optimal way of producing both commodity and non-commodity outputs. The use of inputs and land allocation that maximises social welfare from multifunctional agriculture is then solved in a parametric model.

The rest of the chapter is organised as follows. Chapter 3.2 formulates a model of crop production with endogenous land allocation to different crops and studies the properties of private agricultural production. Based on the description of the diversity valuation and runoff functions, the socially optimal multifunctional production is developed in Chapter 3.3. Chapter 3.4 provides a

[3] For an analysis of land allocation, see e.g. Orazem and Miranowski 1994, Shah et al. 1995, Plantinga 1996, Hardie and Parks 1997, Miller and Plantinga 1999.

numerical application of the analytical model for characterising socially optimal multifunctional agriculture. Findings are summarised in Chapter 3.5.

3.2 The model of crop production

Consider a watershed with a single river running through it and agricultural land bordering this stream (see Figure 3). For analytical convenience the river is treated as a straight line. Next, the agricultural land in this watershed is divided into production units, rectangular parcels with the same width and length, each of which having an edge along the stream and each extending perpendicularly away from the stream. The nutrients running off from each parcel are assumed to move in a straight line toward the stream. Finally, the size of each parcel is normalised to 1 unit (hectare) of the land area.

Next, land quality is added to the spatial arrangement of land in the watershed. Let us consider a representative farm, which has a fixed amount of arable land *(G)* available for agricultural production. The land quality depends on physical, chemical, and biological factors, such as soil erosion, soil acidity (pH), and soil organic matter. Following Lichtenberg (1989), it is assumed that the variation in land quality 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, that is, $0 \le q \le 1$. Thus, $G(q)$ is the cumulative distribution of q (acreage having quality q at most), while $g(q)$ is its density. For analytical convenience it is assumed that $g(q)$ is continuous and differentiable.

(3.1) $$G = \int_0^1 g(q)dq$$

It is assumed that the farmer wishes to allocate his arable land between two cereal crops, crop 1 and crop 2, and the shares of land devoted to them are denoted by L_1 and L_2, defined as $L_1 = \int_0^{q_1} g(q)dq = G(q_1)$ and $L_2 = \int_{q_1}^1 g(q)dq = G(1) - G(q_1)$, where $G(1) = N$ and denotes the total amount of land. Production exhibits constant returns to land of any given quality but is neoclassical with respect to inputs and land quality. Production of crops requires the use of fertilizer input l.[4]

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[4] The model would easily allow the use of many inputs, such as labour and capital, but they are not crucial for the present analysis and so they are ignored.

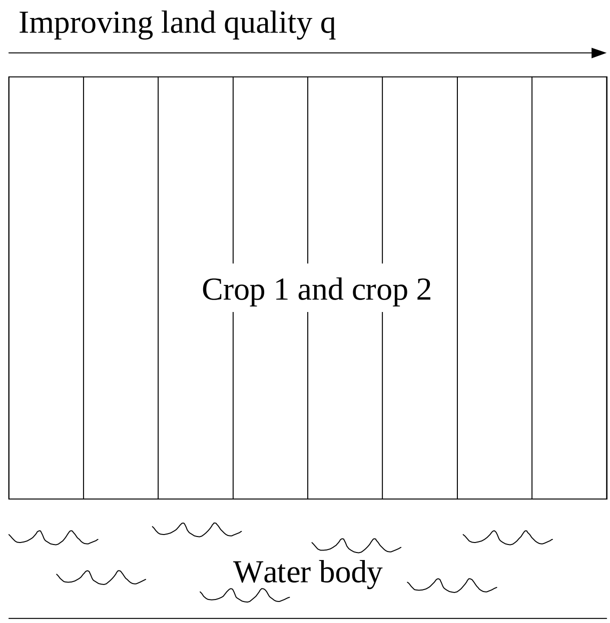

Figure 3. The spatial properties of agricultural landscape.

Figure 3 describes the spatial structure, the "landscape mosaic", of the multifunctional framework (the concept of landscape mosaic is discussed more closely in Chapter 3.3). The land is divided into uniform rectangular parcels and land quality improves from left to right.

Next, an additional type of land use problem is introduced into the model. In Chapter 4 it is assumed that the government pays a subsidy to the farmer for the arable land allocated to a buffer strip to be established between the field and a water body. In order to get the buffer strip subsidy, the farmer has to establish a buffer strip on each parcel (see Figure 5 in Chapter 3.2.2).

The focus in the present chapter is on the private and social optimum in the absence of government intervention. However, for presentational convenience (that is, to avoid repeating analytical steps) the comparative statics of input use and land allocation for fertilizer tax and buffer strip subsidy that will be needed later in Chapter 4 are also solved here.

The model has a recursive structure allowing it to be solved in separate stages. Thus, the model is solved in three stages: (1) optimal fertilizer intensity on land of given quality q and crop i; (2) optimal buffer strip size on land of quality q and crop i given fertilizer intensity (and thus short-run profit) on crop i; and (3) optimal land allocation.

3.2.1 Input use

In the two first stages the farmer takes the land allocation between the two crops as given. In the first stage he chooses the fertilizer intensity *(l)* for land of given quality *q* and crop *i*. Then, in the second stage, the farmer chooses the proportion of the land of quality *q* and crop *i* to be allocated to a buffer strip (*m*), given the fertilizer intensity on crop *i*. Due to the internal homogeneity of each parcel, production is linear in *m*. Finally, production also depends positively on land quality *q* so that agricultural productivity is greater when land quality is higher. Thus, production per each parcel for both crops can be expressed as

(3.2) $$y_i = (1 - m_i) f^{\,i}(l_i; q) \quad \text{for } i = 1,2$$ [5]

This production function is assumed to be concave in the use of fertilizer, that is $f_l > 0;\ f_{ll} < 0$, (subscripts denote partial derivatives).

Next, the corresponding per parcel profit function is developed. The farmer takes the prices of crops p_i and fertilizer *c* as given. The government intervenes in agriculture through two environmentally motivated policy instruments, a fertilizer tax and a buffer strip subsidy. A basic level of tax *t* is levied on the use of fertilizers so that the after-tax price of fertilizer is $c^* = c(1+t)$. The basic level of subsidy on buffer strips is denoted by $b(m_i)$, and to solve for comparative statics it is defined as $b(m_i) = (\lambda - \frac{1}{2}\lambda m_i)m_i$. Thus, the subsidy is decreasing in *m*, which reflects the decreasing ability of wider buffer strips to further reduce runoff and increase species diversity. The per parcel formulation adopted here allows either a uniform (such as the basic levels of the instruments discussed above) or differentiated tax and subsidy. In the case of differentiated instruments, the tax and subsidy rates are differentiated according to crop and parcel. In other words, the policy incentives are fine-tuned with respect to heterogeneous land quality. With uniform instruments a constant per-unit tax on fertilizer and a constant subsidy for buffer strips are implemented irrespective of heterogeneous conditions. Later on in Chapter 4, it will be examined whether a differentiated or a uniform tax and subsidy are socially optimal. Moreover, it is assumed that the ability of buffer strips to prevent runoffs and promote agrobiodiversity is independent of land quality. The farmer's problem is to choose the inputs, l_i and m_i so as to maximise the short-run profit per parcel:[6]

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[5] To avoid complex notation parcels are not indexed, but naturally equation (3.2) and other per parcel equations hold for every parcel.

[6] The following terminology is adopted: "fertilizer intensity" refers to l_i, while "fertilizer used per parcel" refers to $(1 - m_i)l_i$.

(3.3) $$\max_{\{l_i, m_i\}} \pi^i = p_i(1-m_i) f^i(l_i;q) - c^*(1-m_i)l_i + (\lambda - \tfrac{1}{2}\lambda m_i)m_i$$

for i =1,2

The first-order conditions for the optimal solution are

(3.4a) $$\pi^i_{l_i} = p_i f^i_{l_i} - c^* = 0$$

(3.4b) $$\pi^i_{m_i} = -p_i f^i(l_i;q) + c^* l_i + \lambda - \lambda m_i = 0$$

and require that the value of the marginal product of input use equals their respective costs. Because the productivity of each parcel differs due to q, the optimal fertilizer intensity and the size of the buffer strip differ in every parcel as well (see Figure 5).

The second-order conditions are given in equations (3.5a) to (3.5d)

(3.5a) $$\pi^i_{l_i l_i} = p_i f^i_{l_i l_i} < 0$$

(3.5b) $$\pi^i_{m_i m_i} = -\lambda < 0$$

(3.5c) $$\pi^i_{l_i m_i} = 0 = \pi^i_{m_i l_i}$$

(3.5d) $$\Delta = \pi_{ll}\pi_{mm} - \pi_{lm}{}^2 = \pi_{ll}\pi_{mm} > 0$$

where Δ is the determinant of the Hessian matrix of the second-order partial derivatives. Since the principal minors are $|H_1| < 0$ and $|H_2| > 0$, the Hessian matrix is negative definite and the solution maximises profit (see e.g. Chiang 1984). Given that the second-order conditions hold, the comparative statics can be solved from the first-order conditions by differentiating them with respect to the exogenous parameters and applying the Cramer's Rule (see also equations (A1.4a) and (A1.4b) in Appendix 1).

The effects of a change in the price of cereal are given by (3.6a) and (3.6b)

(3.6a) $$\frac{dl}{dp} = -\Delta^{-1}\left\{ f^i_{l_i} \pi_{mm} \right\} > 0$$

(3.6b) $$\frac{dm}{dp} = \Delta^{-1}\left\{ f(l_i;q)\pi_{ll} \right\} < 0$$

The effects of a change in the price of fertilizer are given by (3.7a) and (3.7b)

(3.7a) $$\frac{dl}{dc} = \Delta^{-1}\{(1+t)\pi_{mm}\} < 0$$

(3.7b) $$\frac{dm}{dc} = -\Delta^{-1}\{(1+t)l_i\pi_{ll}\} > 0$$

The effects of a fertilizer tax are given by (3.8a) and (3.8b)

(3.8a) $$\frac{dl}{dt} = \Delta^{-1}\{c\pi_{mm}\} < 0$$

(3.8b) $$\frac{dm}{dt} = -\Delta^{-1}\{cl_i\pi_{ll}\} > 0$$

Finally, the effects of a buffer strip subsidy are given by (3.9a) and (3.9b)

(3.9a) $$\frac{dl}{d\lambda} = 0$$

(3.9b) $$\frac{dm}{d\lambda} = -\Delta^{-1}\{(1-m_i)\pi_{ll}\} > 0$$

The comparative statics of the model are condensed to (see also Appendix 1)

(3.10) $$l_i = l_i(\underset{+}{p_i}, \underset{-}{c}, \underset{-}{t}, \underset{0}{\lambda}) \quad \text{and} \quad m_i = m_i(\underset{-}{p_i}\ \underset{+}{c}, \underset{+}{t}, \underset{+}{\lambda})$$

Hence, as regards market and policy parameters we have

Result 1. *For internally homogenous parcels, a higher output (input) price increases (decreases) fertilizer intensity and decreases (increases) the size of buffer strips. A higher fertilizer tax decreases fertilizer intensity and increases the size of buffer strips. A higher buffer strip subsidy increases the size of buffer strips, but does not affect the fertilizer intensity.*

Result 1 is quite logical. The effects of the market parameters and the own effects of the policy parameters are conventional: a higher output (input) price increases (decreases) the profitability of fertilizer use and increases (decreases) the opportunity cost of buffer strips, a tax decreases the profitability of fertilizer use, and a subsidy increases the marginal revenue from buffer strips. The cross-effects of the policy parameters are asymmetrical, because a fertilizer tax decreases the opportunity cost of buffer strips while a higher subsidy for buffer strips does not change the marginal profitability of fertilizer use. Moreover, for a given fertilizer tax and buffer strip subsidy rate, the use of fertilizer is greater and the size of the buffer strip is smaller for the crop that is grown on higher quality parcels.

In the absence of government intervention, the private solution results in the first-order conditions $\pi^i_{l_i} = p_i f^i_{l_i} - c = 0$ and $\pi^i_{m_i} = -p_i f^i(l_i;q) + cl_i \leq 0$. Thus, without a buffer strip subsidy, the optimal level of buffer strips is zero from the farmer's viewpoint. This is because the foregone value of cultivating the cereal exceeds the savings in fertilizer costs, wherefore establishing buffer strips would result in a net loss of income. Moreover, the fertilizer intensity is higher in the absence of a fertilizer tax.

3.2.2 Land allocation

Following Lichtenberg (1989), it is assumed that the lowest quality parcels are better suited for crop 1, which has a lower fertilizer intensity than crop 2. Lower quality parcels are thus allocated to crop 1. The proportion of land of quality *q* allocated to crop 1 is denoted by $L_1(q)$. The farmer maximises the sum of restricted profit functions π^*_i, *i = 1,2* by allocating his land between both crops, that is,

(3.11) $$\max_{L_1(q)} \int_0^1 \left[\pi^*_1 L_1(q) + \pi^*_2 (1 - L_1(q))\right] g(q)dq$$

The first-order condition for optimal land allocation is

(3.12) $$\pi^*_1(q, p_1, c, t, \lambda) - \pi^*_2(q, p_2, c, t, \lambda) \leq 0$$

This first-order condition leads to a corner solution for every homogenous parcel in the given acreage with differential land quality. Thus, if $\pi^*_1(q) > (<) \pi^*_2(q)$, then all land of quality *q* is allocated to crop 1 (crop 2). In order to ensure that it is optimal to cultivate both crops in the given acreage the following assumptions are made. In order to assure that crop 1 is cultivated on land of quality $0 \leq q < q_1$ and crop 2 on land of quality $q_1 \leq q \leq 1$, so that there is only a single crossing of restricted profits, assumptions have to be made concerning how profits relate to land quality. It is first assumed that restricted profits are higher for crop 1 in lower quality parcels. Second, the restricted profits are higher for crop 2 in the highest land quality $\pi^*_2(1, p_2, c, t, \lambda) > \pi^*_1(1, p_1, c, t, \lambda)$ and restricted profits as a function of land

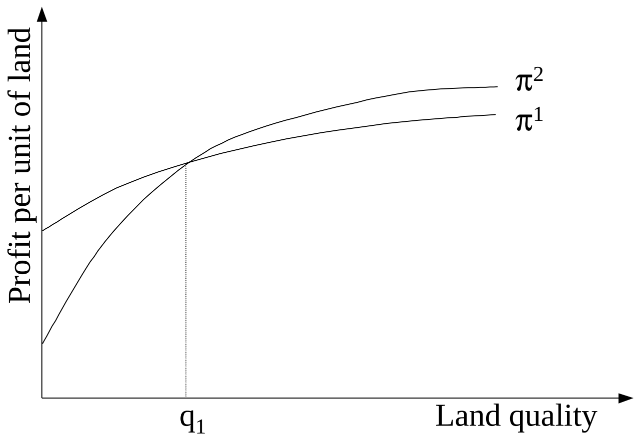

Figure 4. A single crossing of restricted profits at the unique value of land quality q_1.

quality increase more rapidly for crop 2 for all land of quality q as indicated by the fact that $\pi_q^2 > \pi_q^1$. These assumptions assure that there is a single unique q_1 and each crop will be cultivated on a unique, compact range of land qualities (see Figure 4). This is assumed to hold in the following.

The comparative statics of land allocation can be solved as follows. Following Lichtenberg (1989), if crop 1 is cultivated on lower quality parcels, the acreage allocated to it is given by $L_1 = \int_0^{q_1} g(q)dq = G(q_1)$, where the critical quality, q_1, is determined by $\pi_1^*(q_1, p_1, c, t, \lambda) = \pi_2^*(q_1, p_2, c, t, \lambda)$. Then the acreage allocated for crop 2 is $L_2 = G(1) - G(q_1)$. Let θ represent the exogenous variables, that is, $\theta = (p_i, c, t, \lambda)$. To solve for $\frac{\partial q_1}{\partial \theta}$, equation (3.12) that defines the critical value q_1 is totally differentiated to get

$$\pi_{p_1}^1 dp_1 + \left[\pi_c^1 - \pi_c^2\right]dc - \pi_{p_2}^2 dp_2 + \left[\pi_\lambda^1 - \pi_\lambda^2\right]d\lambda + \left[\pi_t^1 - \pi_t^2\right]dt + \left[\pi_{q_1}^1 - \pi_{q_1}^2\right]dq_1 = 0 \quad (3.13)$$

From equation (3.13) we can develop an appropriate expression for each $\frac{\partial q_1}{\partial \theta}$.

Using the Leibnitz rule (see e.g. Taylor and Mann 1983) and accounting for the fact that the derivative of the upper integration limit is defined by the condition defining the critical value q_1, changes in the land allocated for crop 1 are given by $\frac{\partial L_1}{\partial \theta} = g(q_1)\frac{\partial q_1}{\partial \theta}$. The comparative statics of L_2 are simply given by $\frac{\partial L_2}{\partial \theta} = -g(q_1)\frac{\partial q_1}{\partial \theta}$.

Let us recall from equations (3.4a) and (3.4b) and the subsequent discussion that crop 1 is cultivated on lower quality parcels with a lower fertilizer intensity and larger buffer strips, so that $l_2 > l_1$, but $m_2 < m_1$.

The effect of a price increase of crop 1 on land allocation is given by

(3.14a) $$\frac{\partial L_1}{\partial p_1} = \frac{-g(q_1)(1-m_1)f^1(l_1;q)}{\underset{(-)}{\Delta}} > 0$$

(3.14b) $$\frac{\partial L_2}{\partial p_1} = \frac{g(q_1)(1-m_1)f^1(l_1;q)}{\underset{(-)}{\Delta}} < 0$$

where $\Delta = \left[\pi^1_{q_1} - \pi^2_{q_1}\right] < 0$ because $\pi^1 > \pi^2$ for $q < q_1$ and $\pi^1 < \pi^2$ for $q \geq q_1$.

The effect of a price increase of crop 2 on land allocation is given by

(3.15a) $$\frac{\partial L_1}{\partial p_2} = \frac{g(q_1)(1-m_2)f^2(l_2;q)}{\underset{(-)}{\Delta}} < 0$$

(3.15b) $$\frac{\partial L_2}{\partial p_2} = \frac{-g(q_1)(1-m_2)f^2(l_2;q)}{\underset{(-)}{\Delta}} > 0$$

The effect of a fertilizer price increase on land allocation is given by

(3.16a) $$\frac{\partial L_1}{\partial c} = \frac{g(q_1)\overbrace{\left[(1-m_1)l_1 - (1-m_2)l_2\right]}^{(-)}}{\underset{(-)}{\Delta}} > 0$$

$$\text{(3.16b)} \qquad \frac{\partial L_2}{\partial c} = \frac{-g(q_1)\overbrace{\left[(1-m_1)l_1-(1-m_2)l_2\right]}^{(-)}}{\underset{(-)}{\Delta}} < 0$$

These effects of end prices and input costs on land allocation are already familiar from Lichtenberg (1989). The effects of agri-environmental policy instruments are as follows.

The effect of a buffer strip subsidy on land allocation is

$$\text{(3.17a)} \qquad \frac{\partial L_1}{\partial \lambda} = \frac{-g(q_1)\overbrace{\left[(m_1-\tfrac{1}{2}m_1^{\,2})-(m_2-\tfrac{1}{2}m_2^{\,2})\right]}^{(+)}}{\Delta} > 0$$

$$\text{(3.17b)} \qquad \frac{\partial L_2}{\partial \lambda} = \frac{g(q_1)\overbrace{\left[(m_1-\tfrac{1}{2}m_1^{\,2})-(m_2-\tfrac{1}{2}m_2^{\,2})\right]}^{(+)}}{\Delta} < 0$$

For the effect of a fertilizer tax we get

$$\text{(3.18a)} \qquad \frac{\partial L_1}{\partial t} = \frac{g(q_1)c\overbrace{\left[(1-m_1)l_1-(1-m_2)l_2\right]}^{(-)}}{\underset{(-)}{\Delta}} > 0$$

$$\text{(3.18b)} \qquad \frac{\partial L_2}{\partial t} = \frac{-g(q_1)c\overbrace{\left[(1-m_1)l_1-(1-m_2)l_2\right]}^{(-)}}{\underset{(-)}{\Delta}} < 0$$

The effects of exogenous parameters on land allocation are summarised in (3.19)

$$\text{(3.19)} \qquad L_1 = L_1(\underset{+}{p_1}, \underset{-}{p_2}, \underset{+}{c}, \underset{+}{t}, \underset{+}{\lambda}) \quad \text{and} \quad L_2 = L_2(\underset{-}{p_1}, \underset{+}{p_2}, \underset{-}{c}, \underset{-}{t}, \underset{-}{\lambda})$$

As regards the agri-environmental policy instruments, we have

Result 2. *Both fertilizer tax and buffer strip subsidy shift land into the production of the less fertilizer-intensive crop.*

Logically, a higher fertilizer tax decreases the application of fertilizers in the production of both crops, and given our assumptions concerning land quality, makes some parcels of crop 2 less profitable relative to crop 1. The same happens with higher input prices. Consequently, the farmer shifts more land to the production of crop 1, which uses fertilizers less intensively. An increase in the buffer strip subsidy increases the size of buffer strips in both production lines, but more so in the production of crop 1, thus shifting additional parcels to crop 1. Hence, in the absence of government intervention, the farmer allocates more land into the production of the more fertilizer-intensive crop 2.

Figure 5 shows how the landscape is structured by land allocation between the two crops and by the use of inputs, namely the buffer strips. With a given basic level of buffer strip subsidy all parcels contain a buffer strip, but its width differs across parcels due to differences in agricultural productivity. Naturally, field boundaries are a crucial part of this landscape. Given, however, that they remain the same throughout the analysis, these are not explicitly accounted for. Changes in the exogenous parameters (crop prices, fertilizer costs, and govern-

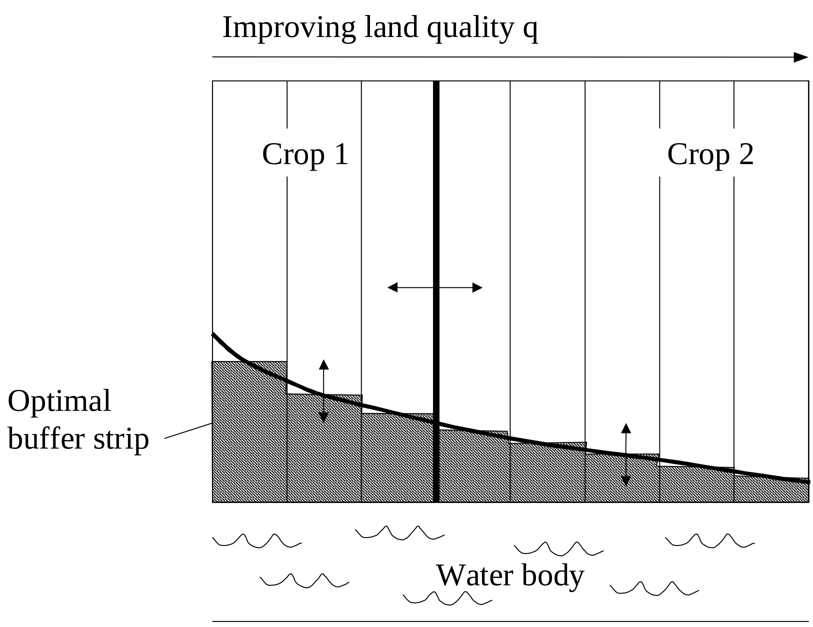

Figure 5. Land allocation and optimal buffer strip area in each parcel.

ment instruments) transform the composition and configuration of this landscape mosaic by inducing adjustments in crop selection and buffer strips that are indicated by arrows in Figure 5. (cf. Eiden et al. 2000).[7]

3.3 Socially optimal provision of multifunctional outputs

Let us assume that society wishes to promote multifunctional agriculture and regards the aesthetic value of diverse agricultural landscapes, agrobiodiversity, and surface water quality as the most important non-commodity outputs. Designing socially optimal multifunctional agriculture requires that we first define how landscape diversity, agrobiodiversity, and nutrient runoffs are related to the commodity production.

3.3.1 Agricultural diversity and runoff functions

How do people value diversity in agriculture? Studies concerning people's attitudes and valuation allow us to make the following three observations. First, there are typically some site-specific features, such as uneven terrain or lake areas, that together with the agricultural land mosaic are the most important sources of landscape aesthetics (see e.g. Dillman and Bergstrom 1991). Second, landscape valuation is often also related to species diversity, for instance, through the preservation of old domestic plant and animal species (see e.g. Drake 1992). Third, people show increasing awareness concerning the effects of the use of chemical inputs on the surrounding ecosystems and on species diversity (see e.g. Siikamäki 1997).

The effects of agricultural production practices on the surrounding ecosystems and on species diversity can also be assessed from an ecological angle. Kleijn (1997), Wossink et al. (1999), and Bäckman et al. (1999) show that in arable fields the largest number of species of both flora and fauna are to be found at the field boundary. Field boundaries provide forage, shelter, reproduction and over-wintering sites, and ecological corridors for wildlife, thus belonging to the most important semi-natural habitats sustained by agriculture. Field boundaries and buffer strips support many flowering plants and insects, such as butterflies and bees, which are important for bird species, and provide important habitats for pest predators, but also for weeds and insects (Swifth and

[7] An alternative spatial description would be one where land quality improves along with the distance from the shore. In this case, a buffer strip would be established only on the lowest quality parcels next to the water body, and the rest of the parcels would be allocated between crop 1 and crop 2.

Anderson 1994). Hence, in addition to promoting species diversity and reducing sediment, nutrient and pesticide runoffs, they may have both positive and negative agronomic effects.[8]

The agricultural landscape mosaic can be described by different indices, such as the edge density index, the Shannon Diversity Index, and the Interspersion and Juxtaposition Index, which capture the number and distribution of different patches in the landscape (see e.g. Eiden et al. 2000). These indices are linked to landscape diversity by Eiden et al. (2000), but they could also be made to apply to the overall diversity comprising both landscape diversity and agrobiodiversity. For example, Duelli (1997) proposes a conceptual "mosaic" model in which biodiversity evaluation is based on structural landscape parameters, such as habitat diversity and landscape heterogeneity, and functionally on metacommunity dynamics. This approach links site-specific biodiversity estimates to the spatial heterogeneity of the landscape mosaic (for a comprehensive treatment of the landscape mosaic see e.g. Forman 1995). In the present study the Shannon Diversity Index is used in the empirical analysis as a proxy for landscape diversity. The analysis of agrobiodiversity is based on Ma et al. (2002), and estimates of floral species richness as a function of the buffer strip area are given.

In the theoretical analysis a general description of people's diversity valuation is applied. The choice of crops is linked to landscape valuation. For brevity this is called *landscape diversity valuation*. Then agrobiodiversity is expressed as a function of fertilizer use and buffer strips. This part of the overall diversity is called *agrobiodiversity valuation*. Hence, the description of the diversity valuation function in this study is a product of "landscape diversity" and "agrobiodiversity". This multiplicative form of the diversity valuation function reflects the fact that in order for diversity to have a non-zero value, both of its components must have non-zero values.

(3.20) $$\Omega = k(L_1, L_2)h(m, \bar{l})$$

where $\bar{l}$ refers to the total amount of the fertilizer used $\bar{l} = \int_0^1 ((1-m_1(q))l_1(q)L_1 + (1-m_2(q))l_2(q)L_2)g(q)dq$, and m to the total

[8] In the absence of buffer strip management, that is, mowing and removing of the cuttings, the nutrient content of the buffer strip increases, counter-affecting agrobiodiversity promotion. A higher nutrient status reduces botanical species-richness, and thereby the abundance and diversity of wildlife in buffer strips. The reason for this is the tighter competition for light, which leads to the displacement of shorter species by taller species (Tilman 1993, Kleijn and Snoeijing 1997). This adverse effect may even be more severe than that of herbicide applications (Kleijn and Snoeijing 1997).

area of the buffer strips $m = \int_0^1 (m_1(q)L_1 + m_2(q)L_2)g(q)dq$, and L_1, L_2 are defined in equation (3.1). The term $k(L_1, L_2)$ indicates the valuation of landscape diversity as a function of land allocation to different crops, and the latter term $h(m, \bar{l})$ indicates the valuation of agrobiodiversity as a function of input use. Via L_1, L_2, m and $\bar{l}$, the diversity valuation function also depends on land quality. It is assumed that increasing the acreage for each crop increases landscape diversity but in a diminishing way, that is, $k_1 > 0$ and $k_2 > 0$, but $k_{11} < 0$ and $k_{22} < 0$.[9] For the use of fertilizer input it is assumed that $h_{\bar{l}} < 0$ and $h_{\bar{l}\bar{l}} < 0$, indicating that the higher the amount of fertilizer used the greater the loss in agrobiodiversity. Finally, increasing the size of the aggregate buffer strip area increases agrobiodiversity by enlarging the field boundary but with decreasing returns, that is, $h_m > 0$ and $h_{mm} < 0$ (this is confirmed e.g. by Ma et al. 2002).

The runoff of nutrients (kg) from each parcel can be expressed as a function of fertilizer intensity l_i and the size of the buffer strip m_i as follows: $z_i = v_i(\bar{l}_i(q), m_i(q))$; for $i = 1,2$, where $\bar{l}_i(q) = (1 - m_i(q))l_i(q)$ with $v_{\bar{l}} > 0$, $v_{l\bar{l}} > 0$ and $v_m < 0$, $v_{mm} > 0$. Thus, the runoff function is convex in the fertilizer application but concave in buffer strips (for the former see e.g. Simmelsgaard 1991, and for the latter Uusi-Kämppä et al. 2000). The total amount of runoff from the land area devoted to crop 1 and crop 2 can be described as

$$(3.21) \qquad z = \int_0^1 \left[v_1(\bar{l}_1(q), m_1(q))L_1 + v_2(\bar{l}_2(q), m_2(q))(1 - L_1) \right] g(q)dq$$

3.3.2 Command optimum

Let us now turn to the determination of the socially optimal solution for multifunctional agriculture as a command optimum in the absence of taxes and

[9] These are ceteris paribus assumptions. Given the definition of L_1 and L_2, and the fixed amount of arable land, the effects of land allocation on the marginal landscape valuation will depend on the difference between the marginal valuations, $k_1 - k_2$, as will be seen later on.

subsidies (see e.g. Weitzman 1974 for social planner's problem in general and Lichtenberg 2000 in the context of agriculture). The command optimum means that a social planner determines the optimal use of inputs for all parcels, as well as the land allocation between the crops. It is assumed that the government maximises the producers' surplus augmented with diversity valuation and the damage from nutrient runoffs. Society values diversity, $k(L_1, L_2)h(m, \bar{l})$, but derives disutility from the nutrient runoffs. The runoff damage function $d(z)$ is convex, that is, $d'(\cdot) > 0$ and $d''(\cdot) > 0$. Thus, the social welfare function can be expressed as

$$SW = \int_0^1 (L_1\pi_1 + (1 - L_1)\pi_2) g(q) dq - d(z) + k(L_1, L_2)h(m, \bar{l}) \tag{3.22}$$

The command optimum is solved recursively. Choosing the use of inputs for each parcel so as to maximise (3.22) yields

$$SW_l^i = p_i f_l^i - c - d'(z)\frac{\partial v_i}{\partial \bar{l}_i} + k(\cdot)h_{\bar{l}} = 0 \tag{3.23a}$$

$$SW_m^i = -p_i f^i(l_i; q) + cl_i - d'(z)\frac{\partial v_i}{\partial m_i} + k(\cdot)h_m = 0 \tag{3.23b}$$

According to (3.23a), in each parcel fertilizer is used up to the point where the value of its marginal product is equal to its unit price, adjusted for its marginal effects on runoffs and agrobiodversity. The marginal damage of runoffs comprises two components, a constant marginal damage term multiplied with marginal runoffs from fertilizer used in each parcel. Note that marginal runoffs vary across land quality because l_i and m_i do. The marginal agrobiodiversity effect is constant over parcels. The buffer strip size is optimal when the net loss of income due to decreased production equals the marginal value from runoff reduction (varying over parcels) and the constant marginal benefits from agrobiodiversity promotion (3.23b). Note that the choice of the width of the buffer strip clearly differs across parcels, causing the fertilizer used per parcel to do so, too. This has an interesting implication. Runoff no longer depends solely on the amount of fertilizer used, since the vegetation on the buffer strip removes nutrients from the runoff. The wider the buffer strip the greater is this nutrient uptake. Let us recall the private per parcel solution in the absence of taxes and subsidies, that is, $\pi_l^i = p_i f_{l_i}^i - c = 0$ and $\pi_m^i = -p_i f^i(l_i; q) + cl_i \leq 0$. Thus, the private solution neglects the effects on the production of public goods and on runoff damage. While the use of the fertilizer input is excessive, the use of buffer strips is too small from the viewpoint of society. In fact, in the absence of incentives provided by society, the privately optimal area of buffer strips is zero.

The social planner maximises (3.22) by allocating land to crops 1 and 2, and accounting for the effects of land allocation on diversity and nutrient runoffs. Due to uniform land quality within each parcel, a corner solution is obtained where each parcel is allocated to the crop with the highest social return. Recalling the assumptions from Chapter 3.2, there will be only a single crossing of social rent curves, and thus a single unique value of land quality dividing the area into a unique, compact range of land qualities for both crops defined by

(3.24) $$\pi_1^* - d'(\cdot)v_1 + \Omega_{L_1} = \pi_2^* - d'(\cdot)v_2 + \Omega_{L_2}$$ [10]

Comparing this with the privately optimal solution, where $\pi_1^* = \pi_2^*$, reveals the difference. In addition to maximum profits, the land allocation will depend on the marginal valuation of the diversity benefits and runoff damages. Consequently, the socially optimal land allocation, as determined also by the marginal valuation of the diversity benefits and runoff damages, shifts more land to crop 1, which uses fertilizer less intensively, has larger buffer strips and thus produces a higher level of diversity.

Summing up, it has been shown that

Proposition 1. *Socially optimal multifunctional agriculture promotes landscape diversity and agrobiodiversity, as well as surface water protection by reducing the use of the polluting fertilizer input, increasing the size of buffer strips and increasing the mosaic pattern of fields relative to privately optimal agricultural production.*

Figure 6 illustrates how socially optimal multifunctional agriculture changes the landscape relative to the landscape emerging from the privately optimal solution.

Socially optimal multifunctional agriculture implies that a buffer strip, with decreasing size in land quality, is established on each parcel, while the private market solution (without a buffer strip subsidy) does not entail buffer strips. Relative to the private optimum, the socially optimal solution shifts more land to crop 1, which uses fertilizer less intensively. Hence, the socially optimal land-

[10] The partial derivatives are $\Omega_{L_1} = k_1 h(\cdot) + k(\cdot) h_{L_1} \bar{l}_1 + k(\cdot) h_{L_1} m_1$ and $\Omega_{L_2} = k_2 h(\cdot) + k(\cdot) h_{L_2} \bar{l}_2 + k(\cdot) h_{L_2} m_2$

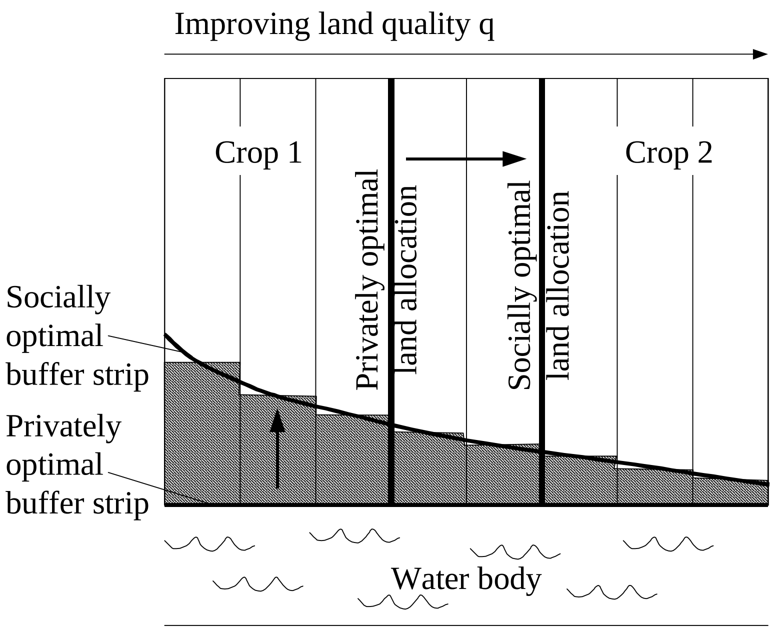

Figure 6. The privately and socially optimal buffer strips and land allocation.

scape sustains higher agrobiodiversity and landscape diversity, as well as lower nutrient runoffs, than the private optimum.

Correcting the privately optimal use of inputs and land allocation to reflect the socially optimal multifunctional agriculture requires an appropriate use of policy instruments. The socially optimal rates of these instruments are solved in Chapter 4. Now let us turn to the numerical characterisation of the socially optimal multifunctional agriculture.

3.4 Numerical characterisation of multifunctional agriculture

This section illustrates in a parametric model the analytical approach to multifunctional agriculture. The basic features of the socially optimal multifunctional agriculture are determined using Finnish data. The private optimum in the absence of policy instruments as a market solution is solved first, followed by the social optimum as a command optimum. After this the privately optimal solution is compared with the command optimum in terms of input use, production, short-run profits, nitrogen runoffs, and diversity, for which two measures are offered, floral species richness (agrobiodiversity) and the Shannon Diversity Index (landscape diversity). Finally, social welfare outcomes under these two solutions are calculated.

3.4.1 Parametric model

Following the analytical model, let us start with the farmer's production function and apply a quadratic nitrogen response function with parameters estimated for barley (crop 1) and wheat (crop 2) in clay soils by Bäckman et al. (1997).

$$y_i = a_i + \alpha_i l_i + \beta_i l_i^2 \quad \text{for } i = 1,2 \tag{3.25}$$

where y_i = yield response, kg/ha
a_i = intercept parameter
l_i = nitrogen fertilizer intensity, kg/ha
α_i, β_i = parameters, $\alpha_i > 0$, $\beta_i < 0$

Land quality q is continuous and incorporated through the intercept parameter a_i and response parameter α_i, both of which are concave in land quality:

$$\begin{aligned} a_1 &= e_0 + e_1 q - e_2 q^2 \qquad & \alpha_1 &= \mu_0 + \mu_1 q - \mu_2 q^2 \\ a_2 &= n_0 + n_1 q - n_2 q^2 \qquad & \alpha_2 &= \eta_0 + \eta_1 q - \eta_2 q^2 \end{aligned} \tag{3.26}$$

In (3.26), e_0 and n_0 are the lowest levels of natural productivity, a_i, e_1 and n_1 are the slopes of the natural productivity change, e_2 and n_2 are concavity terms, and q is land quality (increasing from 1 to 60).[11] For the response parameter α_i, we denote the basic levels by μ_0 and η_0, the slopes by μ_1 and η_1, and the concavity terms by μ_2 and η_2.

The farmer's short-run profits per parcel for crop i in the absence of government intervention are given by

$$\pi^i = p_i(1-m_i)\left[a_i + \alpha_i l_i + \beta_i l_i^2\right] - c(1-m_i)l_i \quad \text{for } i = 1,2 \tag{3.27}$$

where a_i and α_i are defined by (3.23).

Next a parametric description of the environmental parts of the social welfare function, namely the runoff damage and diversity valuation, is developed. The nitrogen leakage function used is (Simmelsgaard 1991)

$$y(N_i) = y_n \exp(b_0 + bN_i) \quad \text{for } i = 1,2 \tag{3.28}$$

[11] Parcel size in the parametric model is one hectare. Total acreage in the model is 60 hectares, resulting in 60 parcels.

where $y(N_i)$ = nitrogen leakage at fertilizer intensity level N_i, kg/ha, y_n = nitrogen leakage at average nitrogen use, b_0 = a constant (<0), b = a parameter (>0), and N_i = nitrogen fertilization relative to the normal fertilizer intensity for the crop, $0.5 \le N \le 1.5$. The reductive effect of the buffer strip on the nitrogen runoff z_i is incorporated as follows

(3.29) $$z_i = (1 - jrm_i)y(N_i) \quad \text{for } i = 1,2$$

where j = share of surface runoff in total (surface and drainage) runoff, r = nitrogen removal effectiveness of the buffer strip, and m is the size of the buffer strip (recall that given the size and shape of each parcel, m unambiguously defines the width of the buffer strip).

Based on Finnish experimental studies on grass buffer strips (Uusi-Kämppä and Yläranta 1992, 1996) and on the leaching of nitrogen (Turtola and Jaakkola 1987, Turtola and Puustinen 1998), the following assumptions are made. Of the total nitrogen load, 50% is surface runoff. A 10-meter-wide grass buffer strip is able to reduce 50% of the total nitrogen of this surface runoff. Moreover, since combined surface and drainage nitrogen leakages (y_n) at the fertilization level of 100 kg N/ha have been in the order of 10–20 kg N/ha in Finnish experimental studies, the parameter y_n is set at the value of 15.[12]

For the social value of runoff damages an estimate provided by Vehkasalo (1999) is used. He approximated the social benefits of reducing nitrogen runoffs from Finnish agriculture by applying the averting expenditure valuation method, and estimated the costs of a corresponding nitrogen reduction at municipal wastewater treatment facilities. The cost estimate is FIM 9.5 per reduced kg of nitrogen (for a 10–20% reduction).[13] The diversity valuation for this study is taken from Aakkula (1999), which suggests an average WTP/ha of FIM 466 as an estimate for the economic value of pro-environmental farming in Finland.[14] However, besides landscape diversity and agrobiodiversity, this estimate also includes the value of nutrient runoff reduction. Therefore, the estimate of FIM 340 per hectare is used, which is 27% lower than Aakkula's average WTP.[15] Moreover, Aakkula's estimate is the sum of both components of

[12] We are obligated to use Finnish parameters in a Danish leakage function, because no estimations for a Finnish leakage function are available despite the leaching experiments made in Finland. However, the Danish leakage function was estimated for sandy and clay soils cultivated by barley and wheat, and the Finnish leaching experiments also took place on clay soils cultivated by barley and wheat. Thus, data from these two sources can be reasonably combined.

[13] FIM 1= 0.1682 euro.

[14] More precisely, Aakkula used the contingent valuation method to elicit a monetary value for the conversion from conventional agriculture to pro-environmental farming.

[15] This 27% (FIM 126) reduction is in the range of Vehkasalo's runoff damage estimates per hectare (between FIM 95 and 190 when runoffs range between 10 and 20 kg N/ha).

diversity (landscape diversity and agrobiodiversity) and their separate values cannot be detached from that estimate. Therefore, the diversity valuation estimate is linked to buffer strip areas which contribute to both diversity types (see the empirical diversity indices below). The runoff damage and agrobiodiversity valuation parts of social welfare are thus given by

(3.30) $$9.5z_i + 340m_i^{0.08} \quad \text{for } i = 1,2$$

where $z_i = (1 - jrm_i)y(N_i)$, and $y(N_i)$ is defined in (3.28) and (3.29), and the exponent 0.08 in the last term calibrates the valuation to Finnish levels.

After obtaining the privately and socially optimal solutions for input use and land allocation, the following diversity indices are calculated on the basis of buffer strip areas and crop areas.

Agrobiodiversity valuation is linked to species diversity with the help of Ma et al. (2002), who use Finnish data to investigate the relationship between the buffer strip area and floral species richness. Ma et al. revised the conventional species-area relationship for buffer strips by describing buffer strip area as the product of length (L) and width (W). Their modified species-area model is $S = \psi L^{\varphi_\alpha} W^{\varphi_\beta}$, 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. Ma et al. (2002) estimated the coefficients to be the following: $\psi = 1.6331$, $\varphi_\alpha = 0.0009$, and $\varphi_\beta = 0.0977$. Thus, one can produce a higher number of floral species with a per unit area increment by widening rather than lengthening the buffer strips.

In general terms, the spatial structure of landscape is associated with the composition (variety and abundance) and configuration (distribution or spatial character) of different patch types within the landscape, and a number of mathematical indices have been developed for describing it (see McGarigal and Marks 1994). The Shannon Diversity Index (*SHDI*) is used as a measure of landscape diversity, as it works well when there is no temporal variation in the number of patch types and total area. The Shannon Diversity Index (*SHDI*) is calculated for the reference area by summing over all patch types the product of the proportion of the each patch type multiplied by the natural logarithm of that proportion,

(3.31) $$SHDI = -\sum_{i=1}^{n} (P_i * \ln P_i) \quad \text{for } i = 1,2,3,4$$

where n = number of patch types and P_i = the proportion of the area covered by patch type i out of the four patches in the model. The *SHDI* is a combination of the richness (number of different patch types) and evenness (proportional area distribution among patch types) of the landscape diversity. The *SHDI* is zero when a landscape contains only one patch, and the value increases with the number of patch types and/or as the proportional distribution of the area among patch types becomes more equal (Eiden et al. 2000).

Table 4. Parameter values in the numerical application.

Parameter	Symbol	Value
Price of barley	p_1	FIM 0.73/kg
Price of wheat	p_2	FIM 0.83/kg
Price of nitrogen fertilizer	c	FIM 5.95/kg
Concavity of the slope parameter α		(*)
Basic level of response for crop 1	μ_0	52.9
Basic level of response for crop 2	η_0	35.8
Slope of the response change for crop 1	μ_1	0.005
Slope of the response change for crop 2	η_1	0.005
Concavity term for crop 1	μ_2	0.00004
Concavity term for crop 2	η_2	0.00004
Parameter of quadratic nitrogen response function	β	-0.173 for barley
		-0.094 for wheat
Concavity of the intercept parameter *a*		(*)
Basic level of productivity for crop 1	e_0	800
Basic level of productivity for crop 2	n_0	780
Slope of the productivity change for crop 1	e_1	10
Slope of the productivity change for crop 2	n_1	23
Concavity term for crop 1	e_2	0.07
Concavity term for crop 2	n_2	0.15
Share of surface runoff in total runoff	j	0.5 (i.e. 50%)
Nitrogen removal effectiveness of buffer strip	r	0.5 (i.e. 50%)
Nitrogen leakage at average nitrogen use	y_n	10-20 kg/ha
Parameter of leakage function	b	0.7
Constant parameter of leakage function	b_0	-0.7
Parameters of species-area relationship		
Species diversity in initial area	ψ	1.6331
The coefficient of length increase	φ_α	0.0009
The coefficient of width increase	φ_β	0.0977

Note: Prices are from 1999 (FIM 1 = 0.1682 euro). The price of nitrogen is calculated on the basis of a compound NPK fertilizer.
(*) The estimated, average constants for these crops are a_1 = 1010 for barley and a_2 = 1274 for wheat. The response parameter a is 52.9 for barley and 35.8 for wheat (Bäckman et al. 1997).

Parameter values are reported in Table 4. The arable land area is assumed to be 60 hectares. By assumption the width of the field area (i.e. the distance from the water border to the other edge of each parcel) is 500 m and the length (i.e. the border along the waterway) is 1200 m. Moreover, given the same width and the length of each parcel the share of land allocated to buffer strip m defines uniquely the area and the width of buffer strip (in meters). For example if buffer strip size is 0.005 hectares its width is 2.5 meters and if the size is 0.01 then the width is 5 meters etc.

3.4.2 Numerical solutions

The calibrated model is solved using Mathematica and Excel. Input use, land allocation, restricted profits, and social welfare (social returns) are first solved by using Mathematica. Next, these solutions are used for calculating production, nitrogen runoffs, floral species richness, and the Shannon Diversity Index in Excel. The solution technique is as follows (see Appendix 5 for sample equations of the calibrated model). First, the farmer's privately optimal fertilizer intensity is solved in each parcel for both crops. This is done by taking the first-order conditions from equations (A5.1a) and (A5.1b) in Appendix 5 with respect to fertilizer use and then solving for the optimal fertilizer intensity in each parcel for both crops. In the absence of policy instruments (that is, without a buffer strip subsidy) the privately optimal width of buffer strips is zero. Given the optimal fertilizer intensity for each parcel, the profits for both crops in each parcel are obtained and the optimal land allocation between the crops can be determined. Socially optimal input use for both crops in each parcel is solved from equations (A5.2a) and (A5.2b) in Appendix 5. This is done by taking the first-order conditions with respect to fertilizer use and buffer strips and then solving for the socially optimal fertilizer intensity and buffer strip width for both crops in each parcel. Given the socially optimal input use, the social returns are obtained and socially optimal land allocation can be determined.

Hence, the private optimum in the absence of government intervention is solved first, followed by the social optimum as a command optimum. Tables 5 and 6 present some of the results. Table 5 presents land allocation, the average use of inputs, and the average production per hectare under these two solutions.

The farmer's private optimum allocates 23 hectares for crop 1 (barley) and 37 hectares for crop 2 (wheat). The average nitrogen use is 129.5 kg/ha for crop 1 and 153.0 kg/ha for crop 2. The optimal rate of nitrogen application differs between parcels as the nitrogen response differs because of heterogeneous land quality. In the private optimum the fertilizer intensity and the amount of fertilizer used per parcel coincide, since in the absence of a buffer strip subsidy the

Table 5. Land allocation, average input use and average production under alternative solutions.

	Land allocation, ha		Nitrogen use, kg		Buffer strip size, ha		Production, kg/ha	
	Crop 1	Crop 2	Crop 1	Crop 2	Crop 1	Crop 2	Crop 1	Crop 2
Private optimum	23	37	129.5	153.0	-	-	4863	4762
Social optimum	27	33	122.9	141.5	0.0086	0.0081	4787	4663

optimal amount of buffer strips is zero for the farmer. Average production per hectare is 4,863 kg for crop 1 and 4,762 kg for crop 2.

As Table 5 and Figure 7 show, the socially optimal land allocation differs from the private optimum since land allocation is, in addition to short-run profits, also a function of the valuation of diversity benefits and runoff damages. Thus, it is socially optimal to allocate a greater share of land for crop 1, which uses nitrogen fertilizer less intensively. Indeed, the difference between the privately and socially optimal solution predominantly results from the fact that crop 2 implies higher runoff damages due to higher application of nitrogen fertilizer and smaller buffer strips. Expectedly, the use of fertilizer in each parcel is smaller under the social optimum than in the private solution. This holds even more clearly for buffer strips, for which the private solution provides a zero size. The production of crops per hectare decreases in the social optimum relative to the private solution because of the lower level of nitrogen application and because of the presence of buffer strips.

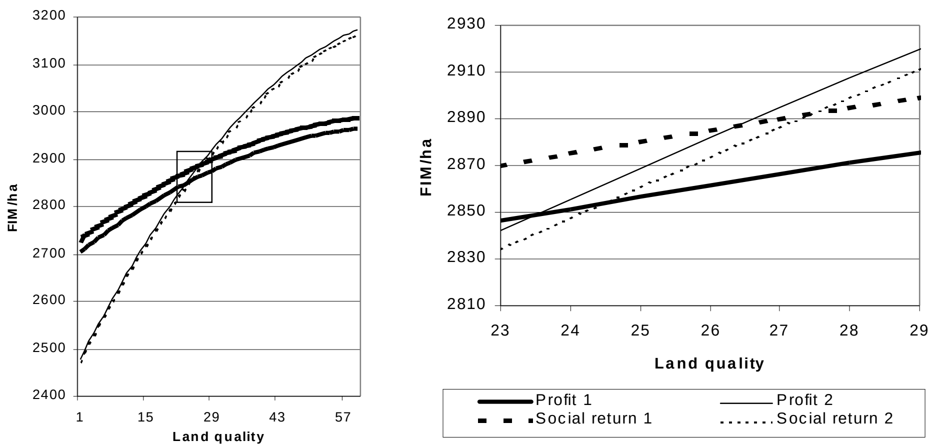

Figure 7. Private profits and social returns.

Table 6. Economic, environmental and social welfare outcomes under alternative solutions.

	Farmer's profits, FIM	Runoffs, kg	Species richness	*SHDI*	*SW*
Private solution	176 481	1375	-	0.67	163 418
Social optimum	174 562	1235	75	0.74	176 739

Table 6 summarises the general economic and environmental features of the private market solution and the social optimum in terms of total short-run profits, total nitrogen runoffs, total number of floral species, the Shannon Diversity Index (*SHDI*), and social welfare (*SW*).

As Table 6 shows, the private market solution in the absence of taxes and subsidies yields higher private profits, but also higher runoffs and a lower value for *SHDI*. In the absence of a buffer strip subsidy the optimal level of buffer strips and thus the estimate of floral species richness in buffer strip areas is zero.[16] Consequently, social welfare is lower in this solution compared to the social optimum. The socially optimal solution produces lower private profits because of the internalisation of negative and positive externalities associated with runoffs and diversity. The *SHDI* increases mainly due to the increased number of patch types through the emergence of buffer strip areas. The buffer strip areas are endowed with 75 floral species. Social welfare is FIM 13,321 higher in this solution than in the private optimum, giving FIM 222 as the difference per hectare.

Figure 8 summarises the private optimum in terms of the analysed commodity and non-commodity outputs and the resulting level of social welfare. The private optimum is presented against the benchmark of the social optimum, which is indexed to 100.

Due to uncertainty related to marginal social benefits and costs of non-commodity outputs a sensitivity analysis relating to the marginal damage of runoffs has been conducted. Both a decrease of 30% (from FIM 9.5 to FIM 6.65) and an increase of 30% (from FIM 9.5 to FIM 12.35) in the runoff damage estimate were examined in order to check the robustness of the benchmark case of social optimum. When the damage estimate was decreased by 30%, the crossing point, that is, the critical land quality for social returns decreases from 28 to 27, and thus one hectare more is allocated for crop 2 which is more intensive in

[16] It should be noted that this estimate only concerns buffer strip areas. Field boundaries do support floral species even in the private solution, but they are excluded from this analysis because they remain unchanged in all solutions.

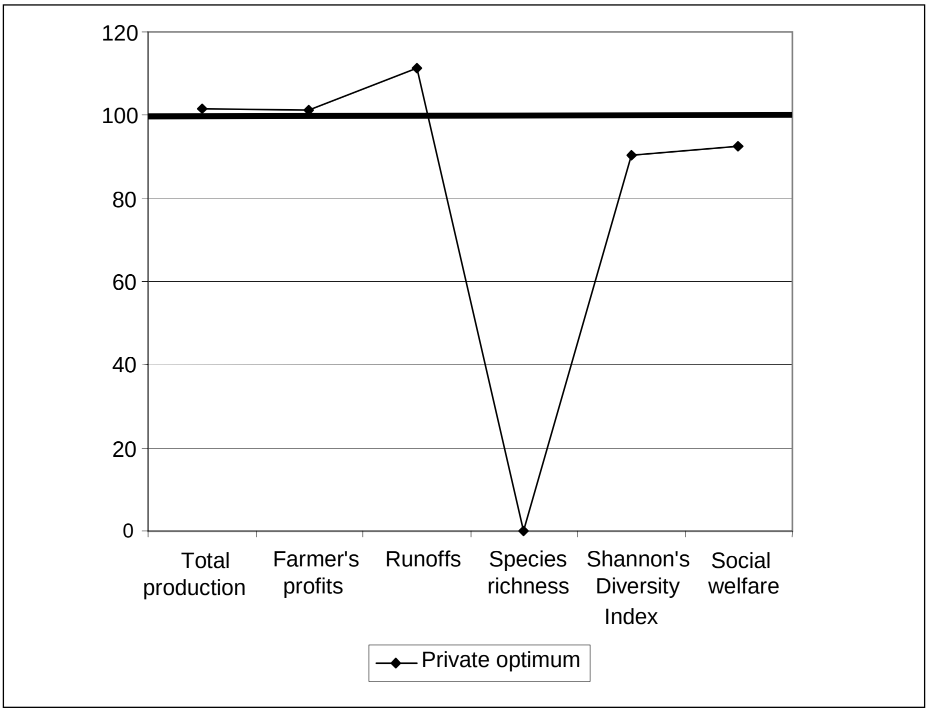

Figure 8. Indicators of multifunctional outputs in the private solution against the benchmark of the social optimum, which is indexed to 100.

fertilizer use and has smaller buffer strips than crop 1. When the damage estimate is increased by 30% the critical land quality increases from 28 to 29 indicating that 1 one hectare more is allocated for crop 1 which is less fertilizer intensive and has larger buffer strips than crop 2. Thus, these results follow the logic of the analytical model and show that the benchmark case of the social optimum is reasonably robust to changes in the valuation estimate.

3.5 Conclusions

In this chapter an analytical framework was developed for analysing multifunctional agriculture as the joint production of a number of commodity and non-commodity outputs. The core of the framework was Lichtenberg's (1989) model of agricultural production with an endogenous input and land allocation choice for two alternative crops, augmented by a description of certain non-commodity outputs of agriculture. Of the non-commodity outputs the study focused on agrobiodiversity, landscape diversity, and nutrient runoffs. Whereas the two former ones represent public good aspects of agriculture, the latter one represents its negative externalities.

The privately optimal land allocation and choice of inputs were solved and compared to the corresponding social optimum. The private optimum results in a higher fertilizer use and smaller size of buffer strips than the socially optimal solution. When compared to the private optimum, the socially optimal land allocation, which is determined also by the marginal valuation of the diversity benefits and runoff damages, shifts more land to crop 1, which uses fertilizers less intensively, has larger buffer strips and thus produces a higher level of diversity.

In the numerical application of the analytical model the private and social optima in the absence of taxes and subsidies were solved, showing that the private solution leads to excessive use of fertilizer, sub-optimal use of buffer strips, and an excessive amount of land devoted to the production of crop 2.

4 Policy design: Uniform and differentiated policy instruments

4.1 Introduction

In Chapter 3 the private solution was compared to the socially optimal way of producing both commodity and non-commodity outputs. The conclusion was that correcting the privately optimal use of inputs and land allocation to reflect socially optimal multifunctional agriculture requires an appropriate use of policy instruments. Thus, the present chapter examines the optimal use of two instruments, a fertilizer tax and a buffer strip subsidy, to guide the private solution towards the social optimum. The choice of the socially optimal fertilizer tax and buffer strip subsidy rates requires that, when maximising the social welfare function, the government knows how farmers react to the tax and subsidy. This reaction is given by the comparative static effects of the basic levels of tax and subsidy on the use of inputs and land allocation. These were already provided in Chapter 3, equations (3.10) and (3.19).

Another question to be studied is whether it is optimal to use uniform or differentiated policy instruments. In the case of differentiated instruments, the tax and subsidy rates are differentiated according to the crop and parcel. Hence, the policy incentives are fine-tuned with respect to heterogeneous land quality conditions. With uniform instruments this is not the case, but a constant per-unit tax on fertilizer and a constant subsidy for buffer strips are implemented.

Given the benchmark of socially optimal multifunctional agriculture which was determined in chapter 3, the socially optimal rates of the policy instruments are solved in the parametric application of the analytical model. The optimal instrument rates are solved from the command optimum to reflect the magnitude of positive and negative externalities per parcel.

The rest of the chapter is organised as follows. Chapter 4.2.1 shortly recalls the private optimum in the presence of basic levels of agri-environmental policy instruments. The characteristics of the first-best policy instruments are examined in Chapter 4.2.2. Chapter 4.2.3 extends the theoretical analysis by considering the modifications for the optimal level of instruments if the government is not free to choose its tax/subsidy policy but has to take into account how the net support to agriculture affects budget revenue and thereby general tax level in economy. Numerical solutions under differentiated and uniform instruments are provided in Chapter 4.3 on the basis of the procedure developed in Chapter 4.2.2. The main conclusions are summarised briefly in Chapter 4.4.

4.2 Optimal level of fertilizer tax and buffer strip subsidy

4.2.1 Comparative statics of fertilizer tax and buffer strip subsidy

Let us recall that the first-order conditions for the farmer's private optimum in the presence of a basic level of fertilizer tax and a basic level of buffer strip subsidy are $\pi^i_{l_i} = p_i f^i_{l_i} - c^* = 0$ and $\pi^i_{m_i} = -(p_i f^i(l_i;q) - c^* l_i) + \lambda - \lambda m_i = 0$, which require that the value of the marginal product of input use equals their respective costs. The optimal fertilizer intensity and buffer strip size differ in every parcel since the productivity of each parcel differs along with q. Moreover, for a given fertilizer tax and buffer strip subsidy rate, the use of fertilizer is greater and the size of the buffer strip is smaller for the crop that is grown on higher quality parcels. Hence, with regard to agri-environmental policy instruments we have the result that for internally homogenous parcels, a higher fertilizer tax decreases fertilizer intensity and increases the size of the buffer strips, and a higher buffer strip subsidy increases the size of the buffer strips, but does not affect fertilizer intensity (see equation (3.10) and Appendix 1). Another point to be borne in mind is the comparative static effects of the policy instruments on land allocation (equation (3.19)): both instruments shift land into the production of the less fertilizer-intensive crop 1.

4.2.2 Characteristics of the first-best policy instruments

Let us recall the optimality conditions (3.23a) and (3.23b) from Chapter 3.3.2

$$SW^i_l = p_i f^i_l - c - d'(z)\frac{\partial v_i}{\partial \bar{l}_i} + k(\cdot)h_{\bar{l}} = 0$$

$$SW^i_m = -p_i f^i(l_i;q) + cl_i - d'(z)\frac{\partial v_i}{\partial m_i} + k(\cdot)h_m = 0$$

These conditions define the first-best policy instruments for multifunctional agriculture. The first-best policy includes the combination of a tax on fertilizers to account for the runoff damages and reduced agrobiodiversity benefits caused by fertilizer use, and a buffer strip subsidy to encourage the establishment of buffer strip areas to reduce runoff damages and increase agrobiodiversity benefits.

The fertilizer tax is a unit tax and one can conclude from (3.23a) that the level of the optimal fertilizer tax depends on the marginal damage of runoff (which is equal over all parcels) and the marginal runoffs from fertilizer use in each parcel. It is then clear that the optimal tax will vary over land quality, because marginal runoffs vary over land quality through variations in l_i and m_i. This means that the fertilizer tax has to be differentiated with respect to parcels and crops. The tax rate should be higher on parcels with higher land quality, since marginal runoffs are greater there due to higher fertilizer intensity and smaller buffer strips. Taking into account the last term in (3.23a), the marginal effect on agrobiodiversity does not change this feature but only tends to increase the tax rate over all parcels. Now the optimal differentiated fertilizer tax ζ can be expressed as the product of a basic tax level, t, common to both crops, and crop and parcel specific coefficient, γ, $\zeta_i^j = \gamma_i^j t$, where j refers to parcels and i to crops. The after-tax price of the fertilizer is thus $c^* = c(1+\zeta_i^j)$.

Similar reasoning also holds for the buffer strip subsidy. From (3.23b) one can conclude that the subsidy rate depends on the marginal effect of the buffer strip on agrobiodiversity (the last term), as well as on the value of the marginal runoff reduction (the second last term). Note that the marginal effect on agrobiodiversity is constant over parcels, while the marginal effect on runoffs varies. The buffer strip subsidy also varies across parcels and crops, because l_i and m_i do so. Moreover, the optimal buffer strip subsidy must be decreasing in m. This reflects the decreasing ability of wider buffer strips to further reduce runoffs and increase species diversity. Now the optimal differentiated buffer strip subsidy can be specified as $b(m_i) = \kappa_i^j(\lambda - \frac{1}{2}\lambda m_i)m_i$, where λ is the basic level and κ is the crop and parcel specific coefficient. The marginal subsidy is $b'(m_i) = \kappa_i^j \lambda(1-m_i)$.

Hence, these findings can be collected in

Proposition 2. *The promotion of multifunctional agriculture under heterogeneous land quality requires the combined use of differentiated instruments to achieve the first-best solution:*

i) a fertilizer tax that equalizes the value of marginal product of fertilizer over its unit price to the marginal runoff damage and marginal runoffs of fertilizer in each parcel, as well as its marginal effect on agrobiodiversity.

ii) a buffer strip subsidy that equalizes the net loss of income due to decreased production to marginal reductions in runoff damages and marginal increase in diversity benefits.

This Proposition is in line with Lichtenberg (2000), which points out in a different setting that, in the presence of heterogeneous land quality, it will be socially optimal to use differentiated instruments if environmental quality depends on land quality. The requirement for the use of differentiated instruments in our case arises from the fact that the non-commodity outputs indirectly depend on land quality through the size of the buffer strips and the amount of fertilizer used.

Uniform policy instruments that are undifferentiated with respect to land quality fail to give the right incentives at both the intensive (input use) and extensive (land allocation) margins in this case (for a further discussion on uniform versus differentiated instruments see e.g. Lichtenberg 2000). It should be noted, however, that uniform instruments can also be optimal with heterogeneous land quality. In our case, a uniform fertilizer tax would be optimal if nutrient runoffs depended only on the total fertilizer use, and not (even indirectly) on land quality.

However, the implementation of differentiated instruments may be difficult and costly to administer in practice. For instance, the implementation of differential fertilizer taxes at the point of sale would hardly be feasible as the same fertilizer formulations are used for different crops and under heterogeneous conditions (see e.g. Lichtenberg 2000). Therefore, societies often have to search for second-best solutions, such as uniform policy instruments.

The procedure developed here will be used to solve the differentiated rates of the policy instruments in the numerical solutions in Chapter 4.3. Before that, however, a short digression is taken to analyse how government budget revenue considerations affect the optimal level of the instruments.

4.2.3 Optimal instruments and net support to agriculture

In Chapter 4.2.2 the first-best instruments for multifunctional agriculture were characterised. The optimal instrument levels were derived by implicitly assuming that the government can freely choose its multifunctional tax/subsidy policy without any budget revenue requirements. This is not always the case as the government has to run, at least in the long run, a balanced budget. Thus, this section analyses how the optimal levels of the instruments are modified when net support to agriculture affects government budget revenue requirements and thereby general tax level in economy. Simplest way to postulate these effects is through consumers' income tax rate which is affected by net support to agriculture.

To introduce the consumers' surplus it is assumed that the preferences of the representative consumer define an additively separable, quasi-linear utility function. Denoting the consumption of crops by Y_1 and Y_2, and the consumer's money income by *I*, the indirect utility function is expressed as $U^* = (1-\varsigma(t,\lambda))I + u_1(p_1(Y_1)) + u_2(p_2(Y_2))$, where $p_i(Y_i)$ is the demand for both crops (solved from the constrained utility maximization problem), and the term $(1-\varsigma\ (\tau,\lambda))$ captures the effects of the multifunctional tax/subsidy policy on the income tax level of the economy.[17] The dependence of the income tax rate on the net support to multifunctional agriculture reflects the fact that the choice of multifunctional tax and subsidy policy affects the government's budget and thus the needs to finance it. This effect is described through the representative consumer's income tax rate, which is increasing in the buffer strip subsidy and decreasing in the fertilizer tax. The producers' surplus is defined by an indirect profit function in the presence of a fertilizer tax and a buffer strip subsidy.[18]

The government chooses the basic levels of fertilizer tax and buffer strip subsidy so as to maximise

$$\text{(4.1)} \quad \max_{\{t,\lambda\}} SW = U^* + \int_0^1 \left[\pi_1^* L_1^* + \pi_2^*(1-L_1^*)\right] g(q)dq - d(z) + k(L_1, L_2)h(m,\bar{l})$$

The following first-order conditions characterise the optimal fertilizer tax and buffer strip subsidy:

$$\text{(4.2a)} \quad \begin{aligned} SW_t &= \underbrace{\Pi_t^*}_{-} + \underbrace{U_t^*}_{+} - d'(\cdot)\underbrace{\left[\frac{\partial v_1}{\partial t}L_1 + \frac{\partial v_2}{\partial t}L_2 + (v_1 - v_2)\frac{\partial L_1}{\partial t}\right]}_{-} \\ &+ \underbrace{\left[k_1 - k_2\right]}_{?}\underbrace{\frac{\partial L_1}{\partial t}}_{+}h(m,\bar{l}) + k(L_1, L_2)(h_m \underbrace{\frac{\partial m}{\partial t}}_{+} + h_{\bar{l}}\underbrace{\frac{\partial \bar{l}}{\partial t}}_{+}) = 0 \end{aligned}$$

---

[17] The use of quasi-linear utility function is convenient, because it implies that the income effect is zero and allows one to focus on the substitution effects only. Quasi-linear utility function is widely used in the public economics and in environmental economics (see e.g. Lichtenberg 2000, and Cornes and Sandler 1996 for more general discussion).

[18] This follows the conventional approach in public economics (see e.g. Persson and Tabellini 1990). The optimal fertilizer tax and buffer strip subsidy are solved as a Stackelberg game between the farmer and the government. The government acts as a Stackelberg leader, announces its agri-environmental tax and subsidy policy and commits itself credibly to it. The farmer then chooses his use of inputs and land allocation.

(4.2b)
$$SW_\lambda = \underbrace{\Pi_\lambda^*}_{+} + \underbrace{U_\lambda^*}_{-} - d'(\cdot)\left[\underbrace{\frac{\partial v_1}{\partial\lambda}L_1 + \frac{\partial v_2}{\partial\lambda}L_2 + (v_1 - v_2)\frac{\partial L_1}{\partial\lambda})}_{-}\right]$$

$$+\underbrace{[k_1 - k_2]}_{?}\underbrace{\frac{\partial L_1}{\partial\lambda}}_{+}h(m,\bar{l}) + k(L_1, L_2)(\underbrace{h_m\frac{\partial m}{\partial\lambda}}_{+} + \underbrace{h_{\bar{l}}\frac{\partial\bar{l}}{\partial\lambda}}_{+}) = 0$$

where

$$\Pi_t^* = -\int_0^1 [c(1-m_1)l_1L_1 - c(1-m_2)l_2(1-L_1)]g(q)dq < 0,$$

$$\Pi_\lambda^* = \int_0^1 [(1-\tfrac{1}{2}m_1)m_1L_1 + (1-\tfrac{1}{2}m_2)m_2(1-L_1)]g(q)dq > 0,$$

$$U_t^* = -\varsigma_t I > 0,\ U_\lambda^* = -\varsigma_\lambda I < 0$$

According to (4.2a), the optimal fertilizer tax rate is found by equating the direct economic loss of the farmer to the marginal benefits from runoff reduction and improvement in diversity, as well as to the lower income taxation of consumers. Looking more closely at the effects of tax on diversity, we can see that, while agrobiodiversity improves, landscape diversity may decrease.

The optimal buffer strip subsidy is set so as to equalise the disutility from higher income taxation to the sum of the direct economic gain to the farmer and the marginal benefit from runoff reduction and diversity promotion. While the run-off unambiguously decreases, the effect on diversity is ambiguous, because landscape diversity may or may not improve.

These optimality conditions illustrate well the complexities involved in the policy design for multifunctional agriculture. Because both input intensity and land allocation are endogenous in this case, their changes may easily counter-affect each other, as is shown in both conditions. Moreover, consumers must be willing to pay for agricultural multifunctionality.[19]

[19] Allowing for endogenous crop prices will not change these results much. The price effects of the fertilizer tax and buffer strip subsidy counter-affect their direct effects, which implies higher runoff damages and lower diversity. Therefore, the optimal rates of the policy instruments are higher than under exogenous prices.

To sum up, we have

Corollary. *Accounting for the effects of net support to agriculture for budget revenue requirements and thereby general tax level in economy modifies the first-best policy instruments: the basic level of fertilizer tax increases and the basic level of buffer strip subsidy decreases.*

On the basis of Chapter 4.2.2 we now turn to solving the socially optimal rates of the policy instruments in the parametric application of the analytical model.

4.3 Numerical solutions

In Chapter 3 the basic features of socially optimal multifunctional agriculture were determined based on Finnish data. Using the socially optimal provision of commodity and non-commodity outputs as the benchmark the optimal fertilizer tax (nitrogen tax) and buffer strip subsidy rates that can sustain this optimality can now be designed. Thus, the command optimum allows us to define the rates for a differentiated fertilizer tax and buffer strip subsidy for each parcel so as to maximise the target function in equation (3.22). Two alternative cases of uniform agri-environmental policy will also be analysed. In the first case (semi-uniform instruments) the agri-environmental policy instruments are crop-specific but uniform with respect to parcel and, thus, land quality. In the second case (uniform instruments) the instruments are uniform with respect to both the crop and parcel.

The farmer's short-run profits, a parametric version of (3.3), per parcel for crop i in the presence of a fertilizer tax and a buffer strip subsidy are given by

$$(4.3)\quad \pi^i = p_i(1-m_i)\left[a_i + \alpha_i l_i + \beta_i l_i^2\right] - c^*(1-m_i)l_i + (\lambda_i - \tfrac{1}{2}\omega_i\lambda_i m_i)m_i$$

where λ_i and ω_i define the non-linear, decreasing buffer strip subsidy payment, and a_i and α_i are defined by (3.26).

The solution technique is as follows (see Appendix 5 for sample equations of the calibrated model). First, the first-order conditions from equations (A5.3a) and (A5.3b) in Appendix 5 are taken with respect to fertilizer use and buffer strips. Secondly, the socially optimal input use in each parcel is inserted into the first-order conditions, and the optimal fertilizer tax and buffer strip subsidy are solved in each parcel for both crops.

As the analytical model suggests, the first-best policy consists of 60 pairs of policy instruments that are differentiated with respect to both the crop and parcel.

In the case of semi-uniform instruments, in order to have an interior solution for buffer strips also in the highest quality parcels, the optimal quadruple of instruments have been solved for the last parcel, which in our case is the 60th hectare. This yields the crop-specific instruments that are uniform with respect to parcels. Solving the optimal level of the policy instruments for the parcel with the highest land quality means that the marginal revenue of buffer strips is too high at lower quality parcels, resulting in too large buffer strips from the social point of view. The optimal levels of the instruments for the last parcel are as follows: the fertilizer tax is 23.6% for crop 1 and 27.3% for crop 2, and the buffer strip subsidy is FIM 2,810 for crop 1 and FIM 2,957 for crop 2.

In the case of uniform instruments, a fertilizer tax of 27.3% and a buffer strip subsidy of FIM 2,957 are applied for both crops in each parcel. Thus, the instruments are determined on the basis of the most profitable crop (crop 2) at the highest possible land quality.

The results are reported in Tables 7 and 8. For comparison, the results of the private and social optimum in the absence of intervention from Chapter 3 are also given. Appendix 3 provides some details of the numerical solutions for selected parcels. Table 7 presents the land allocation, average use of inputs and average production per hectare under alternative solutions.

Table 7 shows that the differentiated instruments result in exactly the same land allocation and input use as the social optimum. Semi-uniform instruments, that is, instruments that are uniform with respect to land quality but differentiated between crops, allocate more land into the production of crop 1. This shift in

Table 7. Land allocation, average input use and average production under alternative solutions.

	Land allocation, ha		Nitrogen use, kg		Buffer strip size, ha		Production, kg/ha	
	Crop 1	Crop 2	Crop 1	Crop 2	Crop 1	Crop 2	Crop 1	Crop 2
Private optimum	23	37	129.5	153.0	-	-	4863	4762
Social optimum	27	33	122.9	141.5	0.0086	0.0081	4787	4663
Differentiated instruments	27	33	122.9	141.5	0.0086	0.0081	4787	4664
Semi-uniform instruments	30	30	115.7	136.8	0.0670	0.0407	4517	4529
Uniform instruments	27	33	107.7	136.1	0.1253	0.0455	4217	4490

land allocation is mainly caused by the fact that the amount of the buffer strip subsidy is determined on the basis of the highest land quality. As a result, the subsidy overcompensates the farmer for the establishment of buffer strips in low quality parcels and, as the analytical model predicts, increases the relative profitability of crop 1. The high marginal revenue for establishing buffer strips results in larger buffer strips for both crops and thus in lower fertilizer use and production per parcel than was the case under differentiated policy instruments.

Uniform instruments, which are uniform with respect to both land quality and crop, again result in the socially optimal land allocation. The difference in land allocation compared to the semi-uniform instruments is mainly due to the application of a higher fertilizer tax rate for crop 1, which shifts more land into the production of crop 2. However, the average buffer strip size increases for both crops, causing fertilizer use and production per parcel to be lower than under semi-uniform instruments. This means that average input use and production are further away from the social optimum than in the case of semi-uniform instruments.

Table 8 presents the total short-run profits, total nitrogen runoffs, total number of floral species, Shannon Diversity Index (*SHDI*), and social welfare (*SW*) under alternative solutions.

As the second and third row in Table 8 show, differentiated instruments yield the social optimum. The only difference between these two solutions concerns profits that are lower under differentiated first-best instruments than under the command optimum. The average nitrogen tax rate is 23.5% for crop 1 and 27.2% for crop 2, and the resulting average cost increase per hectare is FIM 172.1 and FIM 229.3, respectively. The average buffer strip subsidy is FIM 22.8 for crop 1 and FIM 64.9 for crop 2 (see Appendix 3 for details). Thus, the first-best instruments result in negative net support (subsidy less tax) for the

Table 8. Economic, environmental and social welfare outcomes under alternative solutions.

	Farmer's profits, FIM	Runoffs, kg	Species richness	*SHDI*	*SW*
Private solution	176 481	1375	-	0.67	163 418
Private optimum	176 481	1375	-	0.67	163 418
Social optimum	174 562	1235	75	0.74	176 739
Differentiated instruments	164 415	1235	75	0.74	176 739
Semi-uniform instruments	164 041	1039	90	0.90	172 900
Uniform instruments	163 726	947	93	0.96	169 771

farmer and, accordingly, in decreased profits. The total amount of the negative net support is FIM 10,153, or FIM 169 per hectare.

Semi-uniform instruments result in even lower private profits than differentiated instruments. This is mainly due to the higher share of buffer strips and the resulting lower total production. Although semi-uniform instruments are stronger than differentiated ones in the provision of environmental non-commodity outputs, social welfare nevertheless decreases as commodity production is sub-optimal from the societal viewpoint. In other words, semi-uniform instruments result in the overprovision of non-commodity outputs and undersupply of commodity outputs. In this case as well the net support is negative, amounting to FIM 2,584 (FIM 43 per hectare).

The phenomenon described above is even stronger in the case of uniform policy instruments. The provision of environmental non-commodity outputs is further strengthened but, as the total commodity production is further reduced, so is the total social welfare. Although the net support is now positive, amounting to

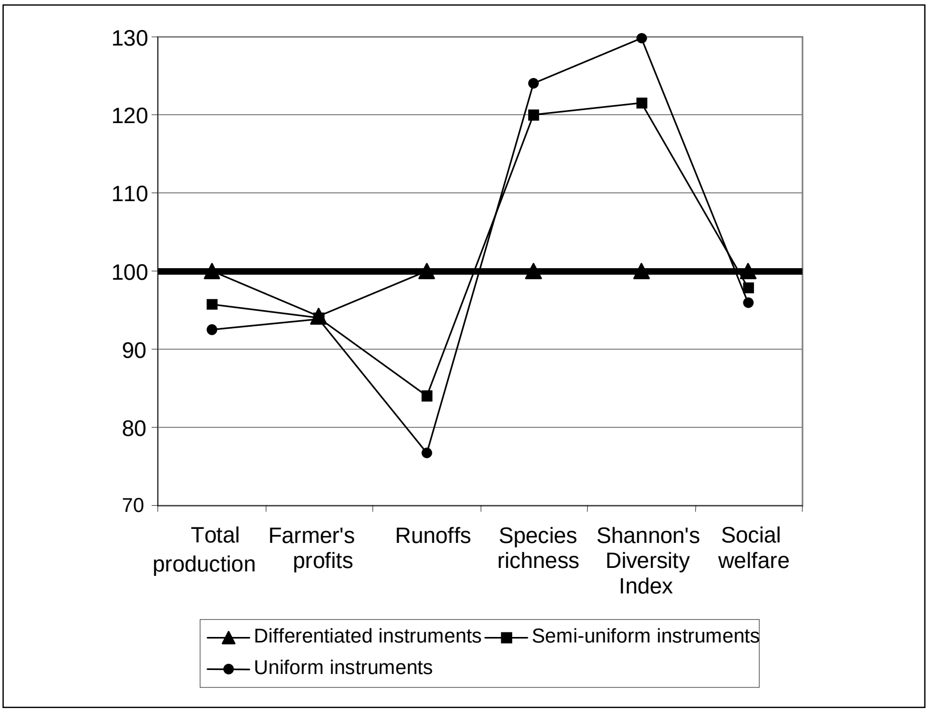

Figure 9. Indicators of multifunctional outputs in the first-best and second-best policy solutions against the benchmark of the social optimum, which is indexed to 100.

FIM 1,649 or FIM 27.5 per hectare, due to the reductions in production the farmer's profits are lower than under differentiated or semi-uniform instruments.

The reduction in social welfare compared to the first-best instruments is FIM 3,839 (FIM 64 per hectare) with semi-uniform instruments and FIM 6,968 (FIM 116 per hectare) with uniform instruments.

Figure 9 summarises the first-best and second-best policy instruments in terms of the analysed commodity and non-commodity outputs and the resulting level of social welfare. The policy solutions are presented against the benchmark of the social optimum, which is indexed to 100.

4.4 Conclusions

This chapter was concerned with the optimal use of a fertilizer tax and buffer strip subsidy to guide the farmer's private solution towards the socially optimal way of producing commodity and non-commodity outputs. The optimal fertilizer tax depends on the marginal runoff from fertilizer use and its marginal damage, as well as on the marginal effect of fertilizer use on agrobiodiversity. The optimal buffer strip subsidy depends on the marginal runoff reduction achieved and its marginal value, as well as on the marginal effect of buffer strips on agrobiodiversity. All these factors are constant over parcels except the marginal runoffs from fertilizer use and marginal runoff reduction from buffer strips. These vary over land quality because of variations in fertilizer intensity and buffer strip width, and make the optimal fertilizer tax and buffer strip subsidy vary over parcels and crops as well. Thus, the promotion of multifunctional agriculture under heterogeneous land quality requires the combined use of differentiated instruments to achieve the first-best solution. Taking into account net support to agriculture modifies the first-best instruments so that the basic level of fertilizer tax increases and the basic level of buffer strip subsidy decreases.

In the numerical application the differentiated first-best tax rates and subsidy levels were solved, yielding 60 pairs of crop- and parcel-specific fertilizer taxes and buffer strip subsidies. For comparison, uniform policy instruments were also applied. The social welfare difference between the first-best differentiated instruments and second-best uniform instruments is FIM 64 (that is, 2.17%) per hectare in the case of semi-uniform instruments and FIM 116 per hectare in the case of fully uniform instruments.

It is worth noting, however, that in practice differentiated instruments are unlikely to be specified for every unit of land. Rather, land would be classified into several groups according to agricultural productivity or environmental sensitivity and policy instruments would be differentiated between these groups.

5 Income support measures and multifunctionality

5.1 Introduction

As already noted, the concept of multifunctionality and its use as a basis for practical policy-making has raised conflicting views among the WTO members. The proponents of multifunctionality fear that further reductions and constraints on domestic agricultural support would reduce their ability to pursue non-commodity objectives. This is opposed by, for instance, countries in the Cairns Group, who argue that multifunctionality is used as a pretext for maintaining high levels of production-related and trade-distorting agricultural support.[20]

Consequently, no consensus on the appropriate policy response for addressing non-commodity outputs has been reached. For the Cairns Group the Green Box measures, that is, measures such as environmental programmes and decoupled income support that do not distort production decisions and trade or do so only minimally, represent effective and universal means for addressing multi-functionality. Countries with high production costs, however, fear that the Green Box compatible measures may not be sufficient to sustain and enhance the multifunctional character of agriculture. Thus, these countries are striving to expand the Green Box to contain some production-linked support in order to address multifunctional non-commodity outputs effectively.[21]

This chapter examines whether various farm income support measures that are used in the European Union countries, including Finland, promote multifunctional agriculture. The analytical model in Chapter 3 offers a benchmark for the socially optimal provision of both commodity and non-commodity outputs.

The relative merits of alternative income support measures in promoting multi-functional outputs will be compared in a parametric model that is based on the analytical model. The basic income support measures included in the analysis are producer price support and acreage subsidy. The so-called environmental cross-compliance measures are also analysed, i.e. measures in which the eligi-

[20] The Cairns Group includes major agricultural exporters from both developed and developing countries: Argentina, Australia, Brazil, Canada, Chile, Colombia, Indonesia, Malaysia, New Zealand, the Philippines, Thailand, Uruguay, and South Africa.

[21] Even though it is the link to the WTO that has raised the issue of multifunctionality and farm income support to the forefront in the international debate, the present chapter is concerned with domestic policy design and domestic distortions. Therefore – although domestic policies do affect production levels and thus trade flows – international trade issues are explicitly analysed in neither the analytical nor the empirical part of the chapter.

bility for farm income support is contingent upon the farmer's undertaking of environmental activities, such as limiting fertilizer use and establishing buffer strips. Thus, we mainly adopt the "red ticket approach" to environmental cross-compliance, where eligibility for agricultural support is made contingent upon a farmer's attainment of given environmental standards (for a further discussion see e.g. Christensen and Rygnestad 2000).

The rest of the chapter is organised as follows. Chapter 5.2 analyses the private optimum in the presence of income support measures. Chapter 5.3 provides the numerical solutions for alternative farm income support measures and environmental cross-compliance mechanisms in promoting multifunctional outputs by using Finnish data against the benchmark of socially optimal multifunctional agriculture. The main conclusions are summarised in Chapter 5.4.

5.2 Private optimum in the presence of income support measures

5.2.1 Input use

We start by developing the per parcel short-run profit function under alternative income support measures. The farmer takes the prices of crops (p_i) and fertilizer (c) as given. The government intervenes in agriculture through three policy instruments. First, it pays a price support τ so that the unit price of cereals is $p_i^* = p(1+\tau)$. Second, cultivated arable land is entitled to a unit acreage subsidy s, and third, a subsidy, $b(m_i)$, is paid for buffer strip areas. Let us assume that the buffer strip subsidy is decreasing in the size of the buffer strip, reflecting the decreasing ability of the buffer strip to further reduce nutrient runoff and increase species diversity. The parametric version of the concave buffer strip subsidy is $\left(\lambda - \frac{1}{2}\lambda m_i\right)m_i$. The farmer's problem is to choose the inputs, l_i and m_i so as to maximise the profit per parcel:

(5.1) $$\max_{\{l_i, m_i\}} \pi^i = p_i^*(1-m_i) f^i(l_i;q) - c(1-m_i)l_i + s(1-m_i) + (\lambda - \tfrac{1}{2}\lambda m_i)m_i$$

for i =1,2

The first-order conditions for the optimal solution are

(5.2a) $$\pi_{l_i}^i = p_i^* f_{l_i}^i - c = 0$$

(5.2b) $$\pi^i_{m_i} = -p^*_i f^i(l_i;q) + cl_i - s + \lambda - \lambda m_i = 0$$

and require that the value of the marginal product of input use equals their respective costs.

The second-order conditions are given in equations (5.3a) to (5.3d)

(5.3a) $$\pi^i_{l_i l_i} = p^*_i f^i_{l_i l_i} < 0$$

(5.3b) $$\pi^i_{m_i m_i} = -\lambda < 0$$

(5.3c) $$\pi^i_{l_i m_i} = 0 = \pi^i_{m_i l_i}$$

(5.3d) $$\Delta = \pi_{ll}\pi_{mm} - \pi_{lm}{}^2 > 0$$

where Δ is the determinant of the Hessian matrix of the second-order partial derivatives. Since the principal minors $|H_1| < 0$ and $|H_2| > 0$, the Hessian matrix is negative definite and the solution maximises profit (see e.g. Chiang 1984). The comparative statics can be solved from the first-order conditions by differentiating them with respect to exogenous parameters and applying the Cramer's Rule (see also equations (A4.4a) and (A4.4b) in Appendix 4). The results for cereal price, fertilizer price, and buffer strip subsidy were already reported in Chapter 3, but the results for price support and acreage subsidy are new.

The effects of price support are given by (5.4a) and (5.4b)

(5.4a) $$\frac{dl}{d\tau} = -\Delta^{-1}\left\{p_i f^i_{l_i} \pi_{mm}\right\} > 0$$

(5.4b) $$\frac{dm}{d\tau} = \Delta^{-1}\left\{p_i f^i(l_i;q)\pi_{ll}\right\} < 0$$

and the effects of an acreage subsidy are given in equations (5.5a) and (5.5b)

(5.5a) $$\frac{dl}{ds} = 0$$

(5.5b) $$\frac{dm}{ds} = \Delta^{-1}\left\{\pi_{ll}\right\} < 0$$

The comparative statics of the model are condensed to (see also Appendix 4 for details)

(5.6) $$l_i = l_i(\underset{+}{\tau}, \underset{+}{p_i}, \underset{-}{c}, \underset{0}{s}, \underset{0}{\lambda}); \quad m_i = m_i(\underset{-}{\tau}, \underset{-}{p_i}\, \underset{+}{c}, \underset{-}{s}, \underset{+}{\lambda})$$

Hence, as regards pure income support measures we have

Result 3. *Producer price support increases fertilizer intensity and decreases the size of buffer strips. Acreage subsidy does not affect fertilizer intensity but decreases the size of the buffer strips.*

5.2.2 Land allocation

The farmer maximises the sum of restricted profit functions π_i^*, $i = 1,2$ by allocating his land between both crops, that is, $\max\limits_{L_1(q)} \int_0^1 \left[\pi_1^* L_1(q) + \pi_2^*(1 - L_1(q))\right] g(q)dq$. The first-order condition for the optimal land allocation is

(5.7) $$\pi_1^*(q, \tau, p_1, c, s, \lambda) - \pi_2^*(q, \tau, p_2, c, s, \lambda) \leq 0$$

As in Chapter 3, this first-order condition leads to a corner solution for every homogenous parcel in the given acreage with differential land quality. Thus, if $\pi_1^*(q) > (<) \pi_2^*(q)$ then all land of quality q is allocated for crop 1 (crop 2). However, by assumption the restricted profits are higher for crop 1 in lower quality parcels whereas in the highest land quality $\pi_2^*(1, \tau, p_2, c, s, \lambda) > \pi_1^*(1, \tau, p_1, c, s, \lambda)$ they are higher for crop 2, and restricted profits as a function of land quality increases more rapidly for crop 2 for all land of quality q, that is $\pi_q^2 > \pi_q^1$. These assumptions ensure that there is a single unique q_1 and each crop will be cultivated on a unique, compact range of land qualities.

The comparative statics of the exogenous parameters on land allocation can be solved as follows. To solve for $\frac{\partial q_1}{\partial \theta}$, equation (5.7) which defines the critical value q_1 is totally differentiated to get

(5.8) $$\begin{aligned} &\pi_{p_1}^1 dp_1 + \left[\pi_c^1 - \pi_c^2\right] dc - \pi_{p_2}^2 dp_2 + \left[\pi_\lambda^1 - \pi_\lambda^2\right] d\lambda + \left[\pi_s^1 - \pi_s^2\right] ds \\ &+ \left[\pi_\tau^1 - \pi_\tau^2\right] d\tau + \left[\pi_{q_1}^1 - \pi_{q_1}^2\right] dq_1 = 0 \end{aligned}$$

From equation (5.8) an appropriate expression for each $\frac{\partial q_1}{\partial \theta}$ can be developed.

Now the effects of uniform price support and acreage subsidy on land allocation between the crops are as follows.

The effects of uniform price support on land allocation are

(5.9a) $$\frac{\partial L_1}{\partial \tau} = \frac{-g(q_1)\overbrace{\left[(p_1(1-m_1)f^1(\cdot)) - (p_2(1-m_2)f^2(\cdot))\right]}^{(-)}}{\Delta} < 0$$

(5.9b) $$\frac{\partial L_2}{\partial \tau} = \frac{g(q_1)\overbrace{\left[(p_1(1-m_1)f^1(\cdot)) - (p_2(1-m_2)f^2(\cdot))\right]}^{(-)}}{\Delta} > 0$$

where $\Delta = \left[\pi^1_{q_1} - \pi^2_{q_1}\right] < 0$ because $\pi^1 > \pi^2$ for $q < q_1$ and $\pi^1 < \pi^2$ for $q \geq q_1$.

The effects of uniform acreage subsidy are

(5.10a) $$\frac{\partial L_1}{\partial s} = \frac{-g(q_1)\overbrace{\left[(1-m_1) - (1-m_2)\right]}^{(-)}}{\Delta} < 0$$

(5.10b) $$\frac{\partial L_2}{\partial s} = \frac{g(q_1)\overbrace{\left[(1-m_1) - (1-m_2)\right]}^{(-)}}{\Delta} > 0$$

The comparative static effects of the exogenous parameters on land allocation are summarised in (5.11)

(5.11) $$L_1 = L_1(\underset{+}{p_1}, \underset{-}{p_2}, \underset{+}{c}, \underset{-}{\tau}, \underset{-}{s}, \underset{+}{\lambda}) \text{ and } L_2 = L_2(\underset{-}{p_1}, \underset{+}{p_2}, \underset{-}{c}, \underset{+}{\tau}, \underset{+}{s}, \underset{-}{\lambda})$$

With regard to farm income support measures we have

Result 4. *Producer price support and acreage subsidy shift land into the production of the more fertilizer-intensive crop 2.*

5.3 Numerical solutions

Given the benchmark of socially optimal multifunctional agriculture established in Chapter 3, this chapter focuses on alternative income support measures and environmental cross-compliance mechanisms in a numerical application of the analytical model. Using Finnish data, the relative efficiency of various income support measures and cross-compliance mechanisms in promoting multifunctional outputs is compared in terms of input use, land allocation, production, short-run profits, nitrogen runoffs, and diversity. For the latter two measures are offered, floral species richness which proxies agrobiodiversity, and the Shannon Diversity Index which captures landscape diversity . Moreover, government budget outlays and social welfare under alternative policies are reported.[22]

The representative farmer's short-run profits per parcel for crop i in the presence of income support measures are given by a parametric version of (5.1):

$$(5.12)\quad \pi^i = p_i^*(1-m_i)\left[a_i + \alpha_i l_i + \beta_i l_i^2\right] - c(1-m_i)l_i + s(1-m_i) + (\lambda - \tfrac{1}{2}\lambda m_i)m_i$$

$$\text{for } i = 1,2$$

Six alternative policies involving income support are compared to the private optimum in the absence of government intervention as well as to the socially optimal multifunctional agriculture determined by the command optimum. The alternative income support measures and cross-compliance mechanisms analysed are listed below. Policies 5 and 6 reflect actual agricultural and agri-environmental policies for cereals in Southern Finland in 1999.

- In *policy 1 (pure acreage subsidy)*, an acreage subsidy (CAP compensation payment) of FIM 1,100/ha is paid.
- *Policy 2 (pure price support)* includes price support of FIM 0.39/kg for crop 1 (barley) and FIM 0.44/kg for crop 2 (wheat), which corresponds to EU intervention prices before the CAP reform in 1992.
- *Policy 3 (acreage subsidy with cross-compliance)* consists of an acreage subsidy of FIM 1,100/ha with cross-compliance,

[22] Social welfare calculations for these alternative policy solutions assume that these support payments are financed wholly by Finnish government. In practice, however, CAP compensation payments are financed by the EU, and agri-environmental support is also partly financed by the EU. Thus, policy solutions involving these transfers would be ”artificially” high in terms of social welfare if we do not assume that payments are totally paid by Finnish government. This assumption applies for price support as well.

that is, with a mandatory 3-metre-wide buffer strip and nitrogen fertilizer limits of 90 kg/ha for crop 1 and 100 kg/ha for crop 2.

- *Policy 4 (price support with cross-compliance)* consists of price support of FIM 0.39/kg for crop 1 (barley) and FIM 0.44/kg for crop 2 (wheat) with cross-compliance, that is, with a mandatory 3-metre-wide buffer strip and nitrogen fertilizer limits of 90 kg/ha for crop 1 and 100 kg/ha for crop 2.
- *Policy 5 (actual policy I)* includes an acreage subsidy of FIM 1,100/ha and agri-environmental support of FIM 1053/ha with cross-compliance. The environmental conditions attached to the agri-environmental support are a mandatory 3-metre-wide buffer strip and nitrogen fertilizer limits of 90 kg/ha for crop 1 and 100 kg/ha for crop 2. In addition, a buffer zone subsidy of FIM 3,610/ha is available for those who establish wider buffer strips on a voluntary basis (minimum width 15 m).
- *Policy 6 (actual policy II)* is otherwise similar to policy 5 but the buffer zone subsidy is not available.

Table 9 reports the additional parameter values related to these policies. (See also Table 4 for a listing of other parameters in the model).

Table 10 reports the land allocation, average input use and average production, and Table 11 the effects of alternative policies on short-run profits, nitrogen runoffs, floral species richness, and the Shannon Diversity Index (*SHDI*). Note that, instead of the production per hectare for each crop, the total production is given in order to show how alternative income support policies and cross-compliance mechanisms affect the level of production.

Table 9. Additional parameter values (see also Table 4).

Parameter	*Symbol*	*Value*
Price of barley with support	p_1	FIM 1.12/kg
Price of barley without support	p_1	FIM 0.73/kg
Price of wheat with support	p_2	FIM 1.27/kg
Price of wheat without support	p_2	FIM 0.83/kg
Buffer zone subsidy	λ	FIM 3610/ha
Acreage subsidy for crop 1	s_1	FIM 1100/ha
Acreage subsidy for crop 2	s_2	FIM 1100/ha

Note: Prices are from 1999 (FIM 1= 0.1682 euro).

Table 10. Land allocation, average input use and total production under alternative solutions.

	Land allocation, ha		Nitrogen use, kg		Buffer strip size, ha		Production, kg
	Crop 1	Crop 2	Crop 1	Crop 2	Crop 1	Crop 2	
Private optimum	23	37	129.5	153.0	-	-	288 038
Social optimum	27	33	122.9	141.5	0.00868	0.0081	283 134
Pure acreage subsidy	23	37	129.5	153.0	-	-	288 038
Pure price support	17	43	137.6	166.2	-	-	289 575
Acreage subsidy with cross-compliance	25	35	89.5	99.4	0.00600	0.00600	250 025
Price support with cross-compliance	23	37	89.5	99.4	0.00600	0.00600	249 094
Actual policy I	25	35	64.5	78.5	0.28300	0.21530	190 250
Actual policy II	25	35	89.5	99.4	0.00600	0.00600	250 025

Table 11. Economic and environmental outcomes under alternative solutions.

	Farmer's profits, FIM	Runoffs, kg	Species richness	*SHDI*
Private optimum	176 481	1375	-	0.67
Social optimum	174 562	1235	75	0.74
Pure acreage subsidy	242 481	1375	-	0.67
Pure price support	298 388	1536	-	0.60
Acreage subsidy with cross-compliance	228 483	924	73	0.72
Price support with cross-compliance	267 122	927	73	0.70
Actual policy I	268 174	312	104	1.23
Actual policy II	291 284	924	73	0.72

In the first policy alternative the farmer is entitled to a unit acreage subsidy of FIM 1,100 per hectare. The effects of the acreage subsidy closely correspond to the private market solution, since the use of inputs and land allocation are exactly the same in these two solutions; only the farmer's profits are higher under policy 1 due to the subsidy. Thus, with regard to promoting multifunctional outputs, the acreage subsidy results in exactly the same solution as the case where the government does not intervene in agriculture.

Under the second policy, producer price support is paid for both crops. The price support shifts land into the production of the more profitable and fertilizer-intensive crop, which is crop 2 in the model. Of all the alternative policies, price support results in the highest level of production and short-run profits, but also in the highest runoffs and the lowest value for diversity. Thus, price support performs even worse than an acreage subsidy in terms of promoting the environmental dimension of multifunctionality.

In the third policy alternative environmental cross-compliance is attached to the acreage subsidy so that in order to be eligible for the acreage subsidy the farmer has to establish a mandatory 3-metre-wide buffer strip for all parcels and limit his nitrogen fertilizer application to 90 kg/ha for crop 1 and 100 kg/ha for crop 2. The comparison of this solution to the pure acreage subsidy (policy 1) demonstrates that cross-compliance schemes can provide a sufficient means for addressing the environmental dimensions of multifunctional agriculture. Production and profits are lower than in the case of pure acreage subsidy but environmental outcomes are clearly better.

In the fourth policy solution, in order to be eligible for producer price support the farmer has to fulfil the same environmental criteria as in policy 3 discussed above. Profits and production are now clearly lower than under pure price support. Again, it can be seen that environmental criteria coupled with price support clearly result in better environmental quality than pure price support. In terms of environmental outcomes this solution also corresponds closely to the case of acreage subsidy coupled with environmental cross-compliance (policy 3).

In the case of policy 5, a buffer zone subsidy of FIM 3,610/ha is paid for the farmer for the establishment of wider buffer strips on a voluntary basis. Specifically because of the establishment of these wider buffer strips, this solution results in smaller cultivated area and thus in the lowest level of production. However, this is the strongest policy in promoting environmental outputs. Runoffs are clearly lower and agrobiodiversity and landscape diversity clearly higher than under other policies. Yet, from society's viewpoint this solution may be sub-optimal in the sense that the level of production is too low and the level of environmental outputs is too high compared to the social optimum.

The environmental outcomes of policy 6 are the same as those of the acreage subsidy coupled with environmental cross-compliance (policy 3). In fact, the only difference between these two solutions has to do with the farmer's short-run profits, which are higher under policy 6 due to agri-environmental support payments.

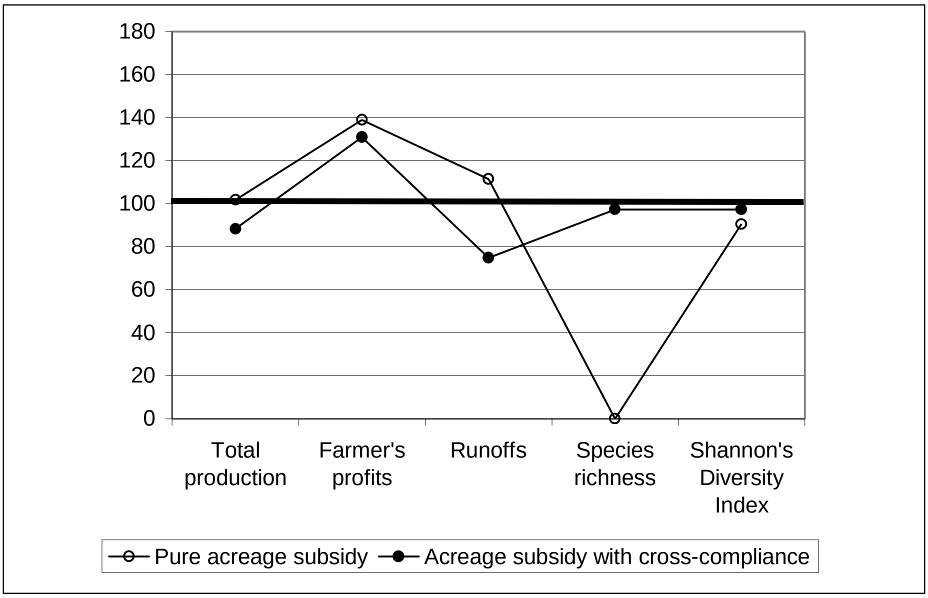

Figure 10. Indicators of multifunctional outputs in two policy solutions including acreage subsidy against the benchmark of the social optimum, which is indexed to 100.

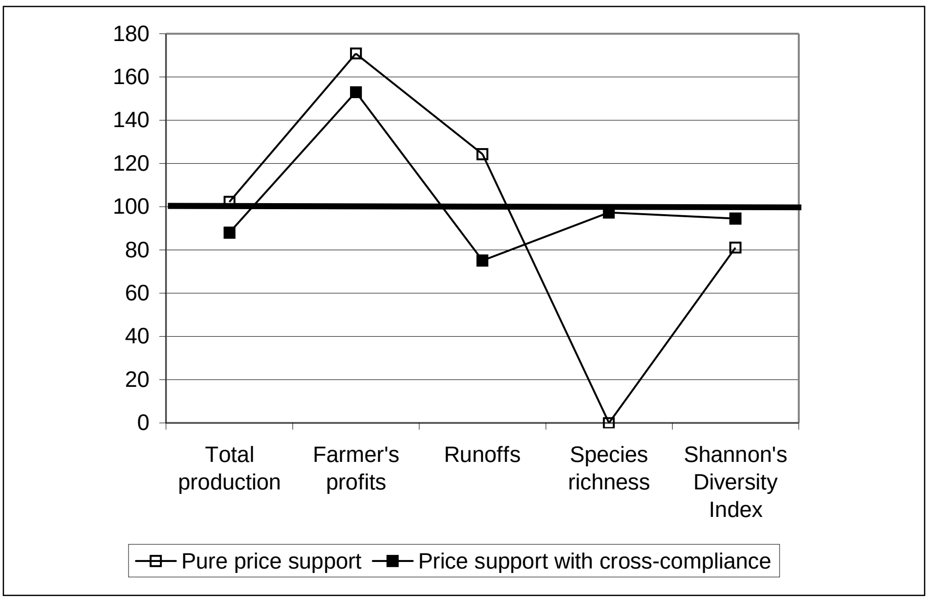

Figure 11. Indicators of multifunctional outputs in two policy solutions including price support against the benchmark of the social optimum, which is indexed to 100.

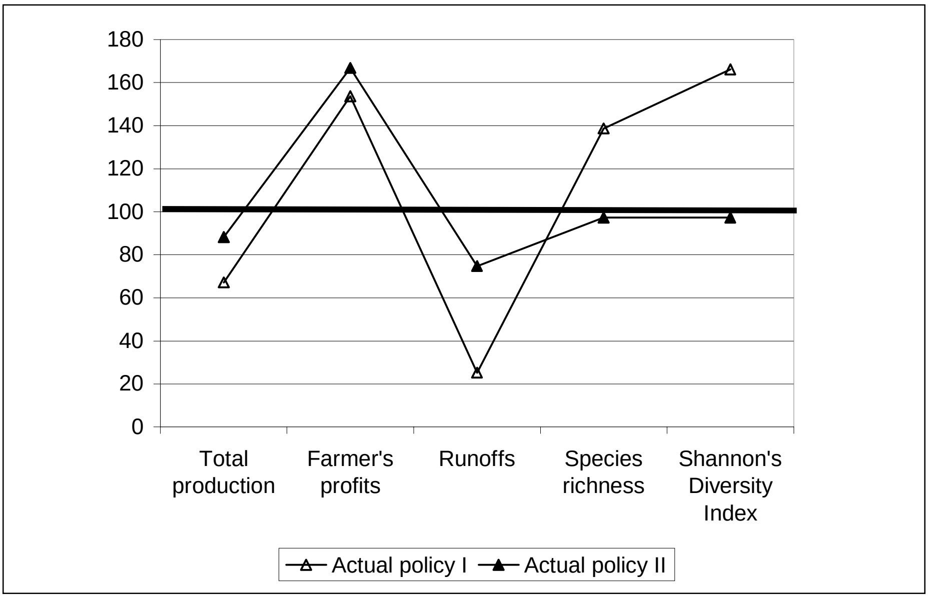

Figure 12. Indicators of multifunctional outputs in two actual policy solutions including income support against the benchmark of the social optimum, which is indexed to 100.

Figures 10 to 12 summarise the six policy solutions including income support in terms of the analysed commodity and non-commodity outputs. The private optimum is presented against the benchmark of the social optimum, which is indexed to 100.

Table 12 reports government budget outlays and social welfare under alternative policy solutions. It should be noted that in all solutions budget outlays are assumed to be financed wholly by Finnish government (including price support). Furthermore, budget outlays are subtracted from social welfare estimate, since they are direct transfers from consumers and taxpayers to farmers. Thus, social welfare estimate proxies domestic distortions related to commodity and non-commodity outputs of alternative policies against the benchmark of the social optimum.

It can be seen from Table 12 that actual policy I (with uniform buffer zone subsidy) results in the lowest level of social welfare although it is the strongest policy in promoting environmental outputs. This clearly demonstrates the need for spatial differentiation of the level of policy incentive.

Table 12. Government budget outlays and social welfare under alternative solutions.

	Budget outlays, FIM/ha	Social welfare, FIM/ha	Difference to social optimum/ha
Private optimum	-	2724	-222
Social optimum	-	2946	-
Pure acreage subsidy	1100	2724	-222
Pure price support	2053	2677	-269
Acreage subsidy with cross-compliance	1093	2794	-152
Price support with cross-compliance	1747	2784	-161
Actual policy I	2508	2216	-730
Actual policy II	2153	2781	-164

5.4 Conclusions

This chapter studied how well farm income support measures, namely producer price support and acreage support, as well as some environmental cross-compliance measures promote multifunctional agriculture.

The results show that a pure acreage subsidy as well as pure producer price support perform poorly in promoting the environmental elements of multifunctional agriculture. However, the performance of these income support measures could be greatly improved by incorporating some environmental cross-compliance mechanisms into them. In other words, eligibility for these types of support should be tied to some environmental criteria.

From the perspective of trade policy reform, however, there is one caveat in the analysis presented in this chapter. This has to do with the fact that the model only includes one extensive margin effect, that is, land allocation between the two crops. From the viewpoint of trade policy implications it would be necessary to also examine the entry and exit of cultivated land under alternative policies. This second extensive margin effect may be significant, for example, in a case where a country has plenty of marginal land under the current subsidy system, and the introduction of new support payments aimed for promoting non-commodity outputs would make also this land profitable to cultivate. In such a case the policies may increase aggregate output, which in turn may affect trade flows. This may make some of the environmental cross-compliance mechanisms and agri-environmental policies analysed here incompatible with the Green Box.

6 Conclusions

6.1 Summary and main findings

Multifunctional agriculture refers to the fact that agriculture produces jointly a number of commodity and non-commodity outputs, and some of these non-commodity outputs exhibit the characteristics of externalities and public goods. Thus, multifunctionality provides an integrated framework for the simultaneous consideration of multiple commodity and non-commodity outputs.

Multifunctionality constitutes a complex problem from the perspective of policy design and implementation. Finding out the socially optimal bundle of multiple commodity and non-commodity outputs involves the identification of the important outputs as well as their relative significance, which as such is a challenging task. Moreover, policies to promote multifunctional agriculture must simultaneously address several outputs, commodity and non-commodity ones, which have tradeoffs and complementarities in their supply. All this is further complicated by the fact that the heterogeneous conditions under which agriculture operates create a spatial dimension both in the supply of and demand for non-commodity outputs. There are spatial differences in productivity and, hence, in the production costs of commodity and non-commodity outputs on the supply side, and spatial valuation differences on the demand side. Finally, there is the practical problem that the information requirements and related transaction costs for designing and implementing spatially differentiated interventions in order to maximise social welfare from optimal bundles of commodity and non-commodity outputs may be considerable, wherefore governments may be obligated to look for less effective solutions, which are less information-intensive but may distort production decisions and thus trade.

The main objective of the present study was to contribute to the understanding of the implications of multifunctionality for effective agri-environmental policy design. The main research question addressed was the performance of various types of policy interventions in achieving the optimal bundle of multifunctional outputs under heterogeneous conditions.

The scope of the present study was restricted to the environmental dimension of multifunctionality. Two commodity outputs (crop production and related farm income) and three environmental non-commodity outputs (nutrient runoffs, landscape diversity, and agrobiodiversity) were analysed, taking into account jointness and heterogeneity in their supply and the externality and public good aspects in their demand. The analytical sections of the study are generally applicable, but the empirical parameters of the parametric model are specific to Finland.

A review of the literature relating to the environmental dimension of multifunctionality (Chapter 2) revealed the fact that formal economic analysis of the supply of, demand for, or policy design for multifunctionality is relatively recent and scarce. A number of studies are available but they tend to concentrate on individual aspects of multifunctionality. A more comprehensive approach to examining multifunctionality as a whole is only emerging.

The core chapters of the present study examined the optimal provision of multifunctional outputs without government intervention (Chapter 3), the use of differentiated (first-best) and uniform (second-best) policy instruments for promoting multifunctional agriculture (Chapter 4), as well as some "real-life" policy instruments (Chapter 5).

In Chapter 3 an analytical framework was developed for analysing multifunctional agriculture as the joint production of a number of commodity and non-commodity outputs. The privately optimal land allocation and choice of inputs were solved and compared to the corresponding social optimum. The private optimum results in a higher fertilizer use and smaller size of buffer strips than the socially optimal solution. When compared to the private optimum, the socially optimal land allocation, which is determined also by the marginal valuation of the diversity benefits and runoff damages, shifts more land to crop 1, which uses fertilizers less intensively, has larger buffer strips and thus produces a higher level of diversity. In the numerical application of the analytical model the private and social optima in the absence of taxes and subsidies were solved, showing that the private solution leads to excessive use of fertilizers, sub-optimal use of buffer strips, and an excessive amount of land devoted to the production of crop 2.

Chapter 4 investigated the optimal use of a fertilizer tax and buffer strip subsidy to guide the farmer's private solution towards the socially optimal way of producing commodity and non-commodity outputs. The optimal fertilizer tax depends on the marginal runoff from fertilizer use and its marginal damage, as well as on the marginal effect of fertilizer use on agrobiodiversity. The optimal buffer strip subsidy depends on the marginal runoff reduction achieved and its marginal value, as well as on the marginal effect of buffer strips on agrobiodiversity. All these factors are constant over parcels except the marginal runoffs from fertilizer use and marginal runoff reduction from buffer strips. These vary over land quality because of variations in fertilizer intensity and buffer strip width, and make the optimal fertilizer tax and buffer strip subsidy vary over parcels and crops as well. Thus, the promotion of multifunctional agriculture under heterogeneous land quality requires the combined use of differentiated instruments to achieve the first-best solution. Taking into account net support to agriculture modifies the first-best instruments so that the basic level of fertilizer tax increases and the basic level of buffer strip subsidy decreases. In the numerical application the differentiated first-best tax rates and subsidy levels were solved, yielding 60 pairs of crop- and parcel-specific fertilizer taxes and buffer strip subsidies. For comparison, uniform policy instruments were also applied. The social welfare difference between the first-best differentiated instruments and the second-best uniform instruments is FIM 64 per hectare in the case of semi-uniform instruments and FIM 116 per hectare in the case of fully uniform instruments.

Chapter 5 studied how well farm income support measures, namely producer price support and acreage support, as well as some environmental cross-compliance measures promote multifunctional agriculture. The results show that pure acreage subsidy and pure producer price support perform poorly in promoting the environmental elements of multifunctional agriculture. However, the performance of these income support measures could be greatly improved by incorporating some environmental cross-compliance mechanisms into them.

6.2 Policy implications

The study brings out how the design of agri-environmental policies against the background of multifunctionality differs from the individual treatment of the various environmental effects of agriculture. Because of the joint production process, the levels of different multifunctional outputs are linked to each other. Hence, the regulation of one environmental effect necessarily influences the other environmental effects and agricultural production, as well as other dimensions of multifunctionality. These interactions need to be accounted for when designing policies inducive to multifunctionality.

Theoretically, first-best policy intervention requires the use of differentiated policy instruments to account for the heterogeneity in the production conditions, which causes spatial variation in the supply of both commodity and non-commodity outputs. However, the information requirements and related information costs of such differentiated instruments may be enormous. Hence, although the performance of policy instruments can be improved through spatial targeting, the increased administrative costs have to be weighed against the potential gains in precision. This is one of the reasons why policy instruments that are used in practice differ from the theoretically optimal ones. Resorting to second-best, that is, not fully differentiated, policy instruments reduces the administrative burden but fails to secure the production of the optimal bundle of multifunctional outputs.

In the international debate on multifunctionality, one of the key issues has to do with the justification for continued domestic support. Such support, it is argued, is needed especially in countries where the commodity production itself is uncompetitive to guarantee the supply of the non-commodity benefits produced by agriculture. The central question in this context is whether the support can be linked to commodity production. Perhaps the two most common farm income support measures, price support and acreage subsidy, perform poorly with respect to the environmental dimension of multifunctionality. Environmental cross-compliance schemes, on the other hand, can provide sufficient means for promoting multifunctionality even with price support and acreage subsidy. Eligibility for these types of support should thus be tied to some environmental criteria.

As far as the case of Finland is concerned, it can be stated that there is no need to establish a new support system dedicated for promoting environmental multifunctionality. The existing Finnish agri-environmental programme already provides a good starting point for this purpose. The programme enjoys wide participation, covering over 90% of farms and 95% of the cultivated area, comprises all relevant environmental non-commodity outputs, and has an administrative structure in place. However, the programme would benefit from an explicit consideration of the themes of the present study: jointness, heterogeneity, and the valuation of externalities and public goods.

6.3 Limitations of the study and suggestions for further research

The present study is concerned with how heterogeneous conditions should be taken into account in policy design for environmental multifunctionality. However, there is also another pervasive feature which affects the agriculture – environment relationship: uncertainty due to stochastic factors (Lichtenberg 2000). The present study has not touched upon uncertainty. Stochastic factors, such as weather conditions, do have an important role in the determination of agricultural commodity production and the associated nutrient runoffs, for example. On the other hand, they may play a minor role, if any at all, in the supply of some other non-commodity outputs like landscape diversity. Heterogeneity, in turn, may be important in the context of all environmental non-commodity outputs of agriculture. Therefore, at this stage of research it seemed that accounting for heterogeneity takes priority over uncertainty. But, as the research on multifunctionality develops, it may be interesting to explore uncertainty as well.

The same conclusion applies to dynamic aspects. They may be significant in environmental multifunctionality, but were ignored in the present study. Future research may examine the possible dynamics of the interdependencies between commodity and non-commodity outputs.

Moreover, the model developed here has a fixed amount of land and thus only includes one extensive margin effect: the land allocation between crops. The other types of extensive margin effects, that is, changes in the amount of cultivated land (entry and exit) under alternative policies and the resulting effects on aggregate output levels remain uncovered. Yet, these effects are important in discussions relating to trade policy reform.

In the empirical sections of the present study multifunctionality was valued on the basis of its individual components. However, it would be interesting to elaborate these results if an estimate on the simultaneous valuation of all environmental non-commodity outputs were available.

The issue of transaction costs is highlighted in the case of multifunctionality and heterogeneity because of the very complicated nature of the optimal solutions. The present study has referred to this problematique several times, but did not directly incorporate transaction costs into the analysis. Future research should try to explicitly account for transaction costs and assess their importance in the face of the multifaceted policy design and implementation problems that are posed by multifunctionality.

Finally, for reasons explained in Chapter 1.2, the present study covers only the environmental dimension of multifunctionality. For example, the socio-economic viability of rural areas was touched only very indirectly through farmers' income. However, it is in the very nature of multifunctionality that all of its aspects should be considered in a simultaneous and integrated manner. Future research may be able to move towards this goal as a common understanding of what exactly constitutes multifunctionality develops.

References

Aakkula, J.J. 1999. Economic value of pro-environmental farming – A critical and decision-making oriented application of the contingent valuation method. Agricultural Economics Research Institute, Publications 92. Helsinki, Finland.

Altieri, M.A. and Nicholls, C.I. 1999. Biodiversity, ecosystem function, and insect pest management in agricultural ecosystems. In: Collins, W.W. and Qualset, C.O. (eds.) Biodiversity in agroecosystems. CRC Press, Florida, USA. P. 69-84.

Anderson, K. 1992. The standard welfare economics of policies affecting trade and the environment. In: Anderson, K. and Blackhurst, R. (eds.) The greening of world trade issues. London: Harvester Wheatsheaf, pp. 25-47.

Antle, J.M. and Just, R.E. 1992. Conceptual and empirical foundations for agricultural-environmental policy analysis. Journal of Environmental Quality 21: 307-316.

Arshad, M.A. and Martin, S. 2002. Identifying critical limits for soil quality indicators in agro-ecosystems. Agriculture, Ecosystems and Environment 88: 153-160.

Baumol, W. and Oates, W. 1988. The theory of environmental policy. Cambridge: Cambridge University Press.

Beattie, B.R. and Taylor, C.R. 1985. The economics of production. New York: John Wiley & Sons.

Bennet, J. and Blamey, R. (eds.). 2001. The choice modelling approach to environmental valuation. Edward Elgar, Cheltenham, UK.

Boisvert, R. 2001. A note on the concept of jointness in production. Technical annexes (Annex 1 pp. 105-123 Annex 2 pp. 125-132) in Multifunctionality: Towards an analytical framework. 159 p. OECD, Paris.

Braden, J.B. and Segerson, K. 1993. Information problems in the design of nonpoint source pollution policy. In: Russell, C.S. & Shogren, J.F. (eds.). Theory, modeling and experience in the management of nonpoint source pollution. Kluwer Academic Publishers, Boston, USA. p. 1-36.

Brunstad, R.J., Gaasland, I. and Vårdal, E. 1999. Agricultural production and the optimal level of landscape preservation. Land Economics: 75(4): 538-546.

Burrell, A. 2001. Multifunctionality and agricultural trade liberalisation. Paper presented at: 77th EAAE Seminar/NJF Seminar No. 325, August 17-18, 2001, Helsinki.

Bäckman, J-P.C., Helenius, J., Maohua, M. and Tarmi, S. 1999. Mitkä tekijät vaikuttavat lajien esiintymiseen peltoympäristön pientareilla ja suojakaistoilla? Agro-Food 1999, Poster abstracts, pp. 26. In Finnish.

Bäckman, S.T., Vermeulen, S. and Taavitsainen, V.-M. 1997. Long-term fertilizer field trials: comparison of three mathematical response models. Agricultural and Food Science in Finland 6: 151-160.

Callan, S.J. and Thomas, J.M. 1996. Environmental Economics & Management: Theory, Policy and Applications. Irwin. 726 p.

Chiang, A.C. 1984. Fundamental methods of mathematical economics. Third Edition. McGraw-Hill.

Christensen, T. and Rygnestad, H. 2000. Environmental cross-compliance: topics for future research. Danish Institute of Agricultural and Fisheries Economics.

Cornes, R. and Sandler, T. 1996. The Theory of Externalities, Public Goods and Club Goods. Cambridge University Press.

Correll, D.L. 1997. Buffer zones and water quality protection: general principles. In: Haycock, N.E. et al. (eds.). Buffer zones: Their processes and potential in water protection. Quest Environmental. Harpenden, UK. p. 7-20.

De Koijer, T.J., Wossink, G.A.A., Van Ittersum, M.K., Struik, P.C. and Renkema, J.A. 1999. A conceptual model for analysing input-output coefficients in arable farming systems: from diagnosis towards design. Agricultural Systems 61:33-44.

Dillman, B. and Bergstrom, J. 1991. Measuring Environmental Amenity Benefits of Agricultural Land. pp. 250-271. In: Hanley, N. (ed.). Farming and the countryside: An economic analysis of external costs and benefits.

Drake, L. 1992. The non-market value of the Swedish agricultural landscape. European Review of Agricultural Economics, 19: 351-364.

Duelli, P. 1997. Biodiversity evaluation in agricultural landscapes: An approach at two different scales. Agriculture, Ecosystems and Environment 62: 81-91.

Eiden, G., Kayadjanian, M. and Vidal, C. 2000. Capturing landscape structures: Tools. At: http://europa.eu.int/comm/agriculture/publi/landscape. Accessed on March 1st. 2001.

Falconer, K., Dupraz, P. and Whitby, M. 2001. An investigation of policy administrative costs using panel data for the English Environmentally Sensitive Areas. Journal of Agricultural Economics 52: 83-103.

Fleming, R.A. and Adams, R.M. 1997. The importance of site-specific information in the design of policies to control pollution. Journal of Environmental Economics and Management 33:347-358.

Forman, R.T.T. 1995. Land Mosaics: the ecology of landscapes and regions. Cambridge University Press.

Gilliam, J.W., Parsons, J.E. and Mikkelsen, R.L. 1997. Nitrogen dynamics and buffer zones. In: Haycock, N.E. et al. (eds.). Buffer zones: Their processes and potential in water protection. Quest Environmental. Harpenden, UK. p. 54-61.

Gliessman, S.R. 2000. Agroecology: ecological processes in sustainable agriculture. Lewis Publishers, Boca Raton.

Griffin, R.C. and Bromley, D.W. 1982. Agricultural runoff as a nonpoint externality: a theoretical development. American Journal of Agricultural Economics 70: 37-49.

Grönroos, J., Nikander, A., Syri, S., Rekolainen, S. and Ekqvist, M. 1998. Maatalouden ammoniakkipäästöt. Suomen ympäristö 206.

Guyomard, H. and Levert, F. 2001. Multifunctionality, trade distortion effects and agricultural income support: a conceptual framework with free entry and land price endogeneity. Paper presented at the seminar on "Multifunctional agriculture", February 2001, Bergen, Norway.

Hanley, N. 1990. The economics of nitrate pollution. European Review of Agricultural Economics 17: 129-151.

Hardie, I.W. and Parks, P.J. 1997. Land use with heterogeneous land quality: an application of an area base model. American Journal of Agricultural Economics 77: 299-310.

Helfand, G.E. and House, B.W. 1995. Regulating nonpoint source pollution under heterogeneous conditions. American Journal of Agricultural Economics 77: 1024-1032.

Heritage Landscapes Working Group. 2000. A national summary report by the heritage landscapes working group.

Hietala-Koivu, R., Tahvanainen, L., Nousiainen, I., Heikkilä, T., Alanen, A., Ihalainen, M., Tyrväinen, L. and Helenius, J. 1999. Visuaalinen maisema maatalouden ympäristöohjelman vaikuttavuuden seurannassa. Maatalouden tutkimuskeskuksen julkaisuja, sarja A, no. 50.

Hill, A.R. 1996. Nitrate removal in stream riparian zones. Journal of Environmental Quality 25: 743-755.

Hochman, E. and Zilberman, D. 1978. Examination of environmental policies using production and pollution microparameter distributions. Econometrica 46(4):739-59.

Holstein, F. 1998. The values of the agricultural landscape: a discussion on value-related terms in natural and social sciences and the implications for the contingent valuation method. In: Dabbert, S., Dubgaard, A., Slangen, L. and Whitby, M. (eds.). The economics of landscape and wildlife conservation. CAB International. Pp. 37-52.

Hueth, B. 2000. The goals of U.S. agricultural policy: a mechanism design approach. American Journal of Agricultural Economics 82: 14-24.

Kleijn, D. 1997. Species richness and weed abundance in the vegetation of arable field boundaries. Ph.D. thesis. Wageningen Agricultural University, Wageningen, 177 pp.

Kleijn, D. and Snoeijing, I.J. 1997. Field boundary vegetation and the effects of agrochemical drift: botanical change caused by low levels of herbicide and fertilizer. Journal of Applied Ecology 34: 1413-1425.

Laitinen, P., Raisio, R., and Siimes, K. 1996. Torjunta-ainepäästöt maataloudessa (MATYVA-projekti). Maatalouden tutkimuskeskuksen julkaisuja, sarja A, no. 12.

Latacz-Lohmann, U. 2000. Beyond the Green Box: The economics of agri-environmental policy and free-trade. Agrarwirtschaft 49(2000), Heft 9/10.

Latacz-Lohmann, U. 2001. A policy decision-making framework for devising optimal implementation strategies for good agricultural and environmental policy practices. OECD COM/AGR/CA/ENV/EPOC(2000)56/FINAL.

Lau, L.J. 1972. Profit functions of technologies with multiple inputs and outputs. Review of Economics and Statistics 54: 281-289.

Leathers, H.D. 1991. Allocable fixed inputs as a cause of joint production: a cost function approach. American Journal of Agricultural Economics 73:1083-1090.

Lichtenberg, E. 1989. Land quality, irrigation development, and cropping patterns in the Northern High Plains. American Journal of Agricultural Economics 71:187-194.

Lichtenberg, E. 2000. Agriculture and the Environment. Department of Agricultural and Resource Economics, University of Maryland, Working Paper No. 00-15.

Louviere, J.J. 2001. Choice experiments: an overview of concepts and issues. In: Bennet, J. and Blamey, R. (eds.). The choice modelling approach to environmental valuation. Edward Elgar, Cheltenham, UK.

Lynne, G.D. 1988. Allocatable fixed inputs and jointness in agricultural production: implications for economic modeling: Comment. American Journal of Agricultural Economics 70: 947-949.

Ma, M., Tarmi, S. and Helenius, J. 2002. Revisiting the species-area relationship in a semi-natural habitat: floral richness in agricultural buffer zones in Finland. Agriculture, Ecosystems and Environment 89: 137-148.

MAF 2001a. Agriculture and countryside. Ministry of Agriculture and Forestry, Finland 2001. At: http://www.mmm.fi/english/agriculture.

MAF 2001b. Maatalouden kehitysarvio kansallista ilmasto-ohjelmaa varten. Työryhmämuistio MMM 2001:2. Maa- ja metsätalousministeriö, Helsinki. 44 s.

McCann, L. and Easter, K.W. 1999. Transaction costs of policies to reduce agricultural phosphorus pollution in the Minnesota River. Land Economics 75: 402-414.

McGarigal, K. and Marks, B. 1994. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. Corvallis: Forest Science Department, Oregon State University.

Miettinen, A., Koikkalainen, K., Vehkasalo, V. and Sumelius, J. 1997. Luomu-Suomi? Maatalouden tuotantovaihtoehtojen ympäristötaloudelliset vaikutukset –projektin loppuraportti. Maatalouden taloudellisen tutkimuslaitoksen Julkaisuja 83.

Miller, D.J. and Plantinga, A.J. 1999. Modeling land use decisions with aggregate data. American Journal of Agricultural Economics 81: 180-194.

Ministry of Environment. 2002. Everyman’s right in Finland: introduction. At: http://www.vyh.fi/eng/environ/naturcon/everyman/introd.

Moschini, G. 1989. Normal inputs and joint production with allocatable fixed factors. American Journal of Agricultural Economics 71: 1021-1024.

MTTL 2000. Finnish agriculture and rural industries 1999/2000. Agricultural Economics Research Institute, Finland, Publications 95a.

MTTL 2001. Finnish agriculture and rural industries 2001. Agricultural Economics Research Institute, Finland, Publications 97a.

Navrud, S. 2000. Valuation techniques and benefit transfer methods: strengths, weaknesses, and policy utility. In OECD Proceedings “Valuing rural amenities”. OECD, Paris.

NRC 1993. Soil and Water Quality - An Agenda for Agriculture (USA: National Research Council).

OECD 1999. Handbook of incentive measures for biodiversity: design and implementation. OECD, Paris.

OECD 2001a. Multifunctionality: Towards an analytical framework. 159 p. OECD, Paris.

OECD 2001b. Environmental indicators for agriculture: methods and results, Volume 3. OECD, Paris.

Orazem, P.F. and Miranowski, J.A. 1994. A dynamic model of acreage allocation with general and crop-specific soil capital. American Journal of Agricultural Economics 76: 385-395.

Pearce, D.W. and Turner, R.K. 1990. The economics of natural resources and environment. Harvester Wheatsheaf, Exeter. 359 p.

Persson, T. and Tabellini, G. 1990. Macroeconomic policy, credibility and politics. Harwood Academic Publishers, New York. 187 p.

Peterson, J., Boisvert, R. and de Gorter, H. 1999. Multifunctionality and optimal environmental policies for agriculture in an open economy. Working Paper 99-29. Department of Applied Economics and Management, Cornell University, Ithaca, New York.

Pitkänen, M. and Tiainen, J. 2001. Biodiversity of agricultural landscapes in Finland. BirdLife Finland Conservation Series (No. 3). Helsinki, Finland.

Plantinga, A.J. 1996. The effect of agricultural policies on land use and environmental quality. American Journal of Agricultural Economics 78: 1082-1091.

Pykälä, X. and Alanen, A. 1996. Kukkivat niityt ja lehtevät hakamaat pian pelkkä muisto vain. Ympäristö 2: 20-22.

Qualset, C.O., McGuire, P.E. and Warburton, M.L. 1995. Agrobiodiversity: key to agricultural productivity. California Agriculture 49(6): 45-49.

Randall, A. 1991. Total and nonuse values. In: Braden, J.B. and Kolstad, C.D. (eds.). Measuring the demand for environmental quality. Elsevier Science Publishers B.V. (North-Holland).

Randall, A. 2002. Valuing the outputs of multifunctional agriculture. Department of Agricultural, Environmental, and Development Economics, The Ohio State University, Working Paper, AEDE-WP-0023-02.

Rassi, P., Kaipainen, H., Mannerkoski, I. and Ståhls, G. 1991. Report on the monitoring of threatened animals and plants in Finland. Ministry of the Environment, Committee report.

Ribaudo, M., Horan, R. and Smith, M. 1999. Economics of Water Quality Protection From Nonpoint Sources: Theory and Practice. Resource Economics Division, Economic Research Service, U.S. Department of Agriculture. Agricultural Economic Report No. 782.

Romstad, E., Vatn, A., Rorstad, P.K. and Soyland, V. 2000. Multifunctional agriculture: implications for policy design. Agricultural University of Norway, Department of Economics and Social Sciences. Report No. 21. 139 p.

Ruuska, R. and Helenius, J. 1996. GIS analysis of change in an agricultural landscape in Central Finland. Agricultural and Food Science in Finland, Vol. 5: 567-576.

Santos, J.M.L. 2000. Problems and potential in valuing multiple outputs: externality and public good non-commodity outputs from agriculture. In OECD Proceedings “Valuing rural amenities”. OECD, Paris.

Schippers, P. and Joenje, W. 2002. Modelling the effect of fertiliser, mowing, disturbance and width on the biodiversity of plant communities of field boundaries. Forthcoming in Agriculture, Ecosystems and Environment.

Segerson, K. 1988. Uncertainty and incentives for nonpoint pollution control. Journal of Environmental Economics and Management 15: 87-98.

Shah, F.A., Zilberman, D. and Chakravorty, U. 1995. Technology adoption in the presence of an exhaustible resource: the case of groundwater extraction. American Journal of Agricultural Economics 77: 291-299.

Shortle, J.S. 1990. The allocative efficiency implications of water pollution abatement cost comparisons. Water Resources Research 26(5): 793-797.

Shortle, J.S. and Dunn, J.W. 1986. The relative efficiency of agricultural source water pollution control policies. American Journal of Agricultural Economics 68: 668-677.

Shortle, J.S., Horan, R.D. and Abler, D.G. 1998. Research issues in nonpoint pollution control. Environmental and Resource Economics 11: 571-585.

Shumway, C.R., Pope, R.D. and Nash, E.K. 1984. Allocatable fixed inputs and jointness in agricultural production: implications for economic modeling. American Journal of Agricultural Economics 66: 72-78.

Shumway, C.R., Pope, R.D. and Nash, E.K. 1988. Allocatable fixed inputs and jointness in agricultural production: implications for economic modeling: Reply. American Journal of Agricultural Economics 70: 950-952.

Siikamäki, J. 1997. Torjunta-aineiden käytön vähentämisen arvo? Contingent valuation –tutkimus kuluttajien maksuhalukkuudesta. Maatalouden taloudellisen tutkimuslaitoksen Tutkimuksia 217.

Simmelsgaard, S. 1991. Estimation of nitrogen leakage functions - Nitrogen leakage as a function of nitrogen applications for different crops on sand and clay soils. In: Rude, S. (ed.). Nitrogen fertilizers in Danish Agriculture - present and future application and leaching, Institute of Agricultural Economics Report 62 (in Danish: Kvaelstofgödning i landbruget - behov og udvasking nu og i fremtiden). English summary. Copenhagen. p. 135-150.

Stiglitz, J.E. 1988. Economics of the public sector. 2:nd edition, Norton. 692 p.

Swift, M.J. and Anderson, J.M. 1994. Biodiversity and ecosystem function in agroecosystems. In: Schultze, E. and Mooney, H.A. (eds.) Biodiversity and Ecosystem function. Springer, New York, pp. 15-41.

Tahvanainen, L., Tyrväinen, L. and Nousiainen, I. 1996. Effect of afforestation on the scenic value of rural landscape. Scandinavian Journal of Forest Research 11(4): 397-405.

Taylor, A.E. and Mann, W.R. 1983. Advanced Calculus. Third Edition. John Wiley & Sons, Inc. USA.

Tilman, D. 1993. Species richness of experimental productivity gradients: how important is colonization limitation? Ecology 74: 2179-2191.

Turtola, E. and Jaakkola, A. 1987. Viljelykasvin vaikutus ravinteiden huuhtoutumiseen savimaasta Jokioisten huuhtoutumiskentällä v. 1983-1986. (in Finnish). Maatalouden tutkimuskeskus. Tiedote 22/87. Jokioinen. 33 p.

Turtola, E. and Puustinen, M. 1998. Kasvipeitteisyys ravinnehuuhtoutumien vähentäjänä. Vesitalous 1/1998: 6-11.

Ulph, A. 1998. International trade and the environment: a survey of recent economic analysis. In: Yearbook of environmental and resource economics 1997/1998. Edward Elgar. pp. 205- 242.

UNEP 1992. Convention on biological diversity. UNEP, Geneva.

Uusi-Kämppä, J. and Yläranta, T. 1992. Reduction of sediment, phosphorus and nitrogen transport on vegetated buffer strips. Agricultural Science in Finland 1: 569-575.

Uusi-Kämppä, J. and Yläranta, T. 1996. Effect of buffer strip on controlling erosion and nutrient losses in Southern Finland. In: Mulamoottil, G., Warner, B.G. & McBean, E.A. (eds.). Wetlands: environmental gradients, boundaries and buffers. Boca Raton: CRC Press/Lewis Publishers. p. 221-235.

Uusi-Kämppä, J., Braskerud, B., Jansson, H., Syversen, N., and Uusitalo, R. 2000. Buffer zones and constructed wetlands as filters for agricultural phosphorus. Journal of Environmental Quality 29: 151-158.

Valpasvuo-Jaatinen, P., Rekolainen, S. and Latostenmaa, H. 1997. Finnish agriculture and its sustainability: environmental impacts. Ambio 26: 448-455.

Van Wenum, J.H., Oude Lansink, A.G.J.M., and Wossink, G.A.A. 1999. Farm heterogeneity in wildlife production. Paper presented at AAEA Annual Meeting, August 8-11, Nashville, Tennessee.

Van Wenum, J.H., Wossink, G.A.A. and Renkema, J.A. 2001. Location-specific modelling for optimising wildlife management on crops farms. Forthcoming in Ecological Economics.

Vatn, A. 2001. Transaction costs and multifunctionality. Paper presented at the OECD “Workshop on Multifunctionality”. Paris, 2-3 July, 2001.

Vatn, A. 2002. Multifunctional agriculture: some consequences for international trade regimes. European Review of Agricultural Economics 29(3): 309-327.

Vehkasalo, V. 1999. Ympäristötuen yhteiskunnallinen kannattavuus (Abstract: Social profitability of the Finnish agri-environmental programme) pp. 42-77. In: Maatalouden ympäristöohjelma 1995-1999:n taloudellinen analyysi. Agricultural Economics Research Institute, Publications 90. Helsinki, Finland.

Weitzman, M. 1974. Prices vs. quantities. Quarterly Journal of Economics 41(4): 477-91.

Whitby, M.C. and Saunders, C.M. 1996. Estimating the supply of conservation goods in Britain: a comparison of financial efficiency of two policy instruments. Land Economics 72: 313-325.

Wiens, J.A.1995. Landscape mosaics and ecological theory. In: Hansson, L., Fahrig, L. and Merriam, G. (eds.) Mosaic landscapes and ecological processes. Chapman and Hall, London, UK. Pp. 1-26.

Wossink, A., van Wenum, J., Jurgens, C. and de Snoo, G. 1999. Co-ordinating economic, behavioural and spatial aspects of wildlife preservation in agriculture. European Review of Agricultural Economics 26(4): 443-460.

Wossink, G.A.A., Oude Lansink, A.G.J.M. and Struik, P.C. 2001. Non-separability and heterogeneity in integrated agronomic-economic analysis of nonpoint-source pollution. Ecological Economics 38:345-357.

Yrjölä, T. and Kola, J. 2001. Cost-benefit analysis of multifunctional agriculture in Finland. Agricultural and Food Science in Finland 10: 295-307.

Appendix 1 (1/1). Comparative statics for Chapters 3 and 4

Input use

The per parcel profit function of the representative farmer is

A1.1 $$\max_{\{l_i, m_i\}} \pi^i = p_i(1-m_i) f^i(l_i;q) - c^*(1-m_i)l_i + (\lambda - \tfrac{1}{2}\lambda m_i)m_i$$

for i = 1,2

The first-order conditions are

A1.2a $$\pi^i_{l_i} = p_i f^i_{l_i} - c^* = 0$$

A1.2b $$\pi^i_{m_i} = -p_i f^i(l_i;q) + c^* l_i + \lambda - \lambda m_i = 0$$

The second-order conditions are

A1.3a $$\pi^i_{l_i l_i} = p_i f^i_{l_i l_i} < 0$$

A1.3b $$\pi^i_{m_i m_i} = -\lambda < 0$$

A1.3c $$\pi^i_{l_i m_i} = 0 = \pi^i_{m_i l_i}$$

A1.3d $$\Delta = \pi_{ll}\pi_{mm} - \pi_{lm}{}^2 = \pi_{ll}\pi_{mm} > 0$$

The comparative statics effects of market parameters, that is output and input prices, can be solved by applying the Cramer's Rule from

A1.4a $$\begin{bmatrix} \pi_{ll} & 0 \\ 0 & \pi_{mm} \end{bmatrix} \begin{bmatrix} dl \\ dm \end{bmatrix} = -\begin{bmatrix} f^i_{l_i} & -(1+t) \\ -f^i(l_i;q) & (1+t)l_i \end{bmatrix} \begin{bmatrix} dp \\ dc \end{bmatrix}$$

and the effects of a buffer strip subsidy and a fertilizer tax from

A1.4b $$\begin{bmatrix} \pi_{ll} & 0 \\ 0 & \pi_{mm} \end{bmatrix} \begin{bmatrix} dl \\ dm \end{bmatrix} = -\begin{bmatrix} 0 & -c \\ 1-m_i & cl_i \end{bmatrix} \begin{bmatrix} d\lambda \\ dt \end{bmatrix}$$

Thus, for instance, for the fertilizer tax $\frac{dl}{dt} = \Delta^{-1}(c\pi_{mm}) < 0$ and $\frac{dm}{dt} = -\Delta^{-1}(cl_i\pi_{ll}) > 0$. Other cases go analogously.

Appendix 2 (1/1). Input use, profits, and social returns in selected parcels in the absence of intervention

This Appendix presents some details of the parametric model for selected parcels. Land quality *q* increases from 1 to 60. Input choices are given first, followed by profits and social returns. Note that buffer strips are absent in the private optimum.

	Private optimum		***Social optimum***			
	Fertilizer intensity		Fertilizer intensity		Buffer strips	
q	Crop 1	Crop 2	Crop 1	Crop 2	Crop 1	Crop 2
1	129,3476	152,3207	123,8169	142,0883	0,009065	0,010991
5	129,4026	152,4219	123,8665	142,1695	0,008932	0,010573
10	129,4662	152,539	123,924	142,262	0,008776	0,010091
15	129,524	152,6453	123,9765	142,3445	0,008634	0,009655
20	129,5761	152,7411	124,024	142,417	0,008504	0,009264
24	129,6135	152,81	124,0584	142,4678	0,00841	0,008984
25	129,6223	152,8262	124,0665	142,4795	0,008388	0,008919
26	129,6309	152,8419	124,0744	142,4908	0,008366	0,008855
27	129,6392	152,8573	124,0821	142,5017	0,008345	0,008793
28	129,6473	152,8722	124,0896	142,5122	0,008324	0,008733
29	129,6551	152,8866	124,0969	142,5223	0,008304	0,008675
30	129,6628	152,9007	124,104	142,532	0,008284	0,008619
35	129,6974	152,9645	124,1365	142,5745	0,008194	0,008365
40	129,7263	153,0177	124,164	142,607	0,008116	0,008156
45	129,7495	153,0602	124,1865	142,6295	0,008052	0,007993
50	129,7668	153,0922	124,204	142,642	0,008	0,007875
55	129,7784	153,1134	124,2165	142,6445	0,007962	0,007803
60	129,7841	153,1241	124,224	142,637	0,007936	0,007776

	Private profits		***Social returns***	
q	Crop 1	Crop 2	Crop 1	Crop 2
1	2704,182	2476,553	2729,628	2472,963
5	2733,954	2552,333	2759,056	2547,764
10	2768,7	2641,23	2793,408	2635,56
15	2800,704	2723,65	2825,054	2717,002
20	2829,963	2799,594	2853,992	2792,08
24	2851,396	2855,686	2875,192	2847,553
25	2856,48	2869,061	2880,221	2860,783
26	2861,454	2882,177	2885,141	2873,758
27	2866,318	2895,034	2889,953	2886,478
28	2871,072	2907,631	2894,657	2898,942
29	2875,717	2919,97	2899,252	2911,15
30	2880,252	2932,049	2903,739	2923,103
35	2901,28	2988,56	2924,546	2979,032
40	2919,565	3038,592	2942,641	3028,565
45	2935,105	3082,146	2958,024	3071,699
50	2947,901	3119,221	2970,692	3108,428
55	2957,952	3149,817	2980,647	3138,75
60	2965,259	3173,933	2987,888	3162,662

Appendix 3 (1/1). Input use, profits, and instruments in selected parcels under differentiated instruments

This Appendix presents some details of the parametric model for selected parcels. Land quality q increases from 1 to 60. Input choices are given first, followed by profits and instrument levels.

	Differentiated instruments					
	Fertilizer intensity		Buffer strips		Profits	
q	Crop 1	Crop 2	Crop 1	Crop 2	Crop 1	Crop 2
1	123,8153	142,0886	0,00902	0,011069	2527,406	2337,453
5	123,8657	142,1694	0,008883	0,01057	2556,956	2383,764
10	123,924	142,2626	0,008728	0,010035	2591,445	2453,987
15	123,977	142,3471	0,00859	0,009584	2623,214	2525,802
20	124,0246	142,4228	0,008467	0,0092	2652,263	2594,84
24	124,0589	142,4771	0,00838	0,008955	2673,543	2647,242
25	124,067	142,4898	0,008359	0,008891	2678,591	2659,713
26	124,0748	142,5021	0,008339	0,008825	2683,53	2671,933
27	124,0824	142,5141	0,00832	0,008758	2688,36	2683,902
28	124,0899	142,5258	0,008301	0,008704	2693,082	2695,818
29	124,0971	142,5371	0,008283	0,008652	2697,694	2707,499
30	124,104	142,548	0,008266	0,008598	2702,198	2718,927
35	124,1358	142,5976	0,008186	0,008364	2723,084	2772,753
40	124,1622	142,6384	0,00812	0,008165	2741,25	2820,701
45	124,1833	142,6704	0,008066	0,008001	2756,694	2862,643
50	124,1991	142,6938	0,008025	0,007865	2769,417	2898,464
55	124,2096	142,7084	0,007996	0,007756	2779,418	2928,116
60	124,2148	142,7142	0,007979	0,007672	2786,699	2951,541

	Differentiated instruments per hectare			
	Tax-decomp.		***Subsidy-decomp.***	
q	Crop 1	Crop 2	Crop 1	Crop 2
1	171,454	224,35	22,90087	120,7402
5	171,6896	225,0393	22,81519	91,56428
10	171,9617	225,8325	22,71702	73,25019
15	172,2092	226,5517	22,629	63,00232
20	172,4321	227,2009	22,55136	56,43477
24	172,5928	227,6664	22,49685	53,05129
25	172,6305	227,7779	22,48429	52,20035
26	172,6672	227,8876	22,47216	51,34724
27	172,703	227,9954	22,46047	50,49189
28	172,7378	228,0978	22,4492	49,87302
29	172,7716	228,1978	22,43837	49,27369
30	172,8045	228,2957	22,42798	48,6734
35	172,9541	228,7445	22,38259	46,24823
40	173,0795	229,1293	22,34826	44,38574
45	173,1806	229,4505	22,32509	42,96163
50	173,2575	229,709	22,31317	41,85312
55	173,3102	229,9051	22,31258	41,01593
60	173,3387	230,0394	22,32335	40,38641

Appendix 4 (1/1). Comparative statics for Chapter 5

Input use

The per parcel profit function of the representative farmer is

A4.1 $$\max_{\{l_i, m_i\}} \pi^i = p_i^*(1-m_i) f^i(l_i;q) - c(1-m_i)l_i + s(1-m_i) + (\lambda - \tfrac{1}{2}\lambda m_i)m_i$$

for i = 1,2

The first-order conditions are

A4.2a $$\pi^i_{l_i} = p_i^* f^i_{l_i} - c = 0$$

A4.2b $$\pi^i_{m_i} = -p_i^* f^i(l_i;q) + cl_i - s + \lambda - \lambda m_i = 0$$

The second-order conditions are

A4.3a $$\pi^i_{l_i l_i} = p_i^* f^i_{l_i l_i} < 0$$

A4.3b $$\pi^i_{m_i m_i} = -\lambda < 0$$

A4.3c $$\pi^i_{l_i m_i} = 0 = \pi^i_{m_i l_i}$$

A4.3d $$\Delta = \pi_{ll}\pi_{mm} - \pi_{lm}{}^2 > 0$$

The comparative statics effects of market parameters, that is output and input prices, can be solved by applying the Cramer's Rule from

A4.4a $$\begin{bmatrix} \pi_{ll} & \pi_{lm} \\ \pi_{ml} & \pi_{mm} \end{bmatrix} \begin{vmatrix} dl \\ dm \end{vmatrix} = -\begin{bmatrix} (1+\tau) f^i_{l_i} & -1 \\ -(1+\tau) f^i(l_i;q) & l_i \end{bmatrix} \begin{vmatrix} dp \\ dc \end{vmatrix}$$

and the effects of price support and an acreage subsidy from

A4.4b $$\begin{bmatrix} \pi_{ll} & \pi_{lm} \\ \pi_{ml} & \pi_{mm} \end{bmatrix} \begin{vmatrix} dl \\ dm \end{vmatrix} = -\begin{bmatrix} p_i f^i_{l_i} & 0 \\ -p_i f^i(l_i;q) & -1 \end{bmatrix} \begin{vmatrix} d\tau \\ ds \end{vmatrix}$$

Appendix 5 (1/1). Sample equations from calibrated model

Note that all these equations refer to the lowest quality parcel q which is the first parcel. As land quality varies between parcels so will also *a* (in the first parcel 810 for crop 1 and 803 for crop 2) and α (in the first parcel 52.905 for crop 1 and 35.805 for crop 2) vary between parcels.

Private optimum

Profit 1:

(A5.1a) $$\pi_1 = 0.73(1-m_1)(810+52.905*l_1-0.173*l_1^{\ 2})-5.95(1-m_1)*l_1$$

Profit 2:

(A5.1b) $$\pi_2 = 0.83(1-m_2)(803+35.805*l_2-0.094*l_2^{\ 2})-5.95(1-m_2)*l_2$$

Social optimum

Social returns 1:

(A5.2a) $$\begin{aligned}&SW_1 = 0.73(1-m_1)(810+52.905*l_1-0.173*l_1^{\ 2})-5.95(1-m_1)*l_1\\&-9.5(1-2.5m_1)15\exp\left[-0.7+0.7\frac{(1-m_1)l_1}{90}\right]+340m_1^{\ 0.08}\end{aligned}$$

Social returns 2:

(A5.2b) $$\begin{aligned}&SW_2 = 0.83(1-m_2)(803+35.805*l_2-0.094*l_2^{\ 2})-5.95(1-m_2)*l_2\\&-9.5(1-2.5m_2)15\exp\left[-0.7+0.7\frac{(1-m_2)l_2}{90}\right]+340m_2^{\ 0.08}\end{aligned}$$

First-best instruments

Profits under instruments 1:

(A5.3a) $$\begin{aligned}&\pi_1 = 0.73(1-m_1)(810+52.905*l_1-0.173*l_1^{\ 2})-5.95(1+t_1)(1-m_1)*l_1\\&+(\lambda_1-\tfrac{1}{2}\lambda_1 m_1)m_1\end{aligned}$$

Profits under instruments 2:

(A5.3b) $$\begin{aligned}&\pi_2 = 0.83(1-m_2)(803+35.805*l_2-0.094*l_2^{\ 2})-5.95(1+t_2)(1-m_2)*l_2\\&+(\lambda_2-\tfrac{1}{2}\lambda_2 m_2)m_2\end{aligned}$$

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