SLI WORKING PAPER 2005:2

Methodology for Assessing the Regional Environmental Impacts of Decoupling: A Focus on Landscape Values

Mark Brady

Swedish Institute for Food and Agricultural Economics (SLI)

Konrad Kellermann

Institute of Agricultural Development in Central and Eastern Europe (IAMO)

This Working Paper is Deliverable 11 of the IDEMA project. The research is supported by the European Community's Sixth Framework Programme (SSPE-CT-2003-502171)

IDEMA

Swedish Institute for Food and Agricultural
Economics
Box 730
220 07 Lund
http:www.sli.lu.se
Mark Brady
M1ark.brady@sli.lu.se
SLI-Working Paper 2005:2

CONTENTS

1 Introduction

Agriculture is a major user of land and water resources and has a pervasive influence on the environment. Its environmental impacts can have both negative (damages) and positive (services) implications for the welfare of human beings. For example water pollution is exacerbated by chemical and nutrient residues from crop production, whereas managed agricultural landscapes provide environmental services such as; flood control, conservation of wildlife habitat, and preservation of scenic and cultural landscapes (Brouwer, 2002; OECD, 1998a; Hanley, 1991). Economists refer to environmental impacts as externalities because it is not usually in the farmers' interest to consider them in their production decisions. Consequently it is not guaranteed that the "market" can be relied upon to provide an optimal level of environmental quality. Agricultural-environmental interaction is accordingly an important policy concern, especially in the EU.

Currently a major reform of the EU's Common Agricultural Policy (CAP) is being implemented across the EU. A central ingredient in this reform, the Mid-Term-Review (MTR), is the *decoupling* of agricultural support from commodity production. Generally speaking, decoupling means reforming agricultural policies so as to reduce their interference with farmers' production decisions and, ultimately, world commodity markets (OECD, 2001a). Instead of receiving a subsidy per unit of commodity output, they will receive a single farm or income payment not related to the level of commodity output. In this way, society should obtain greater parity between what consumers are willing to pay for agricultural products and levels of commodities produced.

Since the environmental impacts of agriculture are intertwined with the production decisions of farmers, decoupling might also alter the flow of environmental services *provided* by agriculture and levels of environmental damage *caused* by agriculture. There is thus a broad spectrum of potential environmental consequences of decoupling in fact, all of those associated with agriculture. This is because decoupling is a radical policy change and is expected to influence the regional *structure* of agricultural production (Andersson, 2004; Andersson, 2005). It will therefore be necessary to delineate environmental evaluation in IDEMA (2003) to impacts that are likely to have the most serious implications for environmental quality in the regions being studied.

The structure of the farming sector describes the essential characteristics of agricultural activity: what, where and how commodities are produced (E. Goddard et al., 1993). These same characteristics are also intricately related to the environmental impacts of agriculture. Wheat produced on a light sandy soil, close to the coast and using intensive production practices is likely to have a much greater influence on coastal nitrate pollution, than the same crop produced on a heavy clay soil, far from the coast and using less fertilizer. Abandoning arable land in a region where forest is the dominating land use will have a negative impact on biodiversity, where as the reverse might be the case in areas where forest is scarce. If decoupling support from production were to radically influence the structure of production, or rate of structural change, in a particular region then it could also have significant environmental implications for society.

There is a general concern that reductions in agricultural support will lead to reduced commodity production and increased exodus from the sector, especially in less productive agricultural regions, with a consequent loss in environmental values associated with agricultural land use (OECD, 1997). For example, as a result of the Swedish deregulation of agriculture in 1991 (i.e., prior to EU accession in 1995) some 350 000 ha of agricultural land were taken out of commodity production, of which incidentally, 260 000 ha were moved back into production when Sweden joined the EU (Andersson, 2005).

Structural change is though a natural and integral aspect of evolving economies—a precondition for economic development for that matter—and the agricultural sector is no exception. Evolution of the sector in industrialized countries has been characterized by a reduction in the number of farms, increasing average farm size and greater specialization. From an environmental perspective the process of structural change prompts three overriding questions. What is the relationship between:

a) *farm size*,

b) increasing farm and regional *specialization* in commodity production, and

c) changing land use or land *abandonment*

on the flow of environmental services *provided* by agriculture and levels of environmental damage *caused* by agriculture?

Categorical answers to these questions are not likely to exist but they can function as overriding research questions to guide development of an environmental assessment methodology suitable for the aims IDEMA. That is, we need a methodology that might help us to better understand the relationships implied by these questions.

The purpose of this Working Paper is to develop a methodology for environmental evaluation in IDEMA. The development process will involve four stages:

(a) Identify the *key* environmental issues relevant to decoupling (i.e., what are the serious environmental impacts likely to be?).

(b) Identify a theoretical framework suitable for analysing the environmental impacts of decoupling.

(c) Develop relevant environmental indicators.

(d) Implement the environmental indicators in the regional economic model that is to be used for policy analysis in IDEMA, i.e., the Agricultural Policy Simulator (AgriPoliS) model.

Decoupling represents a new challenge for agri-environmental policy modelling because little work has been done on the implications of structural change for the environment, other than *ex post* or historical evaluations (Björklund et al., 1999). Environmental modelling in IDEMA will build on the AgriPoliS modelling system, which has the capacity to simulate agricultural structural change at the regional level (Balmann, 1997). As AgriPoliS has not previously been used for environmental modelling an important contribution of IDEMA is to develop this capability.

The basic approach to be adopted is to incorporate environmental impacts within a production economic framework. That is, environmental impacts are viewed as outputs (e.g., desirable effects such as landscape services) or inputs (e.g., undesirable effects such as water pollution) of a and multi-output and multi-input agricultural production process. Standard production theory can then be used to analyze the impacts of decoupled payments on both commodity production and the environment. An important goal of this work is to base development of environmental indicators on theories developed in the field of landscape ecology and to combine these with economic theory in order to produce a robust ecological-economic model for environmental evaluation.

Section 2 begins with an overview of the general environmental concerns associated with agriculture and ends by identifying the key environmental issues raised by decoupling. Section 3 provides a theoretical analysis of decoupling and the environment using a production economic framework. Section 4 moves from theory to practice and describes how the key environmental impacts should be measured to be relevant for policy analysis. Section 5 is used to develop relevant environmental indicators and Section 6 describes how these indicators are to be implemented in AgriPoliS. Section 7 is used to test the proposed indicators by an illustrative application to the Jönköping County in Sweden. Finally, Section 8 provides a summary and some conclusions.

2 Overview of environmental issues

2.1 Potential environmental impacts

The World Trade Organisation (WTO) is a driving force behind current agricultural policy reform, where decoupling can be interpreted as a step on the road to liberalization of commodity production in the EU. The environmental issues arising from the liberalization debate, that has raged over the past 10-15 years, are therefore directly relevant to decoupling. The "threat" of liberalization has culminated in the *Multifunctionality* paradigm (or *Model of European Agriculture*, MEA, as it is referred to within the EU), which has evolved from the early contention that liberalization of agriculture would, *a priori*, benefit the environment (Anderson and Blackhurst, 1992).

Multifunctionality recognizes that agricultural activity, beyond its primary function of supplying food and fibre (i.e., commodities), also shapes the landscape, provides environmental benefits and contributes to the socio-economic viability of rural areas (OECD, 1998b). It also recognizes that there are negative impacts of agriculture such as pollution that cannot be ignored in the policy debate. The environmental impacts of agriculture are broadly summarized in Table 1.

Table 1. The environmental impacts of agriculture

BENEFITS	*DAMAGES*
Landscape values	Point source pollution Nonpoint-source pollution

Managed landscapes provide ecosystem services that are just as important for human well-being as commodity production. These include flood control, air and water purification, climate modification, habitat for other species and scenic landscapes. These benefits are collectively referred to as *landscape values* because the spatial arrangement of fields, farmsteads, woodlots, wetlands, roads and other landscape components is critical to system function. Landscape values can be divided, somewhat arbitrarily, into five principle sources of benefits. These are the relationship between agriculture and the:

1) maintenance of *biodiversity*,
2) preservation of *cultural heritage*,
3) expansion of *recreation* possibilities (or access to the countryside),
4) *knowledge* pool available to man (i.e., scientific and educational value), and
5) other *amenities* provided by a managed landscape (e.g., flood control, scenic landscapes, etc.)

The myriad of agricultural landscapes that characterise Europe are of course the product of continual agricultural activity over the eons of European history. Since decoupling represents a threat to the very agricultural activity that has not only formed, but also functions to maintain these landscapes (especially in marginal agricultural regions where it may no longer be profitable to produce at all), its potential threat to landscape values seems significant. In other words, there is a strong relationship between agricultural landscape values and agricultural activity by definition. The impact of decoupling on the landscape should therefore be a key environmental issue to analyze in IDEMA.

Agriculture is on the other hand, a major contributor to water quality problems in the EU and around the globe generally. Diffuse pollutants such as nutrients, sediment, pesticides, salts and pathogens enter water resources as a result of soil erosion or in runoff and leachate from agricultural land. This type of pollution is referred to as nonpoint source pollution (NPS) because it is generally impossible, i.e., prohibitively costly, to identify individual sources of pollution. First of all, the emissions of a large number of polluters (i.e., individual farms), usually contribute to the problem but only combined or ambient effects are observable. Secondly, the contribution of residuals from any particular farm or point in a watershed to ambient pollution is influenced by spatial variability in the fate and transport of residuals. That is, it is the amount of residuals that actually enter a water resource that determines damage levels and not simply those that are applied or leave the field. Thirdly, the effect of commodity production on water quality is uncertain because NPS emissions are driven by random weather events such as the amount of rainfall. These characteristics imply that there is no direct or general relationship between NPS pollution damage and agricultural production in itself.

Whilst decoupling is likely to result in overall reductions in the total volume of commodity output and hence the total amount of nutrients and agro-chemicals used in European agriculture, it is not likely to have a significant impact on water quality, the relevant environmental problem. This is because of the indirect relationship between commodity production and NPS pollution damage. The implication is that the relationship between water pollution and output levels or agricultural activity *per se,* can be very weak (Brady, 2003; FAO, 1996). It is therefore unlikely that water quality will be significantly enhanced by decoupling or liberalization generally (Lekakis and Pantzios, 1999). Rather, targeted pollution abatement measures will still be required as well as environmental policy instruments to steer farmers away from using production practices that heighten pollution risk.

In regard to point-sources of pollution agriculture is not a particularly important contributor. The most obvious point source pollutant emanating from agriculture is greenhouse gas emissions (carbon dioxide and methane gas). In total EU agriculture is estimated to contribute to barely 10 % of total EU greenhouse gas emissions. The impact of decoupling on agriculture's contribution, even if significant, will therefore be of little consequence to society as a whole. As such measuring these effects should be a low priority for IDEMA. In another context such as an emissions trading system it would be important to consider agriculture, as there could be cost savings. But in the context of decoupling point-source pollution can hardly be described as a key issue.

2.2 The key environmental Issue: Landscape values

The overriding conclusion of this section is that IDEMA should focus on analyzing the implications of decoupling for landscape provisioning rather than pollution damage caused by agriculture. Benefit issues also dominate the liberalization of agriculture debate in relation to the environment, which indicates their importance to policymakers. It is this focus that has manifested in the concepts of *Multifunctionality* and the *Model of European Agriculture*.

The principle environmental risk associated with decoupling is the loss of environmental benefits referred to as landscape values, that are provided jointly or in conjunction with agricultural commodities. This follows from the fact that eliminating or "decoupling" support from commodity output reduces the returns to commodity production, and hence profitable levels of output and associated agricultural activity. Conversely, pollution issues are of less concern because the direct relationship between commodity output and water quality is generally poor. In other words, decoupling cannot be expected to lead to significant enhancement nor degradation in water quality: so it is unlikely to be an important policy issue. In cases where abandonment of commodity production is likely to result in increased soil erosion (e.g., Italy and Czech Republic) then the impact should be characterized as the loss of a land management service (i.e., an amenity value), rather than a pollution problem.

2.3 Review of relevant agri-environmental modeling

A significant gap in the extensive body of literature devoted to agri-environmental economic research is the production economics of landscape provisioning, especially from an empirical perspective. In contrast, an enormous amount of empirical work has been done on NPS water pollution issues, especially to find cost-effective abatement strategies and policies (Shortle and Horan, 2001). A lot of research has also been done on the demand for environmental services from agriculture what is valued and why, and policies for procuring environmental services (Buller et al., 2000). On the other hand we still have little knowledge about farmers' opportunity costs of “producing” benefits and the degree of dependence of benefit production on subsidies to commodity production.

In their pioneering studies, Peerlings and Polman (2004) use an econometric approach to investigate the joint production of milk and landscape services, and Bonnieux *et al.* (Bonnieux et al., 1998) analyze farmers' decisions to produce or not produce environmental services, also using a microeconomic approach. A number of Finish studies attempt to evaluate the impacts of decoupling on landscape values (Lankoski and Ollikainen, 2003; Miettinen et al., 2004; Lehtonen et al., 2005). Examples of landscape indicators used in these studies include Shannon diversity of land use, linear habitat indeces, nutrient surplus, livestock density, fertilizer intensity and pesticide application, calculated at the sector level. A significant limitation of these indicators is that they do not consider the role of space and landscape mosaic, a fundamental characteristic of any landscape. Neither do they relate the results to a theory of biodiversity valuation that would be relevant for policy makers, i.e., some sort of social welfare function. These though are weaknesses to be built on since the studies are also of a pioneering nature.

It is hoped that this paper will contribute to the literature on landscape provisioning in two ways. First by developing landscape indicators that are grounded in welfare economic theory, so that they can be used to determine whether a change in the structure of agricultural production is “good” or “bad” for the environment (which, for example, is not a characteristic of the Shannon Diversity Index that is frequently used). This is done by combining an economic theory of diversity value (Weitzman Diversity) with theories from landscape ecology (species-area relationship and mosaic theory) for measuring changes in biodiversity and landscape characteristics. Secondly, the indicators will be integrated with a model (AgriPoliS) that is capable of a) simulating structural change in agricultural production and b) the spatial configuration of agriculture (though in an abstract fashion). Neither space nor structural change seem to have featured in previous studies of the impact of decoupling on environmental quality.

The relevant spatial scale for modeling chemical and nutrient pollution is the watershed, however, IDEMA focuses on homogenous agricultural regions. This reduces IDEMA’s suitability for studying water quality issues. On the other hand landscapes can be considered region specific and therefore suitable for studying at the regional scale. In fact AgriPoliS is particularly suited to this purpose as it can model the impact of structural change on the landscape, though in an abstract fashion (a synthetic or statistical landscape is modeled rather than an actual landscape). Focusing IDEMA on issues related to landscape values should therefore improve our chances of making a contribution to the broad field of agri-environmental economics and to deliver policy relevant analysis.

3 The economics of decoupling and the environment

The objective of this section is to conceptualize the relationship between commodity output and the environment in a production economic framework. The nature of interdependencies between commodity and environmental output is important because it will dictate, to a large degree, the environmental consequences of decoupling. The principle environmental interdependencies associated with agriculture can be described in terms of: *joint production*, and *complements* or *substitutes* in production (OECD, 2001b).

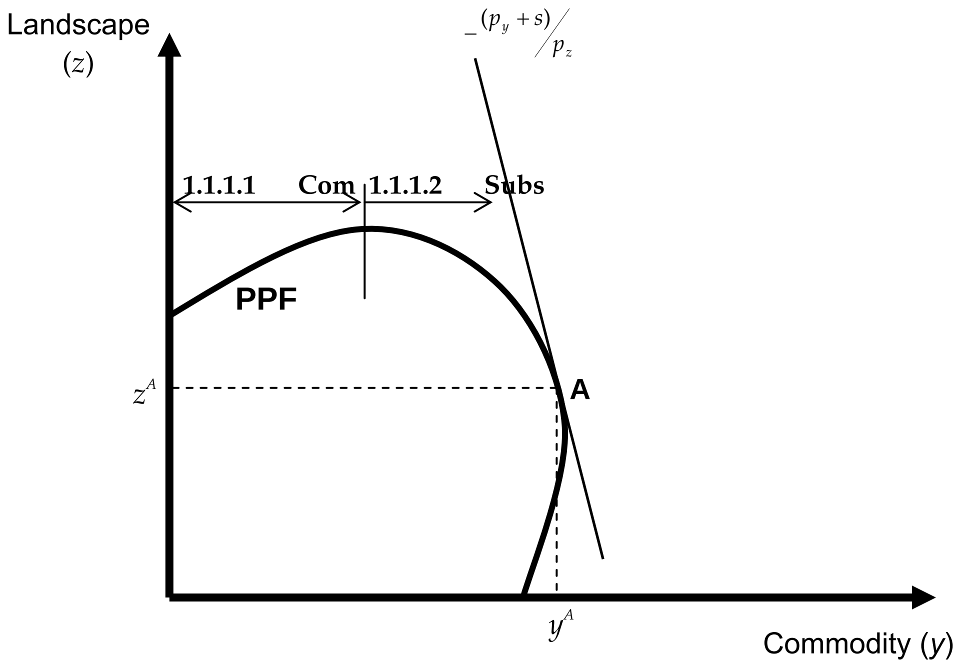
Joint production between commodities and non-commodities implies that their production or generation is linked, so that a change in the supply of one output affects the supply of the other. Most output linkages are attributable to either *non-allocable inputs* (they are produced on the same land, e.g., beef and grazing land) or *technical interdependencies* in which case the production of commodities has a concomitant effect on the environment, e.g., crop fertilization and nutrient pollution.

Commodity and non-commodity outputs can be complements or substitutes in production, depending on the underlying production relationship. Lowering commodity supply may reduce pollution caused by a technical interdependency, whereas raising supply may expand the level of environmental services.

The evidence indicates that the relationship between commodity production and landscape provisioning is complimentary up to a certain level of commodity output after which it becomes competitive (OECD, 2001a). This relationship is illustrated in Figure 1 in terms of the production possibilities frontier (PPF) for a representative commodity y measured in tonnes and a composite landscape indicator, or metric, z (which, at this stage, is simply viewed as an abstract indicator of the total value of the agricultural landscape to human beings).

The supply of commodities and non-commodities alike is a multi-output and multi-input production process. It is therefore preferable to define production possibilities in terms of the cost function rather than a fixed quantity of inputs (Chambers, 1988). Following Romstad (1999) the production possibilities frontier in Figure 1 for commodities (*y*) and landscape (*z*) represents therefore the constant minimum cost of producing any combination of *y* and *z*. This is a more flexible or realistic approach than restricting the farmers' production possibilities to a fixed bundle of inputs (i.e., the elementary approach).

Figure 1. The relationship between commodity and landscape production



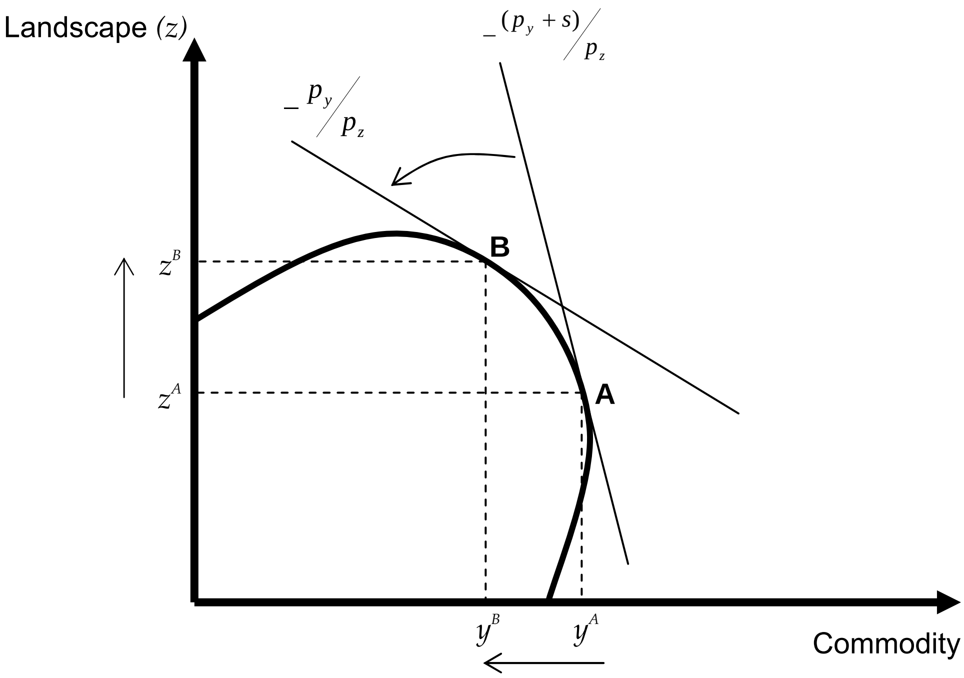
Notice how the PPF described in Figure 1 differs from the familiar frontier for competitive products. For example that for wheat and barley would have negative slope all over to indicate that a trade-off exists between the two outputs: one cannot produce both on the same unit of land. Instead the relationship between *y* and *z* is initially shown to be complementary, the PPF has positive slope, after which it becomes competitive, the PPF takes on a negative slope. Increasing the area of land in commodity production can for instance, increase biodiversity and contribute to the beauty of the landscape especially in regions where agricultural land use is scarce (e.g., Sweden's forested inland). However, at higher levels of y the relationship becomes competitive: greater commodity output brought about by increasing intensity can for example harm biodiversity through the heavy use of chemicals and nutrients (e.g. Sweden's intensively farmed southern plains).

To start the analysis of the implications of decoupling for landscape values assume that the price of commodity y is p_y and that there exists a production subsidy per unit commodity output equal to s. The incentive price relevant for farmers' output decisions is therefore $p_y + s$. There is also an environmental payment p_z based on the level of z (if zero the price line would simply be vertical to indicate maximum commodity output would take place without regard to the environment). The optimal or profit maximizing output mix of y and z given the PPF in Figure 1 is determined by the relative prices $(p_y + s)/p_z$ and occurs at the tangency of the price line $-(p_y +s)/p_z$ and the PPF, i.e., point **A**.

Given the assumed higher returns to commodity production relative to landscape which is illustrative of the current situation in the EU farmers' tend to produce beyond the point that provides maximum landscape values. Commodity production is so high or intensive that it crowds out attainable landscape values. The question is then, how might decoupling affect the levels of y and z?

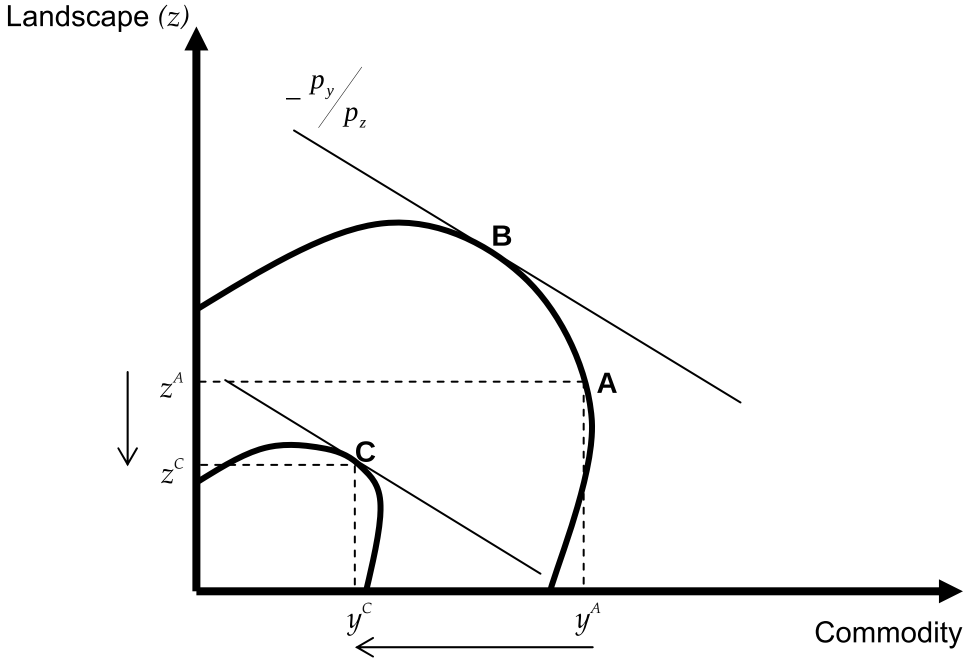
For clarity, decoupling can be analysed as a two-stage process. First, the direct support to commodity production is eliminated: the incentive commodity price relevant to farmers becomes p_y and p_z is assumed to remain unchanged. In the second stage, an income or single farm payment (SFP) is introduced, the size of which is related to the amount of support previously received by each farm but without the requirement to produce commodities. Eliminating production support implies that the slope of the price line declines to $-p_y/p_z$. Optimal production now occurs at point **B** in Figure 2**,** resulting in lower supply of the commodity and an increase in landscape output, measured as z: landscape provisioning increases to substitute for reduced commodity output.

Figure 2. The direct or substitution effect of reduced commodity support



However, as Romstad (1999) points out, this is a naïve understanding of the potential affects. For a marginal price change this would be interpreted as the comparative-static effect. In contrast, elimination of production support involves a radical reduction in price and this can be expected to result in a contraction in the PPF as farmers find better uses for scarce resources such as own-labour and capital. This gives rise to the optimal combination of outputs represented by point **C** in Figure 3 given relative prices $-p_y/p_z$.

Figure 3. Production possibilities on elimination of commodity support



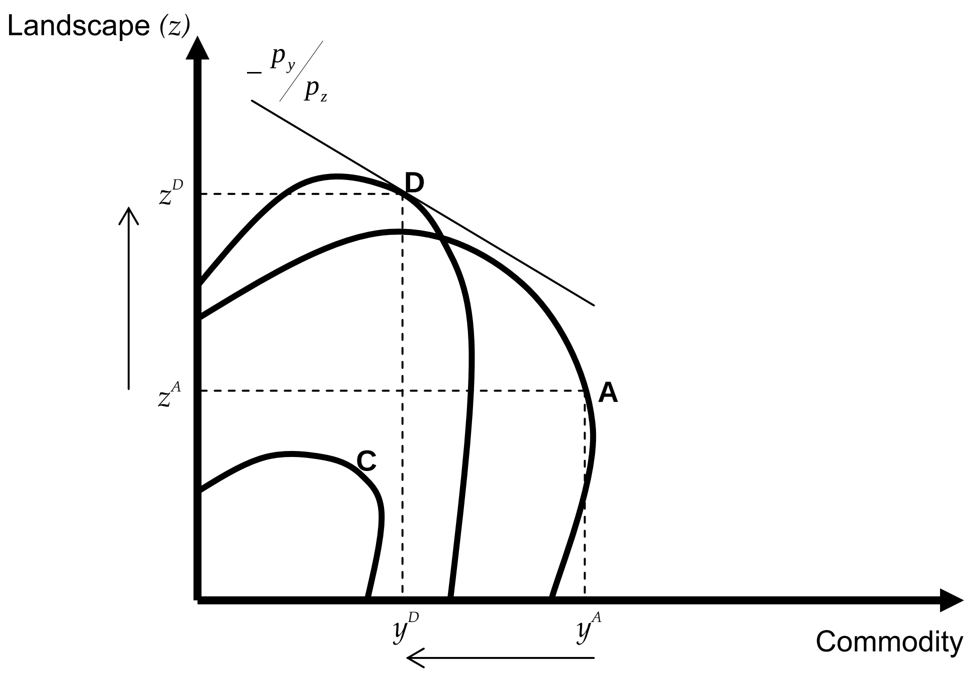
This can be interpreted as the *doom-and-gloom* scenario of critics of agricultural liberalization, and drives the multifunctionality criticism of decoupling agricultural support. Subsidized commodity production falls dramatically but so does the provision of valued non-commodities such as landscape. Current environmental support p_z is "not sufficient" to maintain a "desirable" level of landscape.

In the case of decoupling this conclusion is probably too gloomy or an equally naïve interpretation of its consequences, as there will be a number of potentially strong, but positive, indirect production effects. First of all, farmers are given greater freedom to choose production activities since support is no longer tied to any particular output, e.g., grain or beef. This implies that the costs of achieving a certain level of output should fall if a constraint on the farmers profit maximization problem is eliminated then costs cannot possibly increase. Secondly, income support in the form of the single-farm-payment could augment the choice of on-farm labour supply as on farm income becomes more certain. Thirdly, risk-averse farmers are known to increase output in response to income support (farm income is less exposed to the vagaries of agricultural production). Finally, increased income can influence investment decisions if credit constraints exist. Andersson (2004) elaborates on these effects.

Production neutral income support in the sense that the income payment is not tied to the level of commodity output has therefore the potential to expand the farmers' opportunity set. Further, it will become relatively more profitable to devote scarce resources to landscape management than commodity production as the relative price of commodity output declines (i.e., the opportunity cost of devoting inputs to landscape management declines).

The optimal output mix with decoupling could theoretically move to a point such as **D** in Figure 4. At point **D** there is significantly less commodity output than with price support, point **A**, but more than with liberalization, point **C**. In contrast the level of landscape provisioning obtainable with commodity production is almost maximized; a consequence of the higher relative price for providing landscape.

Figure 4. Production possibilities with decoupled income support



This conclusion though is also an extreme “theoretical” case and as such is probably overly optimistic. In reality the optimal output mix is likely to lie somewhere between the extreme points **C** and **D**, depending on the characteristics of individual agricultural regions and the levels of environmental (and national) support. Nevertheless the landscape impacts of decoupling may be favourable or not too negative due to the increase in relative returns to landscape management and reductions in production intensity which occur at both points **C** and **D** (generally, intensity has a negative relationship with environmental quality, especially for water quality and biological diversity). The implications of decoupling for landscape values remain therefore an empirical issue. On the other hand the impact on pollution should be unequivocally to reduce it (given that farmers have not faced significant credit constraints for purchasing chemical inputs which is conceivable in some new member states).

Another implication of the analysis is that the effects of decoupling will vary from region to region, since the relative returns from commodity production and environmental schemes vary significantly between regions. For example, in marginal regions where there is significant national support and Pilar II schemes the effects of decoupling might, to a large extent, be absorbed by these other sources of income.

In summary, considering decoupling in a dynamic context (i.e., over the long term) implies that the losses in environmental values predicted by a static analysis are likely to be overestimated. Likewise the increase in environmental payments required to maintain current levels of environmental quality are likely to be lower than the reductions in total support that farmers may experience (an implication that is easily overlooked if one relies on simple “engineering” type calculations that assume the need for 1:1 compensation). These conclusions stem from the facts that the cost of producing commodities and non-commodities alike should decline as a consequence of decoupling, and that environmental consideration should become relatively more profitable. Finally, marginal areas are relatively less dependent on Pillar I or coupled agricultural support, because they receive extensive support from other sources (i.e., Pilar II and national support). In the next section I begin the task of developing indicators of landscape value that can be used for empirical analysis of decoupling and landscape values.

4 The economic approach to evaluating landscape impacts

Section 3 conceptualized the relationship between commodity production and the environment, especially landscape values, in a very abstract fashion. The next step is to bridge the gap between theory and practice, and explain more precisely what characteristics of agricultural landscapes are valued by society, how these are influenced by farmers' production decisions and how they might be measured in a way that is relevant for policy evaluation (i.e., cast in a welfare economic framework).

4.1 What is a landscape?

A landscape is a pattern of heterogeneous landforms, vegetation types and land uses that is repeated in a similar format throughout (Urban et al., 1987). The basic structural unit of the landscape is the landscape element, a relatively homogeneous area surrounded by elements of different composition. Elements such as roads, streams, hedgerows, stonewalls, unfenced lines, etc. are referred to as linear or line elements. Nonlinear elements are usually referred to as *patches*, which are defined as contiguous plots (i.e., plots having a shared border where a plot or grid is the smallest unit of area, usually a square, that is used to study a landscape) of land with similar characteristics (e.g., a wheat field).

4.2 Influence of farmers' production decisions on landscape values

Since agricultural landscapes are managed landscapes they can be characterised in terms of how they are managed. Three types of production decisions have been found to characterise farmers' management of the agri-landscape (OECD, 2001b; Romstad et al., 2000):

- *land use* which is the area of agricultural land that is put to a particular use or crop,
- the *intensity* of production which is the amount of commodity output per unit area (usually measured as nutrients/ha, chemicals/ha or livestock/ha), and
- the *mode* of production which is the farmer's choice of land management practices.

The combination of intensity and mode of production is also referred to as management or farming practices. These terms are used interchangeably when convenient.

Furthermore the value of landscape provisioning by any particular farm will also depend on:

a) the particular *characteristics* of the farm or region in which commodity production takes place, and

b) structural *interdependencies* between farms in the landscape or region of interest (e.g., how each farm contributes to the mosaic of the landscape).

Interdependencies exist because people value landscapes based on their total impression rather than individual farms (Berland, 1994). That is they value a combination of landscape components higher than each component in isolation. This fact supports the conclusion that mosaic is an important determinant of overall landscape value. The landscape can also mean different things to different people, which can make it difficult to define, classify and value landscapes. Thus it seems more important to quantify various aspects of landscapes that decision-makers can take into consideration, rather than trying to identify a unique measure of landscape value, which is not likely to exsit.

In the ensuing sub-sections the concepts of land use, intensity and mode of production are related to specific landscape values.

4.2.1 Biodiversity

The relationship between the area of an agricultural crop and biodiversity is initially complementary however turns competitive after a certain point. For example when agricultural land is no longer scarce or when fields are enlarged and merged and the amount of border zones are reduced, resulting in loss of microhabitats. Increasing intensity is generally negative for biodiversity, since it implies increased use of chemical pesticides and nutrients, and higher livestock densities. Broad spectrum pesticides kill not only target species, but even desirable species and nutrient residuals risk polluting water. It is however possible to imagine situations where intensity is too low. For example, reduction in grazing pressure could be detrimental to values provided by semi-natural grazing lands. Increased livestock intensity might therefore benefit diversity even if more animals are reared in stables. Mode of production is also important for biodiversity. Grassland that is grazed by ruminants is generally richer in species than land that is used for silage production. Mechanical weed control can be more favourable for biodiversity than using herbicides (e.g., as practised in organic agriculture).

4.2.2 Cultural heritage

An indicator of cultural heritage value is the number of active or full-time farmers that can maintain the cultural basis of farming (but even other aspects such as farming skills that are important for maintaining biodiversity). A high and increasing level of intensity in agriculture will most likely have a negative impact on cultural heritage, as fewer farmers are required to produce. Variation in the mode of production also adds to cultural values as knowledge of different agricultural practices is maintained. Specialisation of production and production techniques is therefore likely to have a negative impact on cultural heritage.

Many of the point and line elements of the landscape can be provided or maintained independently of commodity production. This includes features such as historical buildings, stonewalls, hedgerows, etc. Environmental policy is therefore more relevant to the preservation of these elements than coupled commodity payments. On the other hand, large area elements (patches) of the landscape are usually closely linked to commodity production, e.g., maintenance of grasslands.

4.2.3 Amenity value

Amenity value is closely linked to diversity in the agricultural landscape and mosaics with the natural landscape, e.g., forests and lakes. Amenity might also be seriously affected by the intensity of production and the mode of production. Increasing intensity of production is likely to have a negative impact on amenity value. Merging of fields and increasing intensity of livestock production increase monotony in the landscape and hence reduce amenity value.

If the value of landscape is attached to the *productive* landscape that reflects productive activity, then area actually farmed is of vital importance. In this case, idled land would have relatively low amenity value, or lower value compared to productive agricultural activities. Accordingly the linkage to agriculture depends on what is looked upon as valuable; the landscape itself or that the landscape mirrors activity.

Variation in modes of production adds to the diversity of the landscape by contributing different components; grains or vegetables, gazing livestock, activity at different seasons, etc. All of which would seem to add positively to the amenity value of the landscape.

4.2.4 Recreation or access to the countryside

The visual features of a landscape are *use values* and therefore have to be evaluated in conjunction with facilities that increase recreation value to potential users (OECD, 2001b) . These include access paths, hiking trails, picnic areas and nature discovery trials. Since there is unlikely to be any interdependencies between access and commodity production, the issue is not particularly relevant to decoupling. Enhancement of recreation values is better studied in an environmental policy framework, as it requires services in addition to commodity production. Accordingly, the issue of recreation and access should only be addressed indirectly in IDEMA, in the sense that certain land uses might be more conducive to recreation than others. Instead, the focus will be on amenity values as they relate to commodity production.

4.2.5 Scientific or educational value

Knowledge value is closely related to diversity. As shown previously maximising diversity value is equivalent to maximising potential knowledge. A loss in diversity of the landscape is therefore paramount to a loss in knowledge. Potential knowledge represents an option value, and is one of the risks of irreversible effects. Land abandonment or idling of agricultural land is therefore likely to have a negative impact on knowledge value. Likewise, specialisation of production and increasing intensity of production are likely to reduce knowledge value (e.g., replacement of traditional farming practices with chemical solutions or loss of genetic variety in plants and domestic animals).

4.3 Characteristics of landscapes and farmers' production decisions

In order to evaluate the effects of decoupling or any other policy change on the landscape, it would be useful to be able to generalise about the relationship between specific characteristics of landscapes as they are affected by farmers' decisions, and overall landscape value. The question is, whether there are any universal characteristics of landscapes that have a strong relationship to their overall value? Species composition and population viability are for example, often affected by the structure of the landscape; for example, the size, shape, and connectivity of individual patches of ecosystems within the landscape (Noss, 1990).

Two characteristics of landscapes that have the potential to meet these requirements are landscape diversity (or heterogeneity) and mosaic. Landscape heterogeneity is a particularly important, even central, determinant of landscape value (OECD, 2001b). In general, landscape diversity and mosaic seem to influence each of the five landscape values described above biodiversity, cultural, amenity, recreation and knowledge value positively. Thus measures of landscape diversity and mosaic should be a useful basis for developing general indicators of landscape value. A problem for policy analysis at the regional level is that landscape benefits are difficult to measure or quantify directly.

In the next section I present an approach for measuring the biodiversity value of the agricultural landscape based on information about farmers' production decisions and the biophysical characteristics of an agricultural region. This is followed in Section ?? by measures of landscape diversity and mosaic. After this in Section ?? an economic model of farmers' decision making is introduced that describes the relationship between exogenous market and policy variables and optimal production choices. The proposed landscape indicators are subsequently linked to the economic model to analyze the theoretical impacts of agricultural policy reform, specifically decoupling, on biodiversity and mosaic. Finally in Section ?? it is shown how the indicators have been integrated with the AgriPoliS model and some test results are presented.

5 Measuring the value of biodiversity in landscapes

Agricultural landscapes are important for the conservation of biodiversity in Europe because they provide habitat for a wide range of species. An important concern for the EU is whether *decoupling* the Common Agricultural Policy (CAP) from commodity production will impact the diversity value of agricultural landscapes. The aim of this section is to develop a model of the biodiversity value of an agricultural landscape as affected by farmers' production decisions. I begin by defining biodiversity and developing a *value-of-biodiversity* function that relates farmers' land use decisions to the value of biodiversity in the landscape.

5.1 The value of biodiversity

Diversity is defined as "*the condition of being different or having differences*". Weitzman (1992) recognized though that such a lose definition of diversity is of little use for economic analysis. To evaluate changes in diversity we need to be able to value it also. He therefore set about to define a meaningful value-of-diversity function, where the value of diversity could be calculated for a species, subspecies, a specimen, an object, or almost anything else depending on the context. Similarity and hence nonadditivity are at the core of the concept of diversity (Nehring and Puppe, 2002): more of the same thing adds nothing to diversity. Only the addition of something *different* to a set of "things" will increase the diversity value of the set.

The question I wish to answer is how to value the biodiversity of an agricultural landscape. Weitzman has provided a theory for going about finding an answer. Avoiding the rigorous mathematics that Weitzman uses to prove his theory (and which has been generalized by Nehring and Puppe (2002)), I will instead focus on clarifying how diversity should be measured according to Weitzman (i.e., from an economic perspective).

The degree of dissimilarity or difference between any pair of "land uses" *(i,j)* in some set *S* of *n* potential land uses, is their distance *d(i,j)*, which takes as given that there is a cardinal measure of distance between the two land uses and satisfies

$$d(i,j) \geq 0 \tag{1.1}$$

The distance between two land uses might be derived as a weighted sum of distances between more fundamental micro-characteristics, so that *d(i,j)* represents the weighted number of observable differences between land-use *i* and *j*. For example, the micro-characteristics might involve the number of species supported by the land use, age and type of the vegetation cover, associated soil management practices, distinguishing features, location and so forth. The problem or issues to be studied should determine the dissimilarity measure. For the time being, assume that it is possible to measure the difference between any two land uses and the difference we are concerned with is the number of species.

To illustrate the meaning of the value of diversity, consider the following example. Assume there are two species, 1 and 2, that we are interested in conserving. Let E_1 stand for the number of genes distinctive or unique to

Species 1 (i.e., $d(1,2) = E_1$), while E_2 are those distinctive to Species 2. Denote the number of genes borne jointly or held in common by both species as *J*. Then the number of genes supported by each species, M_i, are $M_1 = E_1 + J$ and $M_2 = E_2 + J$. The diversity function in this case is $V(S) = J + E_1 + E_2$. Further, the marginal diversity values of each Species are E_1 and E_2 respectively given the existence of the other species. This is because the diversity value of any particular species is its contribution of unique genes and not those held in common.

> *Corollary 1: the marginal biodiversity value of an agricultural land use is its contribution of unique species to society and not those held in common with other land uses.*

Since species evolution follows a hierarchical structure, species diversity can be represented by a tree structure, Figure 5. The value of diversity corresponds to the total length of the tree, i.e., $V(S) = J + E_1 + E_2$. The value of diversity of any hierarchical structure can be represented in this way (By hierarchical, it is meant that some "thing" is assumed to have evolved by "descent with modification" from a common ancestor).

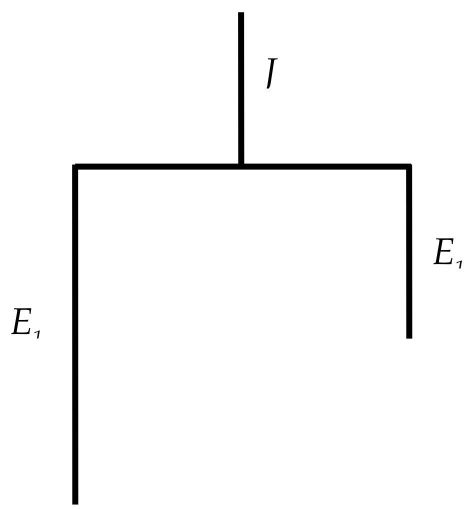


Figure 5. Evolutionary tree representation of value of diversity

This though is only part of the story. In the real word there is uncertainty and each species will run some risk of extinction. The valuation of diversity in an uncertain world should therefore consider the probability of *survival*. The relevant measure of diversity value in this case is the *expected value* of diversity.

5.1.1 Implications of uncertain species survival

Conserving biodiversity is a risky business, because we can never be certain that a species will survive given a course of conservation measures. In the extreme, a new disease might wipe out an entire species no matter what precautions have been taken. Hence, to be meaningful for policy analysis, a model of biodiversity conservation needs to be based on the probability of species survival. The aim of this section is to analyze the implications of uncertainty for valuing biodiversity and then to extend the analysis to valuing biodiversity in an agricultural landscape.

To illustrate the distinction between uncertain conservation of species and a deterministic process of counting species, let us return to the example provided in Figure 5. Uncertainty is introduced as follows. Let ρ_1 be the probability of survival of Species 1, and ρ_2 the probability of survival of Species 2. The appropriate stochastic measure of diversity is the probability weighted diversity of all unique and common genes born by each species. The expected diversity function, denoted $W\left(\rho\right)$, is the average number of different genes borne by each species:

$$W\left(\rho_1,\rho_2\right)=M_1\rho_1+M_2\rho_2-J\rho_1\rho_2\,. \qquad (1.2)$$

That is, the expected value of diversity is the probability of Species 1 surviving multiplied by the genes borne by 1, plus the probability of Species 2 surviving multiplied by the genes supported by 2, *minus* the joint probability of both species surviving multiplied by the number of genes held in common. In the absence of uncertainty, both survival probabilities would be 1, and the expected value of diversity *W(ρ)* would be equal to the value of diversity function *V(S).*

The marginal value of conservation efforts is found by differentiating the expected diversity function *w.r.t.* survival probabilities.

$$\frac{\partial W\left(\rho_1, \rho_2\right)}{\partial \rho_1} = M_1 - J\rho_2 . \tag{1.3}$$

This expression states that a small improvement in the survival probability of Species 1 will increase expected diversity in the landscape by the number of genes supported by 1 less the expected number of genes supported jointly by Species 2. In a certain world, survival probabilities would be equal to 1 and marginal expected diversity value would equate with diversity value (i.e., $E_1 = M_1 - J$). The intuition behind Equation (1.3) is that marginal diversity and distinctiveness are the same concept in the context of an evolutionary tree model (Weitzman, 1998, 1293). In other words the marginal diversity value of a species is equal to its diversity value.

A number of important conclusions emerge from the concept of the measure-of-diversity function, Equation, (1.2).

- When the diversity function is being maximized we are also minimizing the probability of diversity being lost (e.g., the loss of a gene or other biological information that might be useful in the future).
- The most valuable "species" or land use is the farthest distance from the others (e.g., semi-natural grazing land or meadow).
- Let *k* represent a reference land-use in any particular set of land uses that characterise a landscape. The utility of any land-use of interest is always measured relative to the reference *k* and is equal to the "distance" of that object from k (e.g., in Sweden *k* is forest).

Finally, according to Weitzman, a good theory of diversity should pick up the following themes:

i) If two land uses share very little in common, then their joint utility should be approximately the sum of their individual utilities, i.e., $U(1+2) = U(1) + U(2)$.

ii) If substitutes then $U(1+2) < U(1) + U(2)$ U(1+2) < U(1) + U(2).

iii) If complements then $U(1+2) > U(1) + U(2)$.

I intend to develop a measure of diversity for agricultural landscapes that has at least these characteristics.

5.1.2 Expected value of biodiversity in an agricultural landscape

In this section I extend the model of expected diversity value of a set of species, Equation (1.2), to an entire landscape harbouring any number of species. At the landscape level habitat *area* has been identified by ecologists as the primary regulator of species diversity (Rosenzweig, 1995). The presence of a particular habitat improves the survival chances of species that can be supported by that habitat. Farmers, through their land use decisions control the availability of habitat in agricultural landscapes. Consequently, it is through their land use decisions that farmers influence the probability of species survival.

When studying the impacts of agriculture on biodiversity—or the environment in general—it is important to distinguish between *what* is produced and *how* it is produced. Two farms producing exactly the same outputs, say wheat and milk, might have significantly different environmental impacts depending on the type and quantity of inputs being used or how soil is managed. We should therefore think of the land use decision as comprising of the following three component decisions, each of which impact the quality of habitat given the biophysical characteristics of a particular landscape; i) type of *crop* or land cover (e.g., wheat, maize, grass, etc.), ii) crop *area* and iii) land *management practices* (e.g., chemical intensity, grazing density, tillage practices, etc.).

Before I define a measure of the value of diversity of an agri-landscape note the following. Denote the set of species that inhabit a landscape as S, and any particular species in that set as $s \in S$. Contiguous or adjoining plots of agricultural land will be referred to as *blocks*, where J is the set of blocks in a landscape and $j \in J$. The set of crops or land uses available to farmers is I, where $i \in I$. A field (or patch of agricultural land) is defined as a contiguous area of a particular crop grown on a particular block, hence the set of potential fields is $\{i, j\} \subset I \cup J$.

To determine the probability of a species surviving in a landscape, it is necessary to begin with the probability of a species surviving in a particular field (i.e., the probability of species survival given the existence of suitable habitat). This is because a landscape is defined by an arrangement of different field or patch types and sizes. The conditional probability of species s, surviving in any particular field, P_{sij}, can be expressed as:

$$P_{sij} = P\left(s \mid a_{ij}, \mathbf{v}_{ij}\right) = \left[0,1\right], \tag{1.4}$$

where a_{ij} is the area of field (i, j) and $\mathbf{v}_{ij} \in \Re^{+}$ is a vector of management options associated with field (i, j). Equation (1.4) states that the probability of species s being found in patch (i, j) is conditional upon events a_{ij} and $\mathbf{v}_{ij}$ occurring, and that the probability takes on a value between 0 and 1.

This is the most plausible way to model species survival in the agri-landscape, because farmers influence biodiversity via their production decisions. Since we are only interested in modeling the impacts of farmers' decisions on diversity (and ultimately that of agricultural policy), I assume that all other factors that might influence the probability of survival remain unchanged.

The expected value of diversity at the landscape level will be

$$E\left[\mathrm{v}(s)\right] = \sum_{r \in R} P_r - \frac{1}{2} \sum_{r \in R} \sum_{t \in R} P_r P_t \tag{1.5}$$

where $R = S \cup I \cup J$. Equation (1.5) states that the expected value of species in the landscape is equal to the sum of the probability of species s surviving in field (i, j) less the sum of the joint probabilities of the species surviving in other fields. Whilst Equation (1.5) provides an exact definition of the value of diversity, it is not particularly helpful for empirical analysis, since survival probabilities are not generally known. Instead we need to approximate $E\left[\mathrm{v}(s)\right]$ using indirect methods. I begin the process of developing an indirect measure by studying the relationship between agriculture and species.

5.2 Joint production of commodities and species

5.2.1 Species survival and farmers' production decisions

Let us now examine the relationship between species survival and the farmers' decision variables a_{ij} and $\mathbf{v}_{ij}$ more closely. To do this, I draw on research results from the field of landscape ecology, where the relationship between land use and population dynamics has been studied. One of the most robust and useful results, is the *species-area* relationship (Armsworth et al., 2004, 120). If one graphs the number of species, Q, (often referred to as species *richness*), in a patch of intermediate size against the area of the patch, A, then the data are well approximated by a power law (or as economists will recognize a Cobb-Douglas production function):

$$Q = cA^z \tag{1.6}$$

where c and z are positive constants. c can be thought of as the species *productivity* factor of a particular land use or habitat. The higher c the more species a habitat is likely to support. z on the other hand is a scale parameter and tells how species productivity changes in response to area. Typically z falls within a narrow range, 0.18-0.25, for a diverse suite of ecosystems (Rosenzweig, 1995).

Differentiating Equation (1.6) *w.r.t.* area A, yields marginal species productivity (i.e., marginal diversity value of habitat A):

$$\frac{dQ}{dA} = zcA^{z-1} > 0 \quad \text{for} \quad z > 0\,. \tag{1.7}$$

Taking the second derivative yields the rate of change in species production

$$\frac{d^2Q}{dA^2} = \left(z^2 - z\right)cA^{z-2} < 0 \quad \text{for} \quad 0 < z < 1\,. \tag{1.8}$$

Thus for values of $0 < z < 1$ the species-area relationship is concave which is illustrated in Figure 6. That is, the number of species increases with patch size but at a decreasing rate. Further, the average number of species per unit area is decreasing in patch size. In economic terms this translates to the species-area relationship being characterised by decreasing returns to scale.

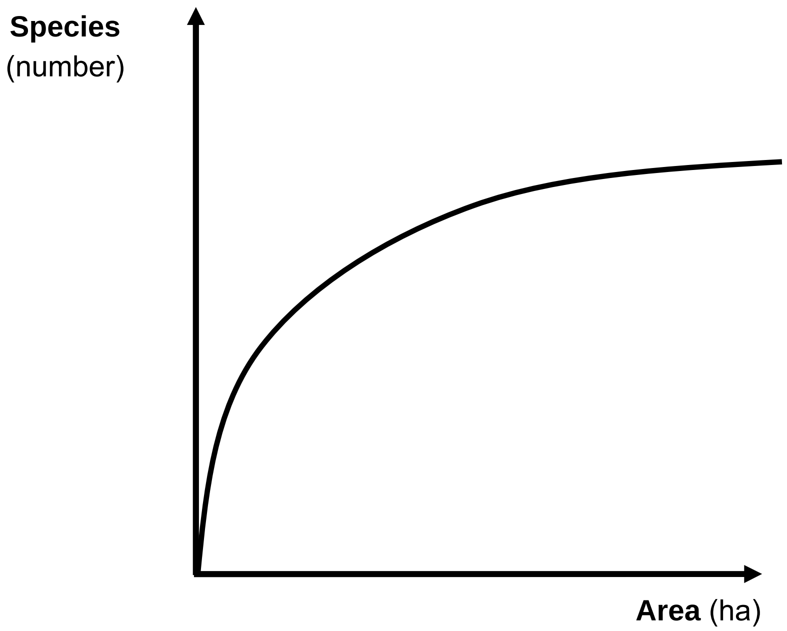


Figure 6. Species-area relationship

The species-area function has a number of characteristics that are both appealing and potentially very useful for economic analysis. First, Equation (1.6) is a homothetic function because it is homogeneous of degree z. This implies that only relative values of c are needed to rank different land allocations in terms of their contribution to the value of biodiversity. For example in the absence of absolute values of the parameter c we could use relative weightings in an empirical analysis. For instance if ecological research indicated that grass land is 10 times more valuable than grain from a diversity perspective, then this would be enough information to rank different land use patterns in terms of species production.

Further, decreasing returns implies a species trade-off exists given that the area of agricultural land will be limited in any particular landscape. First, it will never be optimal for an agricultural landscape to be fully covered by any single land use. The optimal mix of crops (i.e., the mix that maximizes expected species diversity) will involve positive levels of all land uses (an interior rather than corner solution), which is also consistent with the Mosaic hypothesis (Section). Secondly, reductions in the area of crops that take up relatively little area are likely to have a greater impact on biodiversity than an equivalent reduction in the area of a large crop, because of decreasing returns to scale.

To derive the species-area relationship for an agricultural land use we can rewrite Equation (1.6) as

$$\#\left(Q_{\{i,j\}}\right)=c_i a_{ij}^{z_i} \tag{1.9}$$

where $Q_{\{i,j\}} \subset S$ is a subset of species supported by patch $\{i,j\}$ of size a_{ij}, and c_i and z_i are parameters specific to patch type *i*. The function # counts the number of species in the set $Q_{\{i,j\}}$. Since it has been established that $z_i < 1$ for all *i*, then species production will be a concave function of patch size.

Equation (1.4) indicates that species survival probabilities in an agricultural landscape are also influenced by the way a farmer manages his fields, as represented by the choice vector $\mathbf{v}_{ij}$. Two particularly relevant decisions for biodiversity are chemical intensity and livestock density. Another might be soil management practices. These decisions impact biodiversity because they affect the *quality* of habitat and hence the ability of species to survive in a particular patch.

Since the scale factor *z* has proven to be robust and does not vary significantly between habitat types, we can safely say that it is the parameter, *c*, that distinguishes the species productivity of different habitat types. I therefore model the impact of farming practices on species productivity as $c_i = \mathrm{c}_i\left(\mathbf{v}_{ij}\right)$, where $\mathbf{v}_{ij} \geq 0$. Consequently the species-area relationship can be modified in the following way to incorporate the impact of management practices on species productivity:

$$\#\left(Q_{\{i,j\}}\right)=\mathrm{c}_i\left(\mathbf{v}_{ij}\right)a_{ij}^{z_i}. \tag{1.10}$$

Since this function is still homothetic, changes in c_i will simply shift the species-area curve up or down, which ever the case may be. It will never however cause the curves to cross each other, which would result in loss of the ranking property. This property is illustrated in Figure 7.

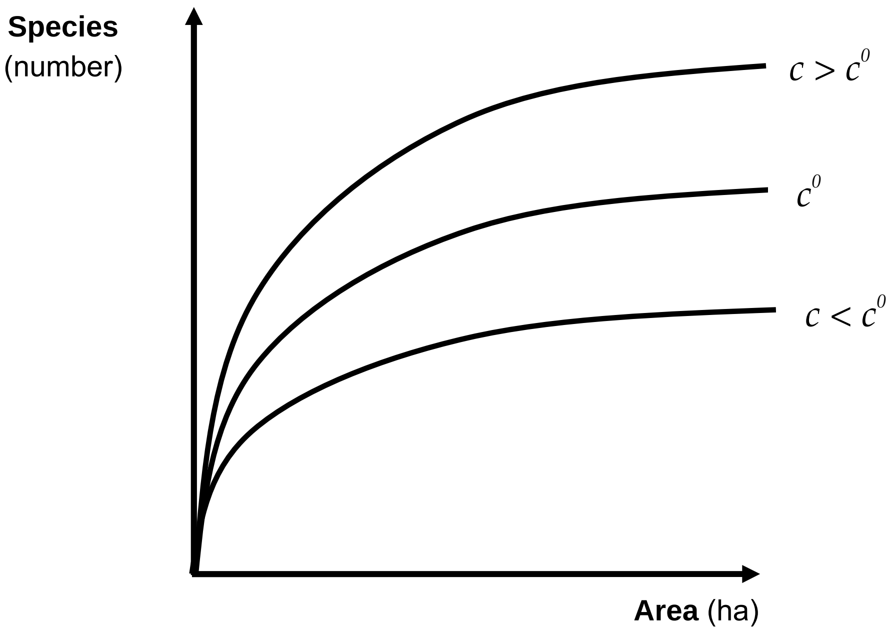


Figure 7. The impact of changing management practices on species-area relationship

5.2.2 Analysis of the species-intensity relationship

Farmers apply chemicals to their fields to improve habitat quality for agricultural crops. For example fertilization improves yields and spraying reduces competition for habitat space from other species: insects, weeds, fungi, etc. Intensively farmed land tends therefore to be dominated by one particular species, the agricultural crop. The implied relationship between farming intensity and species produvtivity is illustrated in Figure 8. The convex shape of the species-intensity graph indicates that diversity decreases with intensity and at an increasing rate.

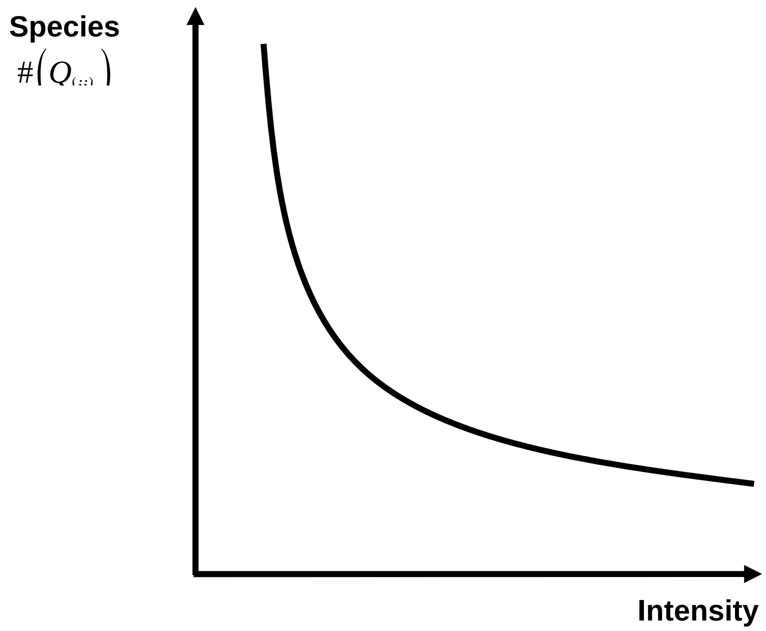


Figure 8. The species-intensity relationship

Convexity of the species-intensity relationship, Figure 8, implies that $\partial \mathrm{c}_i\left(\mathbf{v}_{ij}\right) / \partial v_{ij}^{\text{int}} \leq 0$ and $\partial^2 \mathrm{c}_i\left(\mathbf{v}_{ij}\right) / \partial v_{ij}^{\text{int}2} \leq 0$ whilst keeping area constant.

To determine the impact of intensity on the species-area relationship, I take the first and second partial derivatives of Equation (1.10) *w.r.t.* the intensity variable, $v_{ij}^{\text{int}} \in \left\{\mathbf{v}_{ij}\right\}$:

$$
\begin{aligned}
\frac{\partial \#\left(Q_{\{i,j\}}\right)}{\partial v_{ij}^{\text{int}}} &= \frac{\partial \mathrm{c}_i\left(\mathbf{v}_{ij}\right)}{\partial v_{ij}^{\text{int}}} c_i a_{ij}^{z_i} \quad \leq 0 \\
\frac{\partial^2 \#\left(Q_{\{i,j\}}\right)}{\partial v_{ij}^{\text{int}2}} &= \frac{\partial^2 \mathrm{c}_i\left(\mathbf{v}_{ij}\right)}{\partial v_{ij}^{\text{int}2}} c_i a_{ij}^{z_i} \quad \leq 0
\end{aligned}
\tag{1.11}
$$

These relationships imply that the species-area curve becomes flatter as intensity increases. As such, it will lie entirely below that for the same crop that is farmed more intensively. From a diversity point of view this translates to saying that less intensity is always preferred to higher intensity.

The relationship between area and intensity can be determined by taking the second cross-derivative of (1.11) w.r.t area, a_{ij}

$$\frac{\partial^2 \#\left(Q_{\{i,j\}}\right)}{\partial v_{ij}^{\text{int}} \partial a_{ij}} = z_i \frac{\partial \mathrm{c}_i\left(\mathbf{v}_{ij}\right)}{\partial v_{ij}^{\text{int}}} c_i a_{ij}^{z_i - 1} < 0 \text{ for } \mathrm{z}_\mathrm{i} > 0 . \qquad (1.12)$$

Thus increasing the intensity of farming reduces the rate of species production. In other words, intensive commodity production and species production are *substitutes*.

5.2.3 The species-density relationship

Animal density can impact biodiversity directly and indirectly. Species production on grazing land is by definition closely related to the intensity of grazing or livestock density per hectare grazing land: without grazing livestock grazing lands would not exist. On the other hand, intensive livestock farming can result in large stable manure stockpiles that are disposed of by spreading on arable land. The implications of spreading manure with increasing intensity follow from the species-intensity relationship.

The dynamics of species production on grazing land can be illustrated with help of the well established Gordon-Schaefer renewable resource model (mostly known from the fishery economics literature, but just as applicable to the exploitation of grazing lands). This model is applied to the case of a semi-natural grazing land in Figure 9. Note that this diagram illustrates the relationship between livestock density and the numbers of species per hectare.

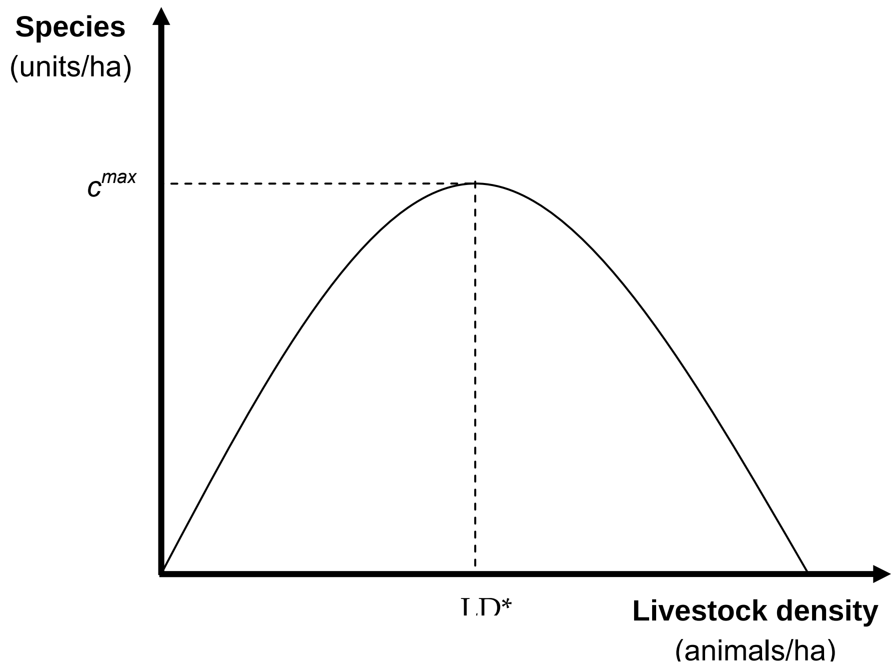


Figure 9. Grazing density and species production

Figure 9 shows that the number of species increases with grazing intensity up to a certain point (c^{max}) after which numbers begin to decline as overgrazing takes place (density > LD*) . This relationship is fairly intuitive. Agricultural land that is not grazed will return to the natural vegetation of the region (e.g., forest in Sweden) and species that require grassland habitat will be lost from the landscape. On the other hand, if grazing intensity exceeds a certain level, LD*, then overgrazing will degrade the grassland habitat and fewer species will survive. For example, in its extreme, overgrazing results in bare ground and soil erosion.

Assuming that the species-density curve is similar in form to the corresponding commodity-density curve (eg., meat output from grazing) and that property rights over grazing land are well defined, then grazing density will never exceed LD*. In fact, the economic optimal density for farmers' will be somewhat lower than LD* as there will be private opportunity costs associated with managing a grassland. Evaluating the function $c_i\left(v_{ij}^{den}\right)$ over the interval $v_{ij}^{\text{den}} = (0, \text{LD*})$ implies that $\partial \text{c}_i\left(\mathbf{v}_{ij}\right) \big/ \partial v_{ij}^{\text{den}} > 0$ and $\partial^2 \text{c}_i\left(\mathbf{v}_{ij}\right) \big/ \partial {v_{ij}^{\text{den}}}^2 < 0$, hence

$$\begin{aligned} \frac{\partial \#\left(Q_{\{i,j\}}\right)}{\partial v_{ij}^{\text{den}}} &= \frac{\partial \text{c}_i\left(\mathbf{v}_{ij}\right)}{\partial v_{ij}^{\text{den}}} c_i a_{ij}^{z_i} \quad > 0 \quad \text{for } v_{ij}^{\text{den}} \leq \text{LD}* \\ \frac{\partial^2 \#\left(Q_{\{i,j\}}\right)}{\partial v_{ij}^{\text{den}2}} &= \frac{\partial^2 \text{c}_i\left(\mathbf{v}_{ij}\right)}{\partial v_{ij}^{\text{den}2}} c_i a_{ij}^{z_i} \quad < 0 \end{aligned} \quad , \tag{1.13}$$

which implies species productivity increases with livestock density up until the optimal livestock density, LD*.

The cross derivative implies that grazing livestock and area are *complementary* in species production:

$$\frac{\partial^2 \#\left(Q_{\{i,j\}}\right)}{\partial v_{ij}^{\text{den}} \partial a_{ij}} = z_i \frac{\partial \text{c}_i\left(\mathbf{v}_{ij}\right)}{\partial v_{ij}^{\text{den}}} c_i a_{ij}^{z_i - 1} \quad > 0 \,. \tag{1.14}$$

This complementarity is no doubt the reason why semi-natural grazing land is very productive habitat relative to other agricultural land uses. Species production on grassland is therefore likely to be sensitive to changes in livestock density (livestock units per unit area). If the slope of the graph in Figure 9 is steeply negative up until $c_{lj}^{\max}$ then species production will be very sensitive to grazing intensity. Thus livestock density will be an important indicator of grassland quality for empirical analysis.

5.3 Expected value of species diversity in agricultural landscapes

Now that we have established the relationship between farmers' production decisions and species productivity, we can return to the question of valuing biodiversity in an agricultural landscape under uncertainty. This section uses the previous results and the definition of the conditional probability of species survival, $P\left(s \mid a_{ij}, \mathbf{v}_{ij}\right)$, to derive an expression for expected species value in an agricultural landscape.

To begin with, note that the area species relationship defined in Equation (1.10) is estimated by sampling from the landscape and hence any estimation of the equation will represent the average number of species found on an average size patch that is managed in a particular way. Since the average is just another word for the expected value, the estimated number of species, $\#\left(\tilde{Q}_{\{i,j\}}\right)$, is actually the expected value of species given area and management practices. Recall that the diversity value of a particular species is 1 if it exists and 0 if it doesn't. Hence the diversity value of a species present in $Q_{\{i,j\}}$ is 1 and the expected biodiversity value of a field (i, j) seen in isolation will be

$$E\left[\#\left(Q_{\{i,j\}}\right)\right] = \sum_{s \in S} P_{sij} = \#\left(\tilde{Q}_{\{i,j\}}\right) = \tilde{c}_i\left(\mathbf{v}_{ij}\right) a_{ij}^{\tilde{z}_i} \tag{1.15}$$

where $\tilde{c}_i$ and $\tilde{z}_i$ are the estimated parameter values.

Valuing species at the landscape level requires the elimination of joint survival probabilities as shown in Equation (1.5). The species-area relationship on the other hand is a measure of *all* species supported by a particular habitat. The number of unique species contributed by any (i, j) will though be some subset $q_{\{i,j\}}$ of all species, i.e., $q_{\{i,j\}} \subset Q_{\{i,j\}}$. Thus we might represent the number of unique species, $\#\left(q_{\{i,j\}}\right)$, as a proportion α_{ij}, of all species. Thus

$$\alpha_{ij} = \frac{\#\left(q_{\{i,j\}}\right)}{\#\left(Q_{\{i,j\}}\right)} = (0,1) \tag{1.16}$$

will be the proportion of total species supported by $\{i, j\}$ that are unique. It follows that we can write the production of unique species as

$$\#\left(q_{\{i,j\}}\right) = \alpha_{ij}\, c_i\left(\mathbf{v}_{ij}\right) a_{ij}^{z_i}\,. \tag{1.17}$$

Since α_{ij} is a constant, homothicity of the species-area relationship implies that the curve for unique species, Equation (1.17), will lie below the curve for all species (e.g., curve $c < c^0$ in Figure 7). Obviously it cannot lie above it because the number of unique species cannot exceed total species for any particular value of area. Thus applying the species-area curve to a subset of species also maintains the ranking property.

Result 1: For a subset of unique species the species-area relationship will maintain the ranking property.

Thus for empirical modeling it is only necessary to have a measure of c that values the uniqueness of the species supported by a particular land use, for the species-area relationship to be useful for evaluating diversity change. Assuming that the same measure of relative uniqueness is used to compare different land uses, then the species area relationship will be an approximation of the diversity value of a particular land use and can be used for ranking alternative land use patterns.

The expected value of species diversity at the landscape level will be approximately equal to

$$E\left[\mathrm{v}(Q)\right] \approx \sum_{i \in I} \sum_{j \in J} \alpha_{ij}\, \mathrm{c}_i\left(\mathbf{v}_{ij}\right) a_{ij}^{z_i} \;. \tag{1.18}$$

The important question left is whether the properties of this function are consistent with our intuition about species conservation in the agricultural landscape? Taking the first and second order partial derivatives yields

$$\begin{aligned} \frac{\partial E\left[\mathrm{v}(Q)\right]}{\partial a_{ij}} &> 0 \\ \frac{\partial^2 E\left[\mathrm{v}(Q)\right]}{\partial a_{ij}^2} &< 0 \end{aligned} \tag{1.19}$$

which are the characteristics we would expect of an expected value of diversity function.

The essence of Equation (1.18) is that if farmers change land use or management practices they indirectly alter the survival probability of unique species supported by the field and hence the expected diversity value of the landscape.

Finally, note that the species-area relationship can be put into practice in the following way. If we have an estimate of the expected number of unique species, $E\left[\#\left(q_{\{i,j\}}\right)\right]$, or an appropriate indicator, given a certain area of habitat i, then we can derive the parameter $\alpha_{ij}\,\mathrm{c}_i\left(\mathbf{v}_{ij}\right)$. This is important because we may not have direct estimations of the curve, but by the following transformation can deduce its value

$$\alpha_{ij}\,\mathrm{c}_i\left(\mathbf{v}_{ij}\right)=\frac{E\left[\#\left(q_{\{i,j\}}\right)\right]}{a_{ij}^{z_i}}. \tag{1.20}$$

5.4 Species-area relationship at the landscape level

Rosenzweig (1995, 381) argues that it doesn't matter whether we preserve a single large or several small (SLOSS) areas of habitat, the number of species we save will be the same (referred to as the SLOSS controversy in conservation biology). A compelling argument for this approach is that fields should not be considered as *islands* of habitat in an agri-landscape because of species interdependencies. The closer the patch work of fields the easier it will be for fields to exchange species and hence increase the expected value of species. In this case the species production of a patch work of fields will liken that of an equivalently sized large field. An island approach assumes no interdependencies between fields and the expected number of species in a landscape will be heavily dependent on the existence of a single large field, and the expected number of species in this case could never be larger than that for the largest field. This seems to be a strong assumption given the potential for species dispersal in a mosaic landscape. On the other hand, in a situation where the landscape becomes heavily degraded or fragmented, causing fields to become more and more isolated, the island approach to valuing diversity might be more appropriate.

Being able to use a single species-area curve for a particular type of habitat rather than the size of individual habitat patches, reduces data requirements for modeling diversity value considerably. Otherwise, we would need to know the conditional survival probabilities of the species supported by a patch, in order to reduce expected species value by common species, and, secondly, the level of interdependencies between patches. An aggregative approach yields results that are consistent with the theory of expected diversity value, and in the absence of reliable data, is a more robust approach than an individual patch approach and approaches used in comparable studies. In this case the value of diversity for a particular landscape will be calculated as

$$E \approx \sum_{i \in I} \hat{c}_i \left(\sum_{j \in J} a_{ij} \right)^{z_i} \tag{1.21}$$

where $\hat{c}_i$ is an indicator of the uniqueness of species associated with land use *i*.

Although the proposed indicator of biodiversity value, Equation (1.21), ignores interdependencies between different habitat types and species, it represents a significant step forward compared to other economic studies (Lankoski and Ollikainen, 2003; Miettinen et al., 2004; Lehtonen et al., 2005) that attempt to value changes in diversity. This is because our measure is based on a theory of diversity value, rather than simply counting species. The other studies though are of a pioneering nature and this paper should be seen as an extension rather than a criticism of their work. Nevertheless, a popular metric of choice for valuing diversity is the Shannon Index (Lankoski and Ollikainen, 2003; Miettinen et al., 2004). This index has little economic meaning when applied to agricultural land use because it does not consider differences in the diversity value of alternative land uses: all land uses are given equal weighting which an environmental policy maker is not likely to do (e.g., a relatively larger area of semi-natural grazing land than intensive grain is preferred in Sweden). Other studies have used a simple linear metric of diversity value (Lehtonen et al., 2005) but such a metric ignores tradeoffs between land uses and, hence, will overestimate the diversity impact of a reduction in the area of a relatively common land use and under estimate the impact of a relatively scarce use.

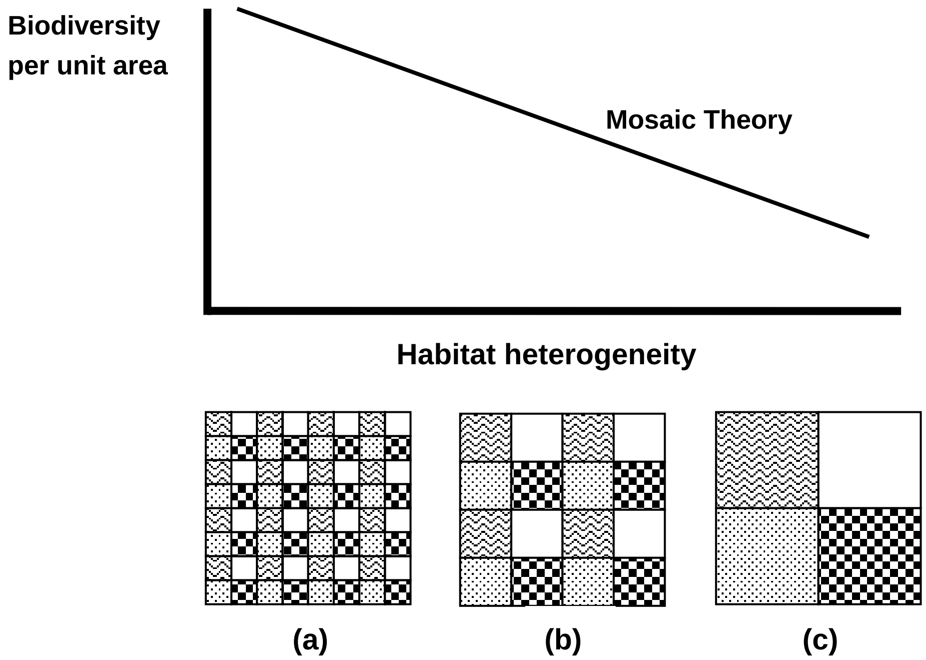
Let us now turn to the species-area relationship at the landscape level. The key to understanding why the aggregate species-area relationship should be a useful indicator for policy analysis is that it satisfies the requirements of a utility function (i.e., preference ordering), and hence will produce consistent rankings of the value-of-diversity of different landscape configurations (i.e., sets of land uses). Further, the aggregate measure also has qualitative characteristics consistent with our theoretical expected value of diversity function, Equation (1.2).

6 Measuring the value of landscape diversity and mosaic

If diversity is the number of different small things, then mosaic can be said to be the *arrangement* of the number of different small things (seen together that seem to form a pattern, i.e., a distinctive landscape). Diversity of land use on its own is too narrow a concept to capture the full value of an agricultural landscape. The spatial arrangement of patches and their physiographic features are also important for overall diversity and ecosystem function. Just as species abundances cannot convey the full complexity of the community, the number of different landscape elements or land uses, does not fully describe a landscape. This is because functional and visual interdependencies can exist between different elements of the landscape. For example, a particular bird species might rely on grassland for shelter and reproduction, and spilled seed on neighbouring cropland to survive the winter. The recreational and cultural value of the landscape is also influenced by the structure or arrangement of the landscape. A central concept for describing the arrangement of a landscape is *mosaic*.

Mosaic is a potentially useful concept for evaluating agri-landscapes because it is an indicator of habitat variability. Many organisms require different habitats for their development, feeding, over-wintering and reproduction, such as birds. They are more likely to find adequate habitat in a mosaic or heterogeneous landscape than in a uniform or homogeneous area. According to the mosaic concept or theory, biodiversity increases with the number and size variation of the mosaic patches in a landscape (Duelli, 1997). For example smaller but more mosaic patches, imply more boarder areas: panel (b) in Figure 10 has more and longer total borders than panel (c). "Soft edges" between different biotope types are known as ecotones, which often harbour a rich, specialized flora and fauna. The best-known examples are field edges, hedgerows and the shores of running and standing water. If a particular land use expands, then the length of borders declines and with it, the likelihood of retaining diversity. A mosaic landscape also has greater visual appeal to humans than a monotonous or homogeneous landscape. Mosaic seems therefore to be an important aggregate indicator of the environmental quality of an agri-landscape.

Figure 10. Biodiversity as a function of the number of habitat patches (with the same number of biotope types) and length of borderlines (ecotones).



Source: Duelli (1997, Fig. 6)

The question is, whether mosaic can be quantified or *uniquely characterized*? For example to measure the differences in "mosaic" between panels (a), (b) and (c) in Figure 10. This can be done by calculating relevant landscape metrics.

6.1 Using landscape metrics to quantify mosaic

Landscape metrics quantify the pattern of the landscape within a designated landscape boundary. There are literally hundreds of metrics described in the landscape ecology literature, and these are usually defined at either the patch, class or landscape level. They also fall into two general categories: those that quantify the *composition* of the map of a landscape, without reference to spatial attributes, and those that quantify the spatial *configuration* of the map, which requires explicit spatial information (e.g., the distance between two patches). Most metrics at the class and landscape levels are derived from patch level attributes. This has the implication that many of the metrics defined at different levels are correlated and therefore risk being redundant, if two or more are used in the same analysis. On the other hand, their interpretations may be somewhat different, so the appropriate choice of metrics to be calculated will rest on the specific questions posed by the investigator.

Patch-level metrics are defined for individual patches and characterise the spatial character and context of patches. For example, the size of individual patches and edge links can be important for various species. It might also be useful to know the nearest neighbour of each patch and the degree of contrast between the patch and its neighbour, as the probability of occupancy and persistence of an organism in a patch may be related to patch insularity.

Class-level metrics are aggregated over all the patches of a given type (class). These may be aggregated by simple averaging, or through some sort of weighted-averaging to bias the estimate to reflect the greater contribution of large patches to the overall index. Often, the primary interest is in the amount and distribution of a particular patch type, for example, in regard to habitat fragmentation. Habitat fragmentation is the landscape level process in which contiguous habitat is progressively subdivided into smaller, geometric more complex, and more isolated habitat fragments as a result of both natural processes and land use change (Forman, 1995). Fragmentation is recognised as a cause of declining biodiversity. Class indices separately quantify the amount and spatial configuration of each patch type, and thus provide a means to quantify the extent and fragmentation of each patch type in the landscape.

Landscape-level metrics are aggregated over all patch classes comprising the entire landscape. The major focus of landscape ecology is on quantifying the relationship between landscape pattern and ecological processes. Consequently, much emphasis has been placed on developing methods to quantify landscape pattern (Riitters et al., 1995; Gustafson, 1998).

Composition is easily quantified, and refers to features associated with the variety and abundance of patch types within the landscape, but without considering the spatial character, placement, or location of patches within the mosaic. Fundamental information about the composition of a landscape is provided by the statistical distribution of patch size at the class level. The size and number of patches comprising a class or the entire landscape mosaic is perhaps the most basic aspect of landscape pattern that can affect myriad biophysical processes.

Spatial configuration is much more difficult to quantify because measuring configuration usually requires information about the position of a patch in relation to every other patch in a landscape (i.e., the spatial coordinates of the patch obtained from a map of the landscape). Some aspects of configuration, however, are also summarised in the statistical distribution of patch size. Such metrics recognise that the function of a patch in a landscape is not only influenced by its surrounding neighbourhood (e.g., edge effects), but also that the magnitude of influence is affected by patch size and shape.

In conclusion, the distribution of patch size is fundamental for quantifying the composition and configuration (mosaic) of a landscape. Most landscape metrics either directly incorporate patch size information or are affected by patch size (Riitters et al., 1995). Consequently, the area of each patch comprising a landscape mosaic is perhaps the single most important and useful piece of information contained in the landscape. In relation to the agricultural landscape a patch is represented by individual fields.

6.1.1 Fundamental metrics: the statistical distribution of patch size

Since the statistical distribution of patch size is the most fundamental indicator of landscape mosaic I now show how relevant moments of this distribution can be calculated. The area of any particular class is calculated as

$$CA_i = \sum_{j=1}^{J} a_{ij} \quad \forall \ i = 1, \ldots, I \tag{1.22}$$

where a_{ij} is the area of patch j of class i, J is the set of all patches in the landscape and I the set of different classes represented in the landscape. In the context of IDEMA a_{ij} can be interpreted as field size (the area of field j used to grow crop i (where j functions as a unique reference to each field in the landscape).

The *proportional abundance* of class i is the proportion of each class relative to the entire map:

$$p_i = \frac{a_{ij}}{\sum_j a_{ij}} \tag{1.23}$$

where p_i is the share of the total land area covered by the i^{th} class (or land use). It follows that mean patch size, $\mu_{i,}$ when the number of patches of type i is NP_i $\left(\sum NP_i = I\right)$, is

$$\mu_i = \frac{CA_i}{NP_i} \tag{1.24}$$

and the variance of patch size in each class i is

$$\sigma_i^2 = \frac{1}{n-1}\sum_{j=1}^{J}\left(a_{ij} - \mu_i\right)^2 \tag{1.25}$$

Since the variance is an indicator of absolute variation it usually needs to be interpreted in conjunction with mean patch size. A more useful indicator of variation might therefore be the coefficient of variation because it considers the mean and variance of patch size simultaneously:

$$CV_i = \frac{\sigma_i}{\mu_i} \tag{1.26}$$

Finally, the range of patch size from the smallest to the largest is

$$RANGE_i = \left\{\min_i\left(y_{ij}\right), \max_i\left(y_{ij}\right)\right\} \tag{1.27}$$

An important aspect of many landscapes is the existence of a matrix element. The matrix is the most extensive and most connected landscape element type, and therefore plays a dominant role in the functioning of the landscape (Forman and Gordon, 1986). In most landscapes, the matrix is obvious to the observer, as in inland Sweden where it is pine forest. Generally, the matrix element should not be included as another "patch" type in any metrics that would be dominated by the matrix element (e.g., average patch size). In this case, the metric would characterise the matrix, rather than patches within the landscape.

6.2 Quantifying landscape diversity

Landscape diversity is an important characteristic because diversity is valuable per se. It also seems that landscape "diversity" could be used as a proxy for the value of biodiversity, cultural heritage, recreational and knowledge linked to a landscape. Though diversity is usually associated with biodiversity, Weitzman points out diversity theory is fairly general, so the "species" of analysis can be adjusted to reflect the set of "things" of interest. Before introducing a measure of landscape diversity let me first explain why diversity has inherent value.

6.2.1 Evolutionary library model of landscape diversity

A useful metaphor for conceptualizing landscape diversity is a library, where the books in a library represent the store of knowledge (diversity value) that is valued or has potential value to human beings. In the case of a set of libraries the *aesthetic value* of any particular library is reflected in the number of different books in each library, whereas the information content of diversity can be thought of as the knowledge contained in the collection of books (e.g., mathematical proofs, cures for various diseases, engineering solutions, etc.). Weitzman (1998) shows in what he refers to as the "Noah's Ark Problem" that when they are appropriately modeled "diversity as aesthetic value" and "diversity as information content" are identical concepts (i.e., the number of different things and their inherent knowledge).

This is a useful insight because maximizing the aesthetic value of diversity (i.e., the number of different things) will also maximize the information content of these different things, and hence the total value of diversity. The direct implication of this result is that: more diversity must be better than less, which gives rise to Corollary 2.

> Corollary *1: The expected diversity of a set of land uses is "essentially" the same concept as the information content or diversity value of the same set of land uses.*

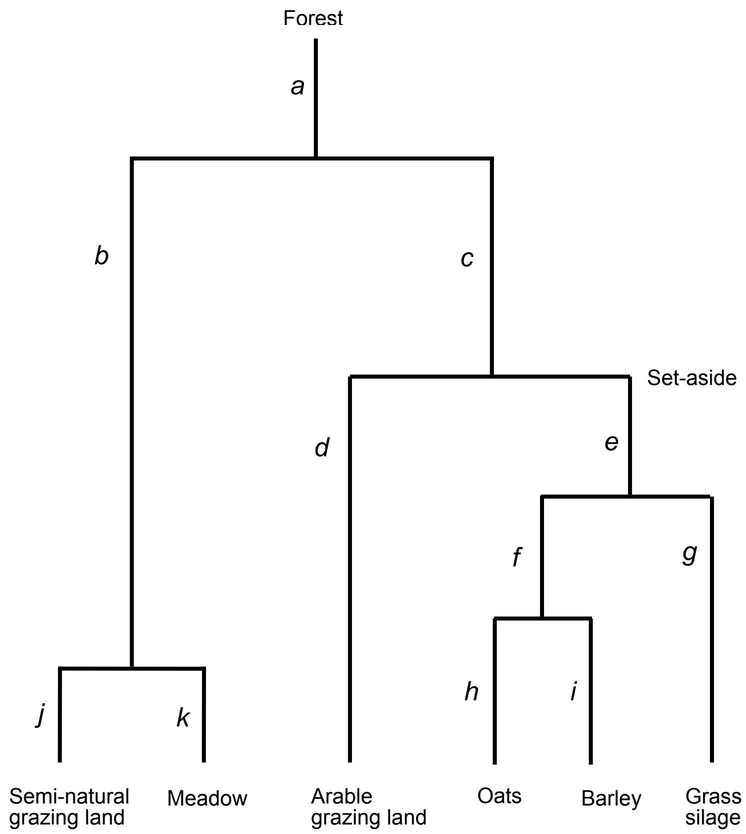
A measure of aesthetic value, or diversity *per se* (i.e., the number of different books\land-uses), could therefore be used as a proxy for the overall value of diversity. If a particular agricultural land use is thought off as a particular book in a library, then maximizing the number of different land uses will also maximize the diversity value (i.e., biological and cultural information content) of the agricultural landscape, just as maximising the number of different books maximises the knowledge content of a library. Alternatively, a reduction in the number of different land-uses/books would result in a loss of diversity/knowledge, and hence a cost to society.

Recall from Section 5.1 that the diversity of a set is the maximum, over all members of the set, of the distance of that member from its closest relative in the set plus the diversity of the set without that member. A measure of difference d(i,j) between two members of a set is therefore critical for evaluating diversity: in order to rank policy outcomes we need to determine some measure of uniqueness or distinctiveness of a land-use (What measure is used in practice, will depend on the goals of the analysis, the data available and what is feasible to do.) At this stage let us assume that such a measure is available. To conceptualize the diversity value of an agricultural landscape I depict in a hypothetical evolutionary tree model of an agricultural landscape.

Figure 11 is based on Swedish history where most agricultural land has at some stage been converted from forest (i.e., the common ancestor is assumed to be forest). An evolutionary tree could conceivably be derived for any agricultural landscape in Europe. The primeval ancestor of all agricultural land uses in this example is forest, to which all land will return by natural regeneration if agricultural production or other land management ceases (which is situation in Sweden). The vertical distance between each node (i.e., branch length) is interpreted as an exact measure of the diversity value of each land-use. Total diversity value of the hypothetical landscape depicted in Figure 11 is therefore: $V(S) = a + b + \ldots + k$.

If a land use is abandoned (i.e., becomes extinct), the loss of diversity equals the land use's distance from its closest relative the vertical distance to its first common ancestor. Thus, if oat production were abandoned, the loss of diversity would be its distance to barley, *d(oats,barley),* which is *h*, since barley is its closest relative on the evolutionary tree. This myopic formula can be repeated over any abandonment pattern, because any sub-evolutionary tree of an evolutionary tree is also an evolutionary tree. When a land-use is abandoned, the loss of diversity is calculated as if its evolutionary branch were snapped off the tree and discarded (This implies of course that land abandonment is an irreversible effect. Such an assumption is though sufficiently realistic for our purposes, because continuity of management is critical for preservation of landscape values.) By converting all arable land to grass silage, we would lose the diversity value inherent in the production of oats and barley ($f + h + i$), but not that common to the production of grass silage, oats and barley ($c + e$). Similarly, if we lost either meadows or semi-natural grazing land, but not both, we would lose a relatively small amount of diversity (either branch j or k). On the other hand, if we lost both semi-natural grazing land and meadows, we would lose a very large amount of diversity ($b + j + k$). This potential loss is the diversity value of semi-natural grazing land and meadows.

Figure 11. Agricultural landscape diversity: evolutionary tree representation



Source. The author's hypothetical example.

This is the theoretically correct way of measuring landscape diversity.

6.2.2 Shannon's diversity index

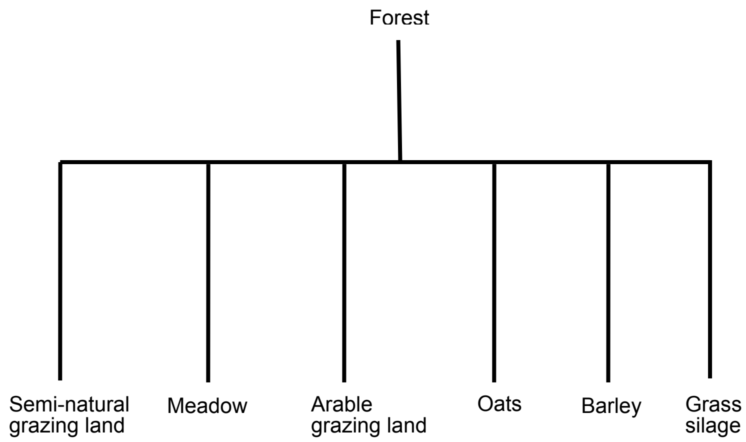
In practice, the diversity of land use is usually expressed in terms of *richness* and *evenness* (Olson and Francis, 1995). Richness refers to the number of different land uses (classes) and evenness to the uniformity of distribution of the area of different land-cover classes. Shannon's diversity index is a proportional abundance index and reflects both evenness and richness of a set:

$$H = -\sum_{i=1}^{I} p_i \ln(p_i) = -\sum_{i=1}^{I} \frac{a_{ij}}{\sum_j a_{ij}} \ln\left(\frac{a_{ij}}{\sum_j a_{ij}}\right), \tag{11}$$

where H denotes diversity, n is the number of different land uses, $i \in n$, y_i is the area of land use i, and p_i is the share of the total land area covered by the i^{th} land use. Using an appropriate normalization, H approaches 0 when there is only a single land use and 1 when the distribution of land uses is uniform.

An important limitation of Shannon's index is that it doesn't measure total diversity value. Remember that an ideal measure of diversity would measure the length of all the branches of the evolutionary tree. Shannon's index on the other hand, assumes that diversity value and hence the length of each branch on the evolutionary tree, is identical for all land uses. If this were the case, then the evolutionary tree of the hypothetical landscape pictured in Figure 10 would look like the one in Figure 12. This of course is unlikely in reality. The Shannon index is a measure of structural diversity, which is only a partial measure of total diversity. It implicitly assumes that all land uses are "genetically" equidistant and there is no prior preference for preserving a particular land use.

Figure 12.. Shannon's diversity index: evolutionary tree representation



Source. The author's example.

For policy analysis a measure of relative change is probably desirable, since decision-makers are usually interested in deviations from a baseline or benchmark (i.e., the status quo). If $H_{baseline}$ is defined as diversity of the landscape before the "decoupling" reform and H_{reform} after a reform scenario, then the relative change in structural landscape diversity, ΔH is:

$$\Delta H = \frac{H_{reform} - H_{baseline}}{H_{baseline}}, \tag{12}$$

where $\Delta H = 0$ implies no change in structural diversity, $\Delta H > 0$ *implies increased* structural diversity and $\Delta H < 0$ implies reduced structural diversity.

In IDEMA, it would be valuable to be able to calculate Shannon's index at both the individual farm and regional level. Calculations at the farm level would provide an indication of local diversity, whereas calculations at the regional level would measure total diversity. Comparing the distributions of farm level diversity before and after policy reform could then be analysed for various types of farms to measure changes in between farm diversity.

Note that Shannon's index does not measure mosaic because it does not consider the pattern of land use, but simply implies that the proportion of each land use is identical if $H=1$, and perfectly homogeneous if $H=0$. For examples, Shannon's index for each of the mosaics pictured in Figure 10 would be identical. Mosaic on the other hand is a measure of landscape complexity. A reasonable measure of total diversity needs to consider both structural diversity and mosaic.

7 Complementary environmental indicators

In addition to landscape metrics a number of complementary environmental indicators will be used to enrich the analysis, e.g., nutrient balances, livestock densities and chemical application. These indicators are fairly standard in the literature so only nutrient balances relevant to IDEMA are discussed . The indicators are only intended to provide some information on the impact of decoupling on negative environmental impacts such as nutrient pollution. However they are also important complementary indicators to landscape values. For example production intensity is relevant to both biodiversity value and nutrient pollution.

7.1 Nutrient balances

A nutrient balance is the difference between the amount of nutrients entering and leaving an agricultural system. The system boundary can be set at any relevant level; field, stable, farm, region, watershed or nation. The choice of system boundary should though, be appropriate for the study at hand. For example, the balance for an individual field might not be representative of the balance at the farm level, and that of any particular farm of a region. It is also important to realise that a nutrient balance only measures the difference between nutrients entering a farm via inputs and leaving it via outputs. What a balance doesn't do, is describe what happens with a surplus (or deficit). That is to say, as there is no general relationship between a nutrient surplus and environmental damage (Section 2.1), a nutrient balance index must always be interpreted with caution. At best, a balance should be interpreted as an indicator of environmental risk, rather than of damage a higher surplus poses greater risk of environmental damage than a lower surplus.

The objective of environmental modelling in IDEMA is to evaluate the environmental effects of decoupling at the farm and regional levels. For that reason, the appropriate system boundaries for calculating balances are the farm and the agricultural region under study. Farm balances are usually calculated at the "farm gate", which implies that a balance measures the flow of nutrients to a farm via inputs (e.g., commercial fertilisers, stockfeed and stable manure) plus nitrogen fixation through cultivation of legumes, and flows from the farm in products (i.e., outflows of cash crops, milk and livestock). Calculating balances at the farm and regional levels is also consistent with the structure of AgriPoliS, which is based on farm level data.

Note that, for policy analysis we are really only interested in relative changes from a pre-reform baseline e.g., conditions in 2002 and not absolute balances. Factors that are not expected to change due to policy reform can therefore be disregarded. For example, atmospheric deposition is not likely to be affected by MTR but rather is expected to remain constant, so it can be ignored in our study. Flows of purchased inputs and outputs though are expected to change, so these will form the basis of the nutrient balance calculations in IDEMA. The nutrient balance for each farm $f \in F$ will be calculated as:

$$NB_f = fertilizer + manure + fixation - harvest - livestock - milk \tag{1.28}$$

and the balance for a region is simply the aggregate of net balances of the farms comprising a region:

$$NB_{region} = \sum_{i=1}^{n} NB_i \ . \tag{1.29}$$

A standard way of presenting a nutrient balance is in terms of a per hectare average. This index is obtained by simply dividing the farm or regional level, nutrient balance by the relevant area of agricultural land for the farm or region. Another interesting index might be the variation in balances across farms, which can be represented as the range, standard deviation and coefficient of variation of farm balances.

8 Analysis: impact of decoupling on landscape values

8.1 Economic model of farmers' land use decisions

In this section I develop a simplified economic model to illustrate the potential impact of decoupling on farmers' land use decisions, variable input use, commodity output and, ultimately, the landscape. To begin with let us assume that the objective of individual farmers is to maximize net family income given the family's wealth and labour endowment. The area of agricultural land in their region is a limited resource and the opportunities for employment in other sectors determine the opportunity cost of on-farm labour. The family's wealth endowment which might be thought of as a portfolio of assets determines the family's cost of capital. Private profit maximization will also maximize total profit in the region if an efficient land market exists, which is assumed to be the case.

EU farmers are of course heavily subsidized and prior to the MTR-reform subsidies were coupled to commodity output. However only certain commodities were *eligible* for subsidies. A coupled payment or production subsidy can be introduced by adding a per unit subsidy, $s_i > 0$, to the market price of eligible commodities. If the set of eligible commodities is $E \subset I$, then the relevant output price for farmers is modeled as $p_i = p_\mathrm{i}^\mathrm{m} + s_i$ where p_i^m is the market price of *i* and $s_i = 0$ for non-eligible crops, i.e, for $i \notin E$. One can quickly see that coupled payments impact the relative price of alternative land uses such that the relative price of eligible crops is increased. Hence farmers were provided with a strong incentive to grow eligible crops, and as a result distorted the market for agricultural commodities. In some regions the coupled payment has been sufficient to induce farmers to grow crops that otherwise would not be profitable.

Suppose the agricultural landscape of interest comprises a set of *F* family farms where individual farms are denoted as $f \in F$. Maximal regional income, *π*, is achieved by farmers choosing levels of A_{fij}, V_{fiv} and K_{fi} that solve the follow optimization problem (note that $A_{fij} \equiv a_{ij}$ and $V_{fiv} \equiv \mathbf{v}_{ij}$ from the previous diversity analysis, but are capitalized to indicate that they are now choice variables and not landscape parameters):

$$\pi = \max_{A,V,K} \sum_{f \in F} \left[\sum_{i \in I} \sum_{j \in J} A_{fij} \left\{ p_i \, \mathrm{g}\left(V_{fiv}, K_{fi}\right) - \sum_{v \in V} c_v V_{fiv} - \mathrm{k}\left(\bar{k}_f\right) K_{fi} \right\} \right] \tag{1.30}$$

s.t.

$$\sum_{i \in I} \sum_{j \in J} A_{fij} \, \mathrm{l}\left(A_{fij}, K_{fi}\right) + \delta_f \sum_{j \in J} \bar{\mathrm{d}}_{fj} \leq \bar{\mathrm{l}}\left(\bar{l}_f, \bar{\mathrm{w}}_f\right) \quad \forall f \;\text{:Labour}\; \lambda_f^l \tag{1.31}$$

$$-\sum_{j \in J} A_{fij} \leq -\sum_{k \in K \subset I} \sum_{j \in J} \alpha_{ik} A_{fkj} \quad \forall f, i \;\text{: Rotation}\; \lambda_{fi}^\mathrm{r} \tag{1.32}$$

$$\sum_{f\in F}\sum_{i\in I} A_{fij} \le \bar{\mathrm{a}}_j \quad \forall j \;\text{ : Field area } \lambda_j^{\mathrm{a}} \tag{1.33}$$

$$A_{fij}, K_{fi}, V_{fiv}, \lambda_f^{\mathrm{l}}, \lambda_{fi}^{\mathrm{r}}, \lambda_j^{\mathrm{a}} \ge 0$$

where

A_{fij} is the area of crop *i* grown on block *j* by farm *f* , which will be referred to as field $\left(f, i, j\right)$ in the set of all potential fields $\left\{f, i, j\right\}$.

V_{fiv} is the amount of variable input *v* used by farm *f* to produce one ha of commodity *i* (homogeneous land quality implies identical input use for identical crops).

K_{fi} is capital used to produce one ha of *i* (e.g., machinery hrs/ha).

p_i is the unit price of commodity *i* including any price support $s_i^{\mathrm{c}} > 0$ for eligible crops.

c_v is the unit cost of variable input *v.*

g_{fi} is the farm specific production function for commodity *i* which is assumed to be increasing in variable inputs and capital such that $\mathrm{g}'_V > 0$ and $\mathrm{g}'_K > 0$, but increasing at a decreasing rate, $\mathrm{g}''_{VV} < 0$ and $\mathrm{g}''_{KK} < 0$. For example nutrient availability is crucial for crop growth and larger machinery capacity implies crops can be sown and harvested closer to the optimal date (Ekman, 2002).

$\mathrm{k}\left(\bar{k}_f\right)$ is the unit cost of capital for farm *f* given the farms wealth endowment $\bar{k}_f$. Greater wealth implies a lower cost of capital, thus $\mathrm{k}' < 0$ and $\mathrm{k}'' = 0$ (e.g., banks provide lower interest rate loans to low risk customers).

There are also physical and technological constraints on the farmers' planning problem. Equation (1.31) ensures that labour input does not exceed the availability of family labour $\bar{\mathrm{l}}\left(\bar{l}_f, \bar{\mathrm{w}}_f\right)$, which is assumed to be a function of the family's labour endowment $\bar{l}_f$ and opportunity cost of labour $\bar{\mathrm{w}}_f$ (i.e., their ability to generate off-farm income). The function $\mathrm{l}\left(A_{fij}, K_{fi}\right)$ determines the amount of labour needed to produce one hectare of commodity *i* given field size A_{fij} and capital input K_{fi}. The labour need is assumed to be declining in field size and machinery capacity, i.e., $\mathrm{l}'_A < 0$ and $\mathrm{l}'_K < 0$. That is, labour savings can be obtained by expanding field size and/or machinery capacity. Further, I assume that $\mathrm{l}''_A \leq 0$, $\mathrm{l}''_K \leq 0$ and $\mathrm{l}''_{AK} \geq 0$ which implies that scale efficiencies are either declining or linear, and that increasing area and capital complement each other (which introduces economies of size).

Finally, $\overline{\mathrm{d}}_{fj}$ is the average distance from farmstead *f* to block *j*. Distance is a very important parameter in reality because it hampers farm expansion: the further it is to a field, the more time that will be spent transporting machinery and other inputs to it, and the less economic it will be to farm. This cost is represented by the fixed labour cost term $\delta_f \sum \overline{\mathrm{d}}_{fj}$, where δ_f is the farm specific unit labour cost of distance (which can vary due to say the quality of access roads near the farm). This cost can be thought of as an intercept term that moves the labour function up or down depending on field distance and farm specific characteristics. If this constraint is binding then the family's shadow wage will be $\lambda^{\mathrm{l}}_f > 0 = \overline{\mathrm{w}}_f$: the opportunity cost of using labour on the farm should equal the unit wage for off-farm labour (if the family is to maximize its total income).

Equation (1.32) is a simple crop rotation constraint that states that some proportion of the total area of a subset of crops $k \in K \subset I$ grown by farm *f*, cannot exceed the area of crop *i*, where α_{ik} denotes the minimum proportion of crop *i* that can be grown in combination with all *k*. In practice there may exist a range of rotation "constraints" such as animal feed requirements, risk diversification benefits, yield enhancing effects, etc. Including all of these would only complicate the model unnecessarily. The point is, farmers are not likely to grow one single crop but a mixture of crops, and this simple constraint is sufficient to capture the essence of rotating crops. If this constraint is binding then it will give rise to a shadow price $\lambda^{\mathrm{r}}_{fi} > 0$ which reflects the implicit benefit of growing a mix of crops.

Equation (1.33) ensures that the area of land farmed on each block *j*, does not exceed the physical size of the block, $\overline{\mathrm{a}}_j$ ha. A block may be subdivided into any number of fields (f, i, j) and managed by any number of different farms, but the total field area cannot exceed block area. Thus a block could be farmed with different crops to obtain rotation benefits or because it is managed by different farms. If block size constrains production then it will generate a land rent that is equal to $\lambda_j^{\mathrm{a}} > 0$ (the shadow price of block *j*). On the other hand if block size does not constrain production then $\lambda_j^{\mathrm{a}} = 0$, reflecting a zero land rent.

Summarizing, a region is assumed to comprise a set of blocks of agricultural land $\left\{\overline{\mathrm{a}}_j\right\}$ and a set of family farms *F*. Farmers' are assumed to make their land use, $\left\{A_{fij}\right\}$, variable input, $\left\{V_{fiv}\right\}$ and capital investment, $\left\{K_{fi}\right\}$, decisions to maximize farm income. Since farms that can pay the highest implicit land rent, $\left\{\lambda_j^{\mathrm{a}}\right\}$, will get to farm a particular unit of land, regional profit will be maximized simultaneously. Farms are differentiated in the model by their labour and wealth endowments, $\left\{\overline{l}_f, \overline{\mathrm{k}}_f\right\}$, off-farm income potential, $\left\{\overline{\mathrm{w}}_f\right\}$, distance to blocks of agricultural land $\left\{\overline{\mathrm{d}}_{fj}\right\}$ and their farming ability that is reflected in the set of farm specific crop production functions $\left\{\mathrm{g}_{fi}\right\}$. Labour and capital are substitutes in the model whilst capital and field size, are complements. This implies that families with a relatively low opportunity cost of labour will utilize relatively more labour in the farm enterprise and that economies can be obtained by farm expansion. Finally concavity of the crop production function and existence of rotation "constraints" implies that it will never be optimal to grow just a single crop, as is usually the case in practice.

8.2 How farmers solve their planning problem

In this section I solve the farmers' planning problem, which yields the profit maximizing configuration of the landscape given the economic parameters and technical constraints of the model. Since g is concave in all its variables and the other functions are linear, the objective function will be concave. Note also that l is convex in its variables and the other constraints are linear. These conditions ensure that the first order conditions for profit maximization also guarantee a global solution because the problem is convex. The Lagrange function for this problem is

$$
\begin{aligned}
L(\mathbf{A},\mathbf{V},\mathbf{K}) &= \sum_{f\in F}\left[\sum_{i\in I}\sum_{j\in J} A_{fij}\left\{p_i\,\mathrm{g}\left(V_{fiv},K_{fi}\right)-\sum_{v\in V} c_v V_{fiv} - \mathrm{k}\left(\bar{k}_f\right)K_{fi}\right\}\right] \\
&+\sum_{f\in F}\left[\lambda_f^l\left(\bar{\mathrm{l}}\left(\bar{\mathrm{w}}_f\right)-\sum_{i\in I}\sum_{j\in J}A_{fij}\,\mathrm{l}\left(A_{fij},K_{fi}\right)+\sum_{f\in F}\delta_f\bar{\mathrm{d}}_{fj}\right)+\sum_{i\in I}\lambda_{fi}^{\mathrm{r}}\left(-\sum_{k\in K\subset I}\sum_{j\in J}\alpha_{ik}A_{fkj}-\sum_{j\in J}A_{fij}\right)\right] \\
&+\sum_{j\in J}\lambda_j^{\mathrm{a}}\left(\bar{\mathrm{a}}_j-\sum_{f\in F}\sum_{i\in I}A_{fij}\right)
\end{aligned}
\tag{1.34}
$$

with $\left\{\lambda_f^{\mathrm{l}},\lambda_{fi}^{\mathrm{r}},\lambda_j^{\mathrm{a}}\right\}$ being the Lagrange-multipliers associated with the labour, rotation and block area constraints respectively. The solution is defined by the Kuhn-Tucker conditions, which are

$$
\begin{aligned}
\frac{\partial L}{\partial A_{fij}^*} = 0 \Rightarrow p_i\,\mathrm{g}\left(V_{fiv},K_{fi}\right)+\lambda_{fi}^{\mathrm{r}} &= \sum_{v\in V} c_v V_{fiv} + \mathrm{k}\left(\bar{k}_f\right)K_{fi} \\
&+\lambda_f^l\left(\mathrm{l}\left(A_{fij},K_{fi}\right)+\frac{\partial \mathrm{l}\left(A_{fij},K_{fi}\right)}{\partial A_{fij}}\right)+\lambda_j^{\mathrm{a}},\quad \forall fij
\end{aligned}
\tag{1.35}
$$

$$
\frac{\partial L}{\partial V_{fiv}^*} = 0 \Rightarrow p_i\frac{\partial\,\mathrm{g}\left(V_{fiv},K_{fi}\right)}{\partial V_{fiv}} = c_v,\quad \forall fiv \tag{1.36}
$$

$$
\frac{\partial L}{\partial K_{fi}^*} : p_i\frac{\partial\,\mathrm{g}\left(V_{fiv},K_{fi}\right)}{\partial K_f}+\lambda_f^l\left(-\frac{\partial\,\mathrm{l}\left(A_{fij},K_{fi}\right)}{\partial K_{fi}}\right) = \mathrm{k}\left(\bar{k}_f\right),\quad \forall fi. \tag{1.37}
$$

$$
\left(\lambda_f^{\mathrm{l}},\lambda_{fi}^{\mathrm{r}},\lambda_j^{\mathrm{a}} > 0 \quad \text{if relevant constraint binding, otherwise } = 0.\right)
$$

Denote the solution to this problem by the set of vectors of optimal choice variables $\left\{\mathbf{A}^*,\mathbf{V}^*,\mathbf{K}^*,\boldsymbol{\lambda}^*\right\}$. This solution set can be used to evaluate agriculture's impact on the environment because as was shown in Section (5.2) the impact is determined by farmers' land allocation choices $\mathbf{A}^*$ and management choices $\mathbf{V}^*$. For example, plugging the set of values $\left\{\mathbf{A}^*, \mathbf{V}^*\right\}$ into Equation (1.18) yields the expected value of species diversity.

Together Equations (1.35)-(1.37) provide us with a method to evaluate the potential impacts of decoupling on environmental quality. That is the decision rules implied by these equations can be used to determine how changes in the exogenous variables of the model (i.e., prices, distances, family wealth, the off-farm wage, etc.) and ultimately decoupling, will influence farmers' choices and hence environmental quality.

Starting with the land allocation decision, Equation (1.35) states that the marginal revenues from farming an additional hectare of crop *i* (the LHS) must equal the associated costs of variable inputs and capital plus the implicit costs of family labour and land (the RHS). The third term on the LHS is the implicit marginal cost of labour and shows that the larger the field, the lower the unit labour need per ha *i* since $\partial \mathrm{l}/\partial A < 0$.

A land market is implied by the model because the set of shadow prices of land, $\left\{\lambda_j^{\mathrm{a}}\right\}$, internalizes "land rents" and the farm that can extract the highest rent or profit from a particular plot of land will be allocated the land by the optimization program. Land rental cost is in other words an imputed cost. In this way the solution to the problem guarantees that regional profits are maximized given the limited area of agricultural land, i.e., $\sum \overline{\mathrm{a}}_j$, and private optimizing behaviour. If some land comprising a particular block is not used in production then $\lambda_j^* = 0$. Thus $\lambda_j^* > 0$ implies that the entire block is used in production, and the higher the shadow price the greater the profitability of that particular block. From a policy perspective the higher the rent for a particular block the less sensitive it will be too changes in market and policy parameters. The shadow price of land is therefore an important policy variable if policymakers are concerned about land use change or land abandonment (which we know they are).

Note that by rearranging Equation (1.35) the shadow price of land can be expressed as an implicit function of prices, the cost of capital, the off-farm wage and field distances, i.e., $\lambda_j^{\mathrm{a}} = \lambda\left(p_i, c_{\mathrm{v}}, \overline{k}_f, \overline{\mathrm{w}}_{\mathrm{f}}, \overline{\mathrm{d}}_{fj}\right)$. Differentiating this function *w.r.t.* its parameters indicates the sensitivity of land rents to changes in the parameters. Increasing commodity prices (e.g, due to a production subsidy) will increase the shadow price of land and hence the likelihood of land being farmed $\left(\partial\lambda_j^{\mathrm{a}} \big/ \partial pi > 0\right)$ The increase in land rent results also, in practice, in higher land prices since rents are usually capitalized into prices (a well known fact). Rising input costs will on the other hand reduce rents $\left(\partial\lambda_j^{\mathrm{a}} \big/ \partial c_v < 0\right)$. Increasing family wealth reduces the cost of capital and increases potential land rent $\left(\partial\lambda_j^{\mathrm{a}} \big/ \partial\overline{\mathrm{k}}_{fi} > 0\right)$. Conversely an increasing off-farm wage increases the required returns from family-labour and hence reduces the implicit land rent $\left(\partial\lambda_j^{\mathrm{a}} \big/ \partial\overline{\mathrm{w}}_f < 0\right)$. Finally, increasing distance to fields reduces land rent$\left(\partial\lambda_j^{\mathrm{a}} \big/ \partial\overline{\mathrm{d}}_{fj} > 0\right)$. The valuation of land and hence the configuration of the landscape after decoupling might therefore be highly dependent on farm specific characteristics

For cases where $\lambda_j^* > 0$ a marginal change in prices/support or distances will not affect the area of the block in production, only the implicit rental price—that is increasing in prices and decreasing in distances—will change. On the other hand if the rental price is zero then a marginal reduction in prices or increased distance from a farmstead (e.g., the nearest farm stops producing) will result in a reduction in the area of farmed land ceteris paribus.

Condition (1.36) implies that farmers will apply variable inputs until the marginal value product of each input is equal to its price (a standard result). Increases in input prices reduce the profitability of farming and hence the likelihood that marginal land is farmed. Increasing prices have the opposite effect: more marginal land is moved into production.

Condition (1.37) states that capital investment should occur until the marginal value product of capital is equal to the marginal cost. In the case of crop production the marginal product of cropping activities increases as machinery capacity is increased because crops can be sown and harvested closer to the optimal date, i.e., $\partial^2 L \big/ \partial K_{fi}^* \partial A_{fij} > 0$. This result implies a complementarity between block size and capital investments. The opportunity to farm large fields would allow this complementarity to be exploited, and implies that average profits per hectare will be higher for a large field than a small field, ceteris paribus. Conversely, small fields would not allow farmers to take advantage of complementarities and farming them will be less profitable, which would reveal itself as a lower implicit land rent for small blocks. Small fields will therefore run greater risk of being idled or abandoned than large fields as returns to commodity production decline.

8.3 Incorporating decoupled payments into the model

The aim of this section is to determine in theory, how a switch from production (or coupled) subsidies to *decoupled* farm payments will impact landscape values (this in order to guide the proposed empirical environmental evaluation). Decoupled payments as implemented by the EU come in two forms; i) A lump sum income transfer to farmers, known as the *single-farm payment* (SFP) and ii) an area based "regional" payment that is contingent on a *cross-compliance* condition that requires a minimal form of agricultural land management in order to obtain the payment. In practice a mixture of the decoupled regimes and coupled payments is being implemented in many countries, but this will only result in outcomes lying between the three extremes and "buffering" of the full impacts of decoupling. Thus in the ensuing theoretical analysis it is only necessary to examine the extreme cases in order to draw conclusions about any combination of policies. I intend therefore to compare i) fully coupled payments with a fully decoupled payment and ii) a fully decoupled with a land use cross-compliance condition. I begin this section by showing how decoupled payments can be introduced into the model, followed by an analysis of the impact of decoupled payments.

A single-farm payment can be modeled by simply adding the payment, denoted $\mathrm{SFP}_f \geq 0$ for farm *f*, to the objective function, Equation (1.30). An interesting aspect of the SFP is that it is expected to impact the opportunity cost of on farm investment, especially in situations where farms have had credit restrictions such as in the new member states (Andersson, 2005). To model this impact we can also include the payment in the family's wealth vector, i.e., $\bar{k}_f + \mathrm{SFP}_f$, such that $\mathrm{k}\left(\bar{k}_f + \mathrm{SFP}_f\right)$. Since the SFP increases family wealth the implications follow from the analysis above. That is the SFP reduces the opportunity cost of capital. This is consistent with the view that it is impossible to implement a fully decoupled payment, as such a payment might compensate losses in production support by reducing the costs of producing commodities. Just how big the compensating effect will be remains to be seen.

An area based regional payment can be modeled in a similar way to a production subsidy, only now the set of eligible production activities includes *all* agricultural land uses and the payment is identical for all *i* (which I assume to include a minimal land management activity, $A^{\mathrm{c}}_{fj} \in \left\{ A_{fij} \right\}$, that satisfies the cross-compliance condition, such as grass coverage and maintaining drainage systems). The relevant commodity price for farmers entitled to decoupled area payments will be $p_i - s^{\mathrm{c}}_i = p^{\mathrm{m}}_i$ (recall p_i was assumed to include price support s^{c}_i). Hence the relative price of crop *i* that was previously eligible for support $\left(s^{\mathrm{c}}_i > 0 \right)$ compared to $k \notin E$ $\left(s^{\mathrm{c}}_i = 0 \right)$ after decoupling will fall, i.e.,

$$\frac{p_i - s^{\mathrm{c}}_i}{p_k - s^{\mathrm{c}}_k} = \frac{p^{\mathrm{m}}_i}{p_k} < \frac{p_i}{p_k}, \quad \forall i \neq k\,, \tag{1.38}$$

or, conversely, the relative profitability of growing crops that were previously *non-eligible* will increase. As it is relative prices that drive economic decisions, the decoupled regional payment should not distort commodity markets because the relevant prices for output decisions will be based on market prices, as shown in Equation (1.38). In the case where no commodity production is profitable farmers will maximize returns by managing land according to the cross compliance condition if that is profitable (otherwise abandoning it).

In some cases there might be additional cross-compliance conditions such as for grazing land in Sweden which entails a minimum animal density requirement or nutrient application limitations to meet water quality goals. Such constraints can be modeled generally as

$$\mathrm{h}\left(A_{fij}, v_{fiv} \right) \leq (\geq) \overline{\mathrm{H}} \tag{1.39}$$

where $\overline{H}$ is the maximum (minimum) value of the cross-compliance variable. If the constraint is binding, then farmers will generate lower profits than if they were able to freely choose activity levels. The marginal profit impact will be equal to the shadow price of the constraint, denoted $\lambda^c_f \geq 0$.

8.4 Farmers' decision problem with decoupled payments

The farmers' objective function on implementing decoupled payments becomes

$$\pi^{\mathrm{d}} = \max_{A,V,K}$$

$$\sum_{f\in F}\left[\sum_{i\in I}\sum_{j\in J} A_{fij}\left\{p_i\,\mathrm{g}\left(V_{fiv},K_{fi}\right)+s^{\mathrm{d}}-\sum_{v\in V}c_v V_{fiv}-\mathrm{k}\left(\bar{k}_f,\mathrm{SFP}_f\right)K_{fi}\right\}+\mathrm{SFP}_f\right] \tag{1.40}$$

where as the constraint Equations (1.31)-(1.33) remain unchanged. I also ignore the possibility of alternative cross-compliance conditions, Equation (1.39), in order to focus on principles. Land use cross-compliance is incorporated directly in the objective function. The possibility for farmers to abandon land and hence miss out on the regional payment is captured by the fact that the block area constraints, Equation (1.33), can be slack, in which case the land rent will be zero and the land abandoned.

Denote the area of land that is not used in commodity production but satisfies the land use cross-compliance condition, as A_{fj}^{c}. The farmers' decision rules with decoupling become

$$\begin{aligned}\frac{\partial L^{\mathrm{d}}}{\partial A_{fij}^{*}}=0\Rightarrow p_i^{\mathrm{m}}\,\mathrm{g}\left(V_{fiv},K_{fi}\right)+\lambda_{fi}^{\mathrm{r}}+s^{\mathrm{d}} &= \sum_{v\in V}c_v V_{fiv}+\ \mathrm{k}\left(\bar{k}_f,\mathrm{SFP}_f\right)K_{fi}\\ &+\lambda_f^{l}\left(\mathrm{l}\left(A_{fij},K_{fi}\right)+\frac{\partial \mathrm{l}\left(A_{fij},K_{fi}\right)}{\partial A_{fij}}\right)+\lambda_j^{\mathrm{a}},\quad \forall fij\end{aligned} \tag{1.41}$$

$$\begin{aligned}\frac{\partial L^{\mathrm{d}}}{\partial A_{fj}^{\mathrm{c}}}=0\Rightarrow s^{\mathrm{d}} &= \sum_{v\in V}c_v V_{fcv}+\ \mathrm{k}\left(\bar{k}_f,\mathrm{SFP}_f\right)K_{fc}\\ &+\lambda_f^{l}\left(\mathrm{l}\left(A_{fij},K_{fi}\right)+\frac{\partial \mathrm{l}\left(A_{fij},K_{fi}\right)}{\partial A_{fj}^{\mathrm{c}}}\right)+\lambda_j^{\mathrm{a}},\quad \forall fj\end{aligned} \tag{1.42}$$

$$\frac{\partial L^{\mathrm{d}}}{\partial V_{fiv}^{*}}=0\Rightarrow p_i^{\mathrm{m}}\frac{\partial\,\mathrm{g}\left(V_{fiv},K_{fi}\right)}{\partial V_{fiv}}=c_v,\quad \forall fiv \tag{1.43}$$

$$\frac{\partial L^{\mathrm{d}}}{\partial K_{fi}^{*}}: p_i^{\mathrm{m}} \frac{\partial \mathrm{g}\left(V_{fiv}, K_{fi}\right)}{\partial K_f} + \lambda_f^l \left(-\frac{\partial \mathrm{l}\left(A_{fij}, K_{fi}\right)}{\partial K_{fi}} \right) = \mathrm{k}\left(\bar{k}_f, \mathrm{SFP}_f\right), \quad \forall f \qquad (1.44)$$

Rearranging the land use conditions, Equations (1.41) and (1.42), it can be seen that farmers allocate crops and the cross-compliance land use so that implicit marginal profits should be equalized across all crops and equated with $\lambda_j^{\mathrm{a}} - s^{\mathrm{d}}$. Since all land uses are eligible for the same decoupled payment s^{d}, this payment does not impact the choice between agricultural land uses. Hence even a decoupled regional payment should not distort commodity markets. If the market revenues from producing a commodity *i* on any block *j* are greater than the costs the farmer will use the block for commodity production. If not, then the block will be managed according to the cross compliance condition, and the land rent for this land will be s^{d} less management costs. If the implied management costs exceed s^{d}, then the land will be abandoned or turned over to the best non-agricultural land use (e.g., forestry).

The SFP does not appear directly in the first-order conditions, but only indirectly as it affects the cost of capital. If farmers were only to receive the SFP without a cross-compliance condition, then the decision to abandon land would rest solely on market returns to commodity production (and the reduced cost of capital implied by the SFP). The regional payment on the other hand "distorts" the choice between agricultural and non-agricultural land uses because agricultural land uses need only return a profit of $\lambda_j^{\mathrm{a}} - s^{\mathrm{d}}$ to motivate choosing them.

It is also apparent that factors that reduce the profitability of commodity production on a particular block will have a relatively stronger impact on production decisions than with coupled payments. Since decoupling reduces the relative profitability of producing commodities, the smaller a block is or the greater the distance from a farmstead the less likely it will be profitable to farm. Thus in the first instance it will be relatively small and distance blocks that are taken out of commodity production and converted to the "cross-compliance land use" or abandoned.

Equation (1.43) indicates that decoupled payments will result in identical or less intensive production since $p_i^{\mathrm{m}} \leq p_i$ (farmers use inputs until the marginal value product is equal to the price). If price falls then the marginal product must be increased through reduced intensity to meet costs. Total input use will also decline if the area of commodities is reduced because the cross-compliance land use does not require chemical inputs.

An potential impact that is not considered in the model is that of credit restrictions (which could be implemented by introducing a budget constraint). If input application has been constrained by credit restrictions (e.g., new member states) then a SFP could provide the capital necessary to purchase inputs on credit. In this case decoupled payments could lead to increased intensity and an increase in total input use. The extent of this effect is though an empirical question.

Equation (1.44) has no obvious interpretation since the RHS decreases in value due to the lower price and the RHS or cost of capital could decline due to increased wealth attributable to the SFP. However it does show that a decoupled payment could partly offset the impact of revenue reductions by indirectly reducing the costs of production.

An important question is whether decoupling will impact the relative profitability of different farms and hence accelerate structural change. Is there anything in the analysis that implies that the equilibrium number of farms might change? Not specifically since it will depend on the way different farms react to the wealth effect and reductions in the profitability of commodity production.

8.5 Predicted impact of decoupling on landscape characteristics

What then, does the analysis tell us about the effects of decoupling on the spatial arrangement of agriculture which is critical to environmental values? Firstly, with a SFP land that is not profitable for commodity production will be abandoned. Relatively small fields and/or fields that lie relatively distant from a farmstead will be the first to go (given homogeneous land quality). A regional payment on the other hand (with an implied cross-compliance condition) would increase the opportunity cost of abandoning land and ensure that a greater area is still managed by farmers. Land that otherwise might have been abandoned would now be managed according to the cross-compliance conditions.

Secondly, marginal farms with poor expansion possibilities will close down if the returns to their labour and capital fall below their opportunity cost. On the other hand more efficient farms that chose to expand to take advantage of scale economies will need more land, and will take over land that was previously farmed by exiting farmers. Hence land might continue to be managed despite a decrease in the number of farms. Farm expansion is therefore not necessarily a bad thing for the landscape. On the contrary, the possibility of expansion will be important for preserving landscape values, since decoupled payments imply commodity production will be less profitable than previously. Land that runs the greatewst risk of not being managed will be fields that are relatively small and/or lie relatively distant from remaining farmsteads. The preconditions for farm expansion (economies of scale) will though result in increasing average field size and reduced size variation. This due to a) abandonment of small fields or conversion of these to the cross-compliance land use, b) amalgamation of fields on the same block and c) specialization of production.

It seems clear then that decoupling will result in negative impacts on the environment because a) small, heterogeneous fields are important for landscape diversity and mosaic, and b) reduced habitat, no matter how small, will reduce the survival probability for species dependent on that habitat. The relevant empirical question is thus, how large will the impacts be? Are they sufficiently small to be overweighed by the social benefits of having commodity production more in tune with market demand? Or so large that the perceived benefits of decoupling are cancelled out by larger reductions in landscape values?

In the next section I examine how these expected changes in the spatial arrangement of agriculture due to decoupling will impact landscape values. This is done by integrating the indicators with the AgriPoliS model and simulating the effects of decoupling on a "provisional" model of the Jönköping region in Sweden. I say provisional because the model at this stage has not been sufficiently tested or validated. Rather the results are intended to be purely illustrative.

9 Landscape modelling in AgriPoliS

AgriPoliS is a spatial dynamic agricultural production model that is capable of simulating regional structural change (Balmann, 1997; Kathrin Happe, 2004). The implicit spatial dimension of AgriPoliS implies that it is also suitable for studying landscape evolution, which is a spatial phenomenon. However it only models space or the landscape in a stylistic rather than an explicit manner. It does not show the exact location of farms and fields found in the actual region but represents an abstraction of the real agricultural landscape (Kathrin Happe, 2004). Space in AgriPoliS is represented by a set of cells or plots, belonging to a two-dimensional grid. Each individual plot represents a standardized spatial entity of a specific size that can take on different states. In this idealized representation all aspects of the landscape not directly related to agricultural land-use (e.g., roads, rivers, lakes, etc.) are ignored. This abstraction level is justified by the fact that only agricultural land is assumed to be affected by changes in agricultural policy, whereas all other features of the landscape are assumed to be unchanged (remain constant). Two issues are addressed in this section. The first is whether AgriPoliS can be used to initialize a two-dimensional landscape that is representative of the actual landscape being modeled—just as it can reflect agricultural production. The second is that AgriPoliS does not currently allocate production activities over the landscape which makes it impossible to calculate any kind of patch or field metrics. A procedure is developed in Section 9.4 which makes this possible.

Before doing this it is necessary to reflect on whether a synthetic or "abstract" landscape is appropriate for modelling the effects of policy change on a real landscape? The practice of using synthetic landscapes to analyse landscapes is fairly common (R.V. O'Neill et al., 1992; Kimberly A. With, 1997). To model landscapes explicitly, requires enormous amounts of computing power and data generated from landscape images. It has been shown however, that synthetic or *neutral* landscapes are quite capable of reproducing the statistical characteristics of real landscapes and therefore being valid for landscape analysis (Saura and Martinez-Millan, 2000; X. Li et al., 2004). If a synthetic landscape has similar statistical properties to a real landscape then it will represent an abstraction of the real landscape and can be used to answer questions that are related to the statistical properties of the landscape (e.g., mean patch size and variation).

A common goal of landscape analysis is to quantify the statistical characteristics of a particular landscape through say image analysis. This is not a problem for agricultural landscapes in the EU, since government authorities have detailed information of field sizes and the type of agricultural crops grown on fields (i.e., patch size), from farmers' applications for support. I intend to show how this information can be utilized by AgriPoliS to initialize a synthetic agricultural landscape that is a statistical representation of the real landscape.

9.1 How "real" is the AgriPoliS landscape?

The current version of AgriPoliS generates a synthetic agricultural landscape in a two-stage process. In the first step it randomly distributes a set of different soil types over a grid of predefined size (e.g., arable and grassland). The soils are distributed according to their observed frequency in the region being modeled without attention to their actual position. In the second stage a set of farms that are representative of the structure of farming in the region are spread over the landscape. First farmsteads are randomly associated with a particular plot on the grid and secondly farms "take turns" to select plots of agricultural land—one plot at a time—until the farm is linked to an area of land that is equal to it's land endowment. The grid can be dimensioned to exactly match the total area of agricultural land in the region or "oversized" to increase the probability of farms being able to select plots of land that lie close to the farmstead.

In the next section I compare the synthetic landscape currently generated by AgriPoliS with a real landscape in terms of the distribution of *block* size, and then show how the landscape initialization procedure can be modified to produce a more "realistic" landscape. A block is a contiguous area of a particular soil type or land quality (a geophysically defined area of land). Blocks are separated by nonagricultural land uses such as forests, lakes, etc. or line elements such as roads and rivers. Blocks can cross farm borders and might be divided into any number of fields of different crops. To make the comparison possible Kellermann and Brady (Forthcoming) have introduced a new function into AgriPoliS to count contiguous plots at the regional and individual farm levels.

9.2 Distribution of blocks in a real landscape

I begin by presenting the distribution of arable and semi-natural grazing land blocks in the Swedish region known as Jönköping County, which is to be modeled in IDEMA. This data was made available by the Swedish Board of Agriculture from a database containing farmers' CAP support applications (SAM applications). After this the distribution of blocks generated by AgriPoliS is compared to the real landscape.

9.2.1 Example: Jönköping County in Sweden

Average block size of arable land in 2003 was 1.57 ha, Table 2, if all registered blocks no matter how small are included. If we exclude blocks that are less than 1 ha then mean size increases to 2.85 ha with standard deviation 2.89 ha. The large number of very small plots was somewhat surprising to find (median < mean). The cut-off to receive area payments however is 0.1 ha, so many small plots are probably not used in commodity production but are simply managed as set-aside to obtain the CAP area payment.

Table 2. Distribution of arable land in Jönköping

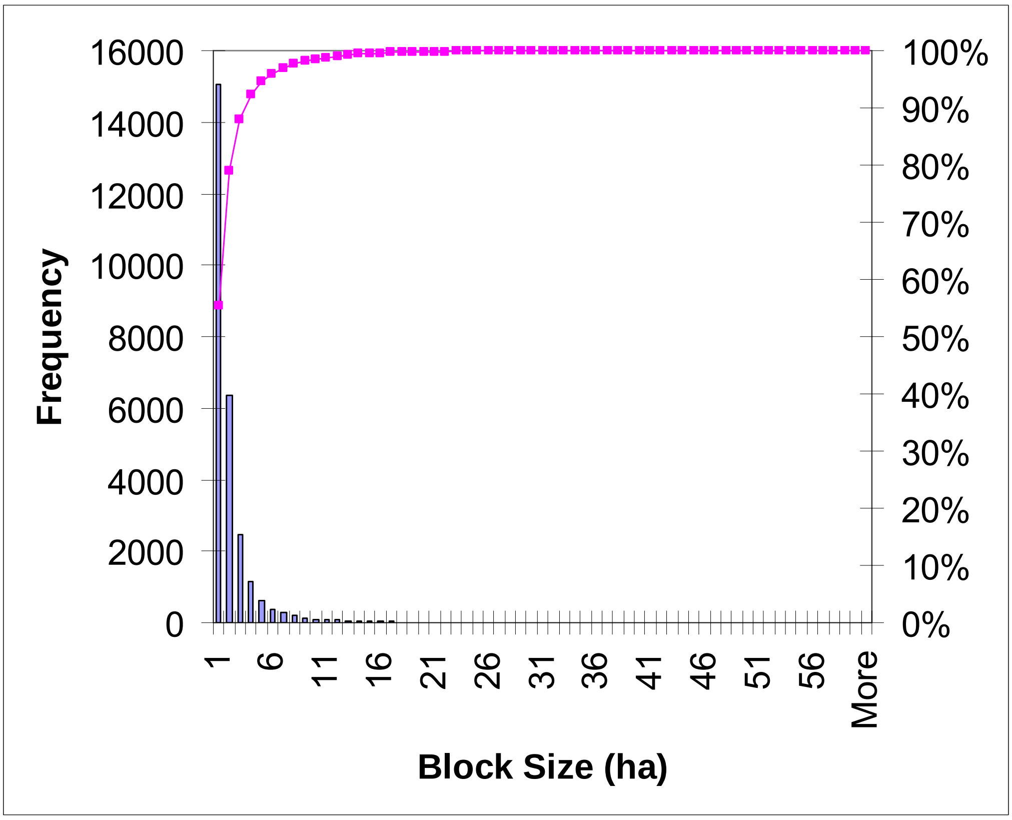
		Blocks	Area	Min	Max	Mean	Median	StD
Jönköping	All	27 124	42 554	0.03	57.75	1.57	0.87	2.29
	≥ 1 ha	12 404	35 314	0.98	57.75	2.85	1.88	2.89

Source:: Swedish Board of Agriculture

The histogram of all arable land, Figure 13, shows that the distribution is heavily skewed to the right. Roughly 55% of all plots are 1 ha or less, which I group into the size unit 1. This type of skewed distribution is typical of a log-normal distribution.

If blocks smaller than 1 ha are exclude then overall pattern is the same but average plot size increases to 2.85 ha. Using a plot size of 2.5 ha for simulations in AgriPoliS seems therefore to be a reasonable size, as the average plot size is concentrated around this level for commercial farms.

Figure 13. Histogram of arable blocks in Jönköping County



Average block size for grazing land in 2003 was 2.33 ha (Table 3). The distribution is heavily skewed to the right with 80% of blocks being 4 ha or less. Ignoring blocks < 1 ha increases average block size to 3.54 ha.

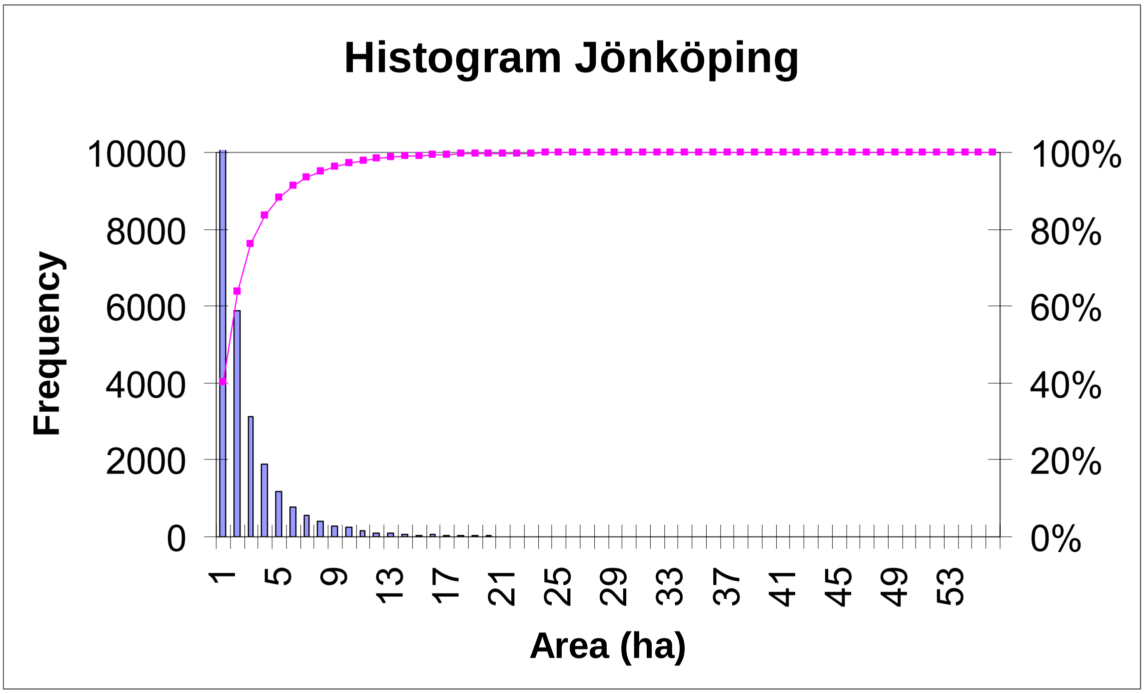
Table 3. Distribution of semi-natural grazing land Jönköping

		Blocks	Area	Min	Max	Mean	Median	StD
Jönköping	*All*	25 072	58 483	0.02	55.35	2.33	1.34	2.97
	> 1 ha	15 072	53 394	1	55.35	3.54	2.45	3.31

Source:: Swedish Board of Agriculture

The distribution also has a similar shape to that for arable land, Figure 14. 98% of blocks are less than 20 ha in size.

Figure 14. Histogram of semi-natural grazing land blocks in Jönköping



The distribution of agricultural land in Jönköping (Note the Västerbotten region follows a similar pattern) indicates that block size follows an asymmetric distribution that is heavily skewed to the right. The majority of blocks are relatively small and less than mean size. Such a distribution could be characterized by a log-normal distribution.

In reality field size is a continuous positive variable. For modeling purposes we need to define a minimum field or plot size to make computation feasible. The distribution of contiguous plots generated by AgriPoliS will therefore be truncated on the left (e.g., if minimum plot size is set at 2.5 ha then the distribution will be truncated at 2.5 ha). Since we are only considering commercial farms larger than 10 ha in IDEMA it seems reasonable to ignore fields that are not commercially viable for commodity production. This aside we need AgriPoliS to generate a synthetic landscape that otherwise reflects the characteristics of the actual landscape being modeled (i.e., similar mean and standard deviation that are log normally distributed).

9.3 Controlling landscape initialization in AgriPoliS

The standard initialization of the AgriPoliS landscape generates a synthetic landscape that is unlikely to be representative of the real landscape. For example the landscape generated for Jönköping had an average block size of 33.79 ha and a maximum of 4,115 ha which are obviously unrealistic for a region like Jönköping. If AgriPoliS is to be useful for landscape modelling then it will be possible to influence the landscape initialization procedure so that it produces a realistic distribution of block size. An essential characteristic of the real landscape that is missing in the standard initialization is the matrix element or proportion of non-agricultural land in the region. The more non-agricultural land the more likely agricultural land is to be spread out in the region.

It was found that the matrix effect could be introduced into the landscape initialization procedure by introducing a soil type representing "nonagricultural land" (Kellermann and Brady, Forthcoming). By introducing one more soil type than the set of agricultural soils (i.e., increasing the degrees of freedom) it was ispossible to reduce the probability that two agricultural soils of the same type occur adjacent to each other. The larger the proportion of nonagricultural land, the less likely two agricultural plots will occur adjacent to each other and hence the less likely a farm will have contiguous plots. On the other hand the standard "oversize" factor in AgriPoliS can be used to increase the likely hood that individual farms have contiguous plots of land.

By varying the proportion of nonagricultural land in the landscape and the "oversize" factor it was found to be possible to control the distribution of block size generated by AgriPoliS (Kellermann and Brady, Forthcoming). Increasing the proportion of nonagricultural land increases the number of blocks appearing in the landscape and reduces the mean and variance of block size. As the number of fields increases the median also moves further to the left. Introducing a non-agricultural soil type seems therefore to be a way of obtaining a more realistic landscape from the AgriPoliS landscape initialization.

In Table 4 the distribution of the synthetic landscape generated by AgriPoliS with the following parameter values: Nonagricultural land 1.6 and Oversize 1.01, is compared to that of the real landscape for blocks > 2.5 ha. From landscape theory it is known that once the matrix is greater than 57 % of the entire landscape then it will perculate the landscape (Saura and Martinez-Millan, 2000). This is the case in Jönköping where agricultural land is scarce, so setting the land factor to 1.6 obtains the effect that the non-agricultural land dominates the landscape. Setting the oversize factor close to one minimizes the probability that blocks of land belonging to any single farm will be contiguous given the land factor. From Table 4 it can be seen that the distributions of the synthetic and real landscapes are quite similar.

Table 4. Comparison of synthetic and real landscape characteristics

	AgriPoliS	**Actual >2.5ha**
Mean	5.20	5.34
Median	2.50	3.96
Standard Deviation	5.52	3.82
Range	72.50	55.25
Minimum	2.50	2.50
Maximum	75.00	57.75

The distribution also follows the skewed or log-normal form as is evident in the histogram of block size of the synthetic landscape, Figure 15. Not also that varying plot size is equivalent to truncating the distribution of contiguous plots and as such will have a direct impact on the distribution of field size. Reducing plot size and increasing matrix area will result in an increase in the number of fields and reduced mean field size (which has been tested but not shown).

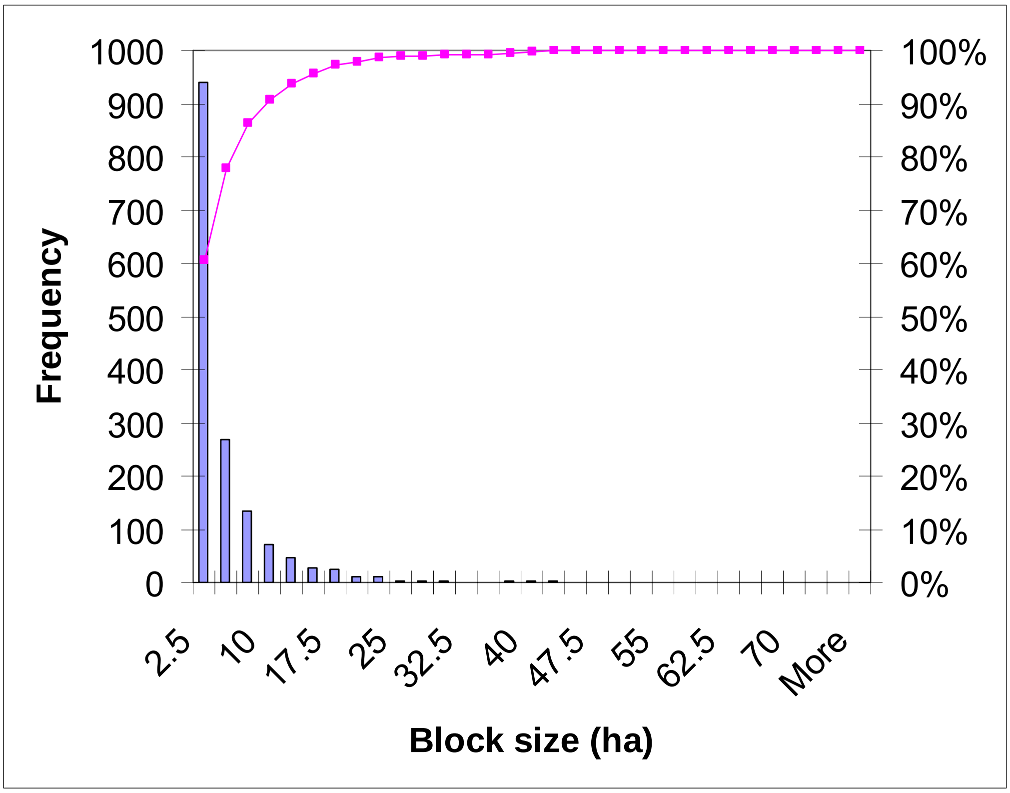


Figure 15. Histogram of arable blocks generated by AgriPoliS

Analysis of the distribution of grazing land blocks generated by AgriPoliS shows similar results to that of arable land so I do not reproduce the results here.

In conclusion it seems that manipulating the area of non-agricultural land and the oversize factor can be used to obtain a more realistic initialization of the landscape in AgriPoliS. Since the distribution of block size is similar to that of the real landscape then it should be adequate for the goals of environmental evaluation in IDEMA, i.e., evaluation at the regional level. It should be noted however that if we were to incorporate a more advanced landscape initioalization algorithm in AgriPoliS, such as that developed by Saura and Martinez-Millan (2000; 2003) then an almost perfect representation of a real landscape would be achievable. However an abstract representation of the real landscape is quite sufficient for our purposes.

9.4 Adapting AgriPoliS to derive the distribution of field size

Currently AgriPoliS doesn't associate specific agricultural activities to a specific plot of agricultural land in the synthetic landscape. Plots of land are only recognized as being of a particular soil type (e.g., arable , grazing land or part of the matrix). In order to calculate landscape metrics it will be necessary to allocate the optimal activity levels for each farm generated by the farm modeling in AgriPoliS to specific plots of agricultural land managed by each farm (see Section 8.2 on the premises for farmer decision making) . Calculating landscape indicators from activity levels alone is very limiting because this doesn't take account of field size, the most fundamental characteristic of an agricultural landscape.

In the next section I describe a procedure for allocating optimal activity levels to plots of agricultural land that is consistent with the optimizing structure of farmer decision making in AgriPoliS (Section 8.1) and the data output files that will need to be created to calculate landscape indicators. A hypothetical example of a simplified agricultural landscape is also used to demonstrate how landscape indicators will be calculated in AgriPoliS, and their applicability for evaluating landscape change.

9.4.1 Procedure for allocating production across the landscape

Like the landscape itself, patches comprising the landscape are not self-evident but must be defined relative to the phenomenon under consideration. In IDEMA, we are studying the evolution of the agricultural landscape in response to agricultural policy reform. Hence, a patch in the context of IDEMA is to be defined as a contiguous area of land that is used for a particular agricultural activity: the *field*. Land that is not agricultural land is assumed to belong to the Matrix element. This is because agricultural policy reform is only expected to affect agricultural land management decisions, with other land uses being held constant.

Given that the land allocation decision is implicit to the solution of the farmers profit maximisation problem, it should be possible to utilise the synthetic landscape and choice of optimal activity levels generated by AgriPoliS, to calculate the statistical distribution of field size. This can be done with recourse to the assumption that farmers aim to maximise farm profits and duality of the associated cost-minimisation problem. In section Section 8.2 it was shown that farmers minimize the costs of producing a crop by maximizing field size given their production constraints: larger fields are more profitable to farm than smaller ones and fields close to the farm are more profitable than those at a distant. These are the same decision rules that apply in AgriPoliS. Given this knowledge of how farmers frame their crop allocation decision, we should be able to replicate or model the allocation of crops to fields, and hence spread optimal levels of each production activity over the landscape in a fashion similar to that of farmers in reality and that is consistent with decision making in AgriPoliS.

The dual of the farmers profit maximisation problem is the minimisation of the costs of producing the optimal level of output. Since activity levels generated by AgriPoliS represent profit maximising levels of production given any set of farm and policy parameters, then a dual of this problem is to allocate optimal cropping levels across plots of agricultural land in the synthetic landscape such that the costs of producing crops is minimised. As was shown in Section 8.2, farmers will minimise the costs of crop production by growing their different crops, as far as is possible, on contiguous plots of land.

Given that farmers have maximized profits, then the following mathematical programme will allocate the optimal activity levels of different crops generated by AgriPoliS across the plots of land managed by each farm in the synthetic landscape so that the costs of production are minimized.

$$\max \sum_{i \in I} \sum_{j \in J} a_{fij}^2 \quad \text{Field size} \quad \forall f \in F \tag{1.45}$$

s.t.

$$\sum_j a_{fij} \le x_{fi}^* \quad \text{Optimal activity levels} \quad \forall f, i \tag{1.46}$$

$$\sum_i a_{fij} \le \bar{y}_{fj} \quad \text{Contiguous plots of land} \quad \forall f, j \tag{1.47}$$

$$a_{fij} \ge 0 \quad \text{No negative field sizes} \quad \forall f$$

where a_{fij} is field size and represents the contiguous area of block *j* that is allocated to cropping activity *i* by farm *f*, x_{fi}^* is the optimal level of activity *i* provided by the solution to farm *f*'s profit maximization problem, and $\bar{y}_{fj}$ is the area of block *j* managed by the farm (i.e., contiguous plots of agricultural land). The data required to solve this problem are optimal activity levels and the contiguous plots of land managed by each farm. Optimal activity levels are standard output from AgriPoliS and calculation of contiguous plots has now been made possible (Section 9.3). The solution to this problem is the set of fields $\{a_{ij}\}$ that maximize the sum of the square of field size subject to the constraints that total activity levels cannot exceed optimal levels and that the sum of field sizes can not exceed the area of contiguous plots of land. The nonlinear form of the objective function implies that larger fields are better than smaller. That is if fields are chosen that are smaller than is permitted by the constraints then the objective function will be lower than is possible. The quadratic form was chosen because it is straight forward to solve.

The solution to this problem is illustrated in Table 5. In this example there are three cropping activities $x_i \in$(Barley, Silage, Grass) of which the optimal activity levels are respectively $\mathbf{x}^* =$ (55, 42, 18) ha, and four contiguous plots managed by the farm $y_j \in$(a,b,c,d), the areas of which are respectively $\bar{\mathbf{y}} =$(60,10, 25,20). Optimal field sizes produced by the solution to (15)-(17) are given in the body of Table 5 (e.g, $a^*_{barley,a} = 55; a^*_{silage,b} = 10$; etc.) As can be seen this results in 6 fields of different crops, hence 6 patches will be present in the landscape: 1 field of barley, 1 field of grass and 4 fields of silage. Thus the farmer managers 4 blocks of land but grows 6 different crops.

This allocation seems reasonable in practice because a) a farmer will, given the availability of contiguous plots, grow the same crop on a single field and b) the allocation results in the optimal level of each cropping activity and all plots of land being used (i.e., the constraints of the problem are satisfied). This procedure can be repeated for all the farms belonging to a region to derive optimal field sizes implicit to the solution to the farmers optimization problem in AgriPoliS .

Table 5. Illustration of solution to crop allocation program

	Fields			
Contiguous plots	*Barley*	*Silage*	*Grass*	$\bar{\mathbf{y}}$
a	*55*	*5*		**60**
b		*10*		**10**
c		*7*	*18*	**25**
d		*20*		**20**
Optimal activity levels $(\mathbf{x}^*)$	**55**	**42**	**18**	**115**

9.4.2 Procedure for calculating edge length

A very important kind of habitat that appears in agricultural landscapes is the field edge. Total edge length is therefore an important landscape metric. Since it is now possible to calculate field and block size in AgriPoliS it should also be possible to calculate a measure of edge length, as this will be a function of field size and the proportion of a block covered by a particular field. An edge is defined as the length of the border between a particular field and nonagricultural landuses or the matrix. Borders between two agricultural land uses are therefore not considered to be an edge. The following procedure can be used to approximate the length of field edge associated with any block of agricultural land.

To make this possible it is necessary to make an assumption about the general shape of blocks in the landscape. Is the typical block round, square, oblong, etc.? The choice of shape is important for the absolute value of edge length but should not impact relative differences. This measure can in other words be used to rank different scenarios, e.g., total length is greater under scenario A than B, but not to say exactly how much longer.

The solution to the problem defined by Equations (1.45)-(1.47) yields the set of fields $\{a_{fij}\}$. The area of any block is $\sum_f \bar{y}_{fj}$. The assumption that blocks are square implies that the length of any side or edge of the block must be $\sqrt{\sum_f \bar{y}_{fj}}$ and total edge length, $\bar{e}_{fij}$, equal to

$$\bar{e}_j = 4\sqrt{\sum_f \bar{y}_{fj}} \quad \forall j . \tag{1.48}$$

If the block also forms a single field then the length of exterior field edge will also be $\bar{e}_j$. In the case where a block is occupied by more than one field then exterior edge length of individual fields will be less than $\bar{e}_j$, but there total length equal to $\bar{e}_j$. Assuming that exterior field edge is proportional to field size, then the length of exterior field edge, $\bar{E}_{fij}$, of any particular field occupying a block will be

$$\bar{E}_{fij} = \frac{a_{fij}}{\sum_f \bar{y}_{fj}} \bar{e}_j \tag{1.49}$$

recalling that $\sum_f \bar{y}_{fj} = \sum_f \sum_i a_{fij}$. This indicator makes it possible to calculate the exterior edge length of individual field types and to aggregate in relevant ways.

Calculation of this indicator is best illustrated with an example. Let us assume that a block comprises two fields: 30 ha of wheat and 20 ha of barley. Total block area is therefore 50 ha. According to Equation (1.48) the length of block edge will be $4\sqrt{30+20} = 28.28$. By Equation (1.49) exterior edge length of the wheat field is $(30/50)28.28 = 16.96$ and for barley $(20/50)28.28 = 11.31$, and total exterior edge length is equal to block edge length, i.e., $16.96 + 11.32 = 28.28$. This completes the procedure for calculating edge length.

10 Illustrative application of proposed landscape indicators

The aim of this section is to illustrate the applicability of the proposed landscape indicators by calculating indicators for a hypothetical landscape and a real landscape, Jönköping County in Sweden. I use a simple hypothetical agricultural landscape that is easily visualized to exemplify how the indicators proposed in Section 6 might be used to evaluate changes in landscape characteristics brought about by decoupling. This will hopefully provide intuition that is useful for interpreting the indicators calculated for a real landscape, that is not easily visualized on paper.

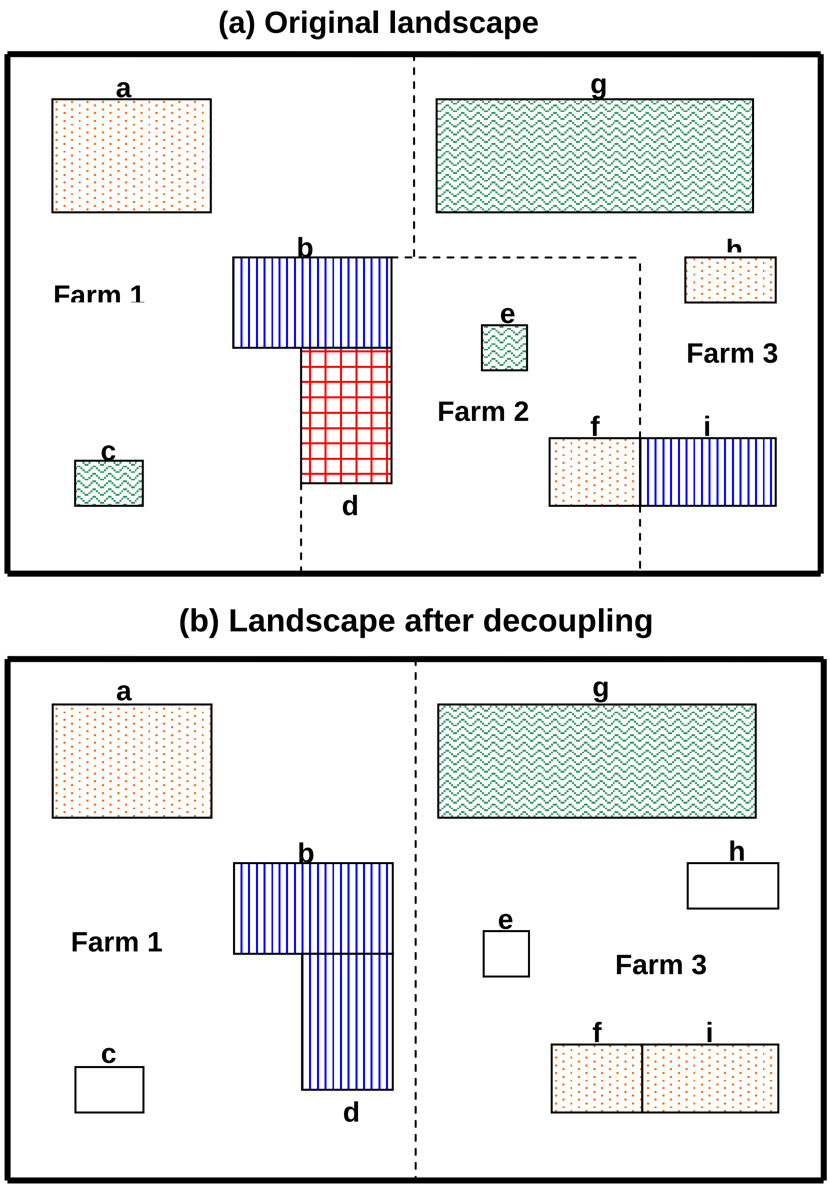
10.1 Exemplification of the landscape evaluation methodology

Figure 7 visualizes a small, hypothetical landscape. In this landscape there are three farms (1, 2 and 3), 4 agricultural land uses (barley, silage, arable grazing and semi-natural grazing land), 7 blocks (i to vii) and 9 fields (a to j) in the base year. Land that is not agricultural land is assumed to be part of the matrix element and therefore will not be affected by farmers' production decisions.

Imagine that a topographical map of the landscape has been overlaid by a 25x35 grid, where the area of each grid or plot is assumed to be 1 ha (100 m^2), implying that the total area of the landscape to be evaluated is 875 ha of which 205 ha is agricultural land. Each grid in the landscape is then shaded/coloured according to the agricultural activity conducted on the plot.

Plots belonging to the matrix element are left blank. A block per definition is an area of agricultural land fully enclosed by the matrix. Contiguous plots devoted to the same land use are defined as fields, and are encased by a hard line. Farm boundaries are marked by a dotted line. Agricultural land that is not used in production is assumed to be abandoned and to become part of the matrix. This is in principle how the AgriPoliS landscape is defined, where the matrix is the proportion of nonagricultural land (Section 9.3).

Figure 16. Evolution of a hypothetical agricultural landscape



Land use legend

Barley — Arable Grazing — Silage

Semi-natural grazing land — Abandoned land

---- Farm boundary

Source: Authors hypothetical example

Panel (a) in figure 7 represents the O*riginal* landscape before agricultural policy reform. Panel (b) on the other hand, represents the *New* landscape that has evolved as a result of "decoupling". By simply comparing the two images, a number of changes in the landscape are easily observed. Firstly Farm 2 has ceased to produce and some its land has been taken over by Farms 1 and 3 (fields *d* and *f*) or abandoned (field *e*). That the number of farms has decreased will not necessarily affect the structure of the landscape. What is important is what the new managers of the land decide to do with it. Fields *b* and *d*, and *f* and *i*, have been amalgamated to form two larger fields. Further, Farms 1 and 3 have abandoned production on their smaller and more distant fields (c and h). These are also the sort of changes in landscape characteristics predicted by the theoretical analysis in Section 8.5 as a result of decoupling. Changes in field size were easily calculated for this landscape and are presented in Table 6.

Table 6. Area of each land use in Original and New landscape (ha)

Farm	Landscape	Barley	Silage	Arable Grass	SemiNat Grazing	Aband-onded	Total Area
Farm 1	*Original*	35	0	28	6	0	**69**
Farm 1	*New*	35	0	52		6	**93**
Farm 2	*Original*	12	24		4	0	**40**
Farm 2	*New*	0	0	0	0	0	**0**
Farm 3	*Original*	8		18	70	0	**96**
Farm 3	*New*	30	0	0	70	12	**112**
Region (∑x_i)	*Original*	**55**	**24**	**46**	**80**	**0**	**205**
Region (∑x_i)	*New*	**65**	**0**	**52**	**70**	**18**	**205**

Source: Authors hypothetical example

What conclusions can be drawn about the New landscape? The process of structural change has resulted in a reduction in diversity of field types: silage is no longer produced and the production of barley and grass has been concentrated to larger fields. By abandoning some smaller fields and amalgamating others, the average field or patch size in the landscape has increased and the variability of patch size has decreased. The new landscape is therefore characterized by larger field size, less diversity of land use and a simpler mosaic. In other words landscape diversity has declined which reduces the value of the agricultural landscape to society (Section 4).

The hypothetical landscapes illustrated in Figure 7 are fairly easy to compare visually. However, this will not be the case when analysing a landscape comprising hundreds of farms and a great variety of land uses, as will be the case in IDEMA. In this situation landscape metrics are needed to summarise the vast quantity of information that would be contained in a map of the landscape. To illustrate how landscape indicators can be applied to evaluate changes in the landscape, I will now compare the landscapes presented in panels (a) and (b) of Figure 12 using the indicators developed in Section 6. The fundamental landscape indicators that represent the distribution of field size (p_i proportional abundance, μ_i mean field size, σ_i standard deviation and CV_i coefficient of variation) are summarized in Table 7.

Table 7. Statistical distribution of field/patch size

Statistic	Landscape	Barley	Silage	Arable Grass	SemiNat Grazing	Total Area	Change %
p_i	*Original*	0.27	0.12	0.22	0.39	**n/a**	
p_i	*New*	0.35	0.00	0.28	0.37	**n/a**	
μ_i	*Original*	18.33	24.00	23.00	26.67	**68.33**	
μ_i	*New*	32.50	0.00	52.00	70.00	**93.50**	**37 %**
σ_i	*Original*	14.57	0.00	7.07	37.54	**28.01**	
σ_i	*New*	3.54	0.00	0.00	0.00	**9.19**	
CV_i	*Original*	0.79	0.00	0.31	1.41	**0.41**	
CV_i	*New*	0.11	0.00	0.00	0.00	**0.10**	**-76 %**

Each of these indicators can be related to the qualitative conclusions drawn above. For example, mean field size has increased by 37 % and the coefficient of variation has decreased by 76 %. Thus mosaic has been significantly reduced

Shannon's Diversity index for each farm and the landscape are presented in Table 8. As can be seen the reduction in the number of small fields and the number of different cropping activities has quite a dramatic impact on this index. The relative reduction in structural diversity for the region is 17%.

Table 8. Shannon's diversity index for Original and New landscape

Scale	Landscape	Shannon's Index	Relative Change
Farm 1	*Original*	**0.92**	
Farm 1	*New*	**0.67**	-27%
Farm 2	*Original*	**0.90**	
Farm 2	*New*	**0.00**	-100%
Farm 3	*Original*	**0.75**	
Farm 3	*New*	**0.61**	-19%
Region	*Original*	**1.31**	
Region	*New*	**1.09**	-17%

For actual landscape evaluation it will be necessary to provide complimentary information about the region that can be used to judge the significance of the impacts, e.g., minor, significant or dramatic. A raw indicator is not particularly useful if it cannot be related to changes in human welfare in some way: even if it is descriptive rather than a monetary valuation. This will require accumulating some knowledge of the environmental characteristics of the region and relating them to changes in the indicators. A significant impact might imply that visual changes in the landscape will be detectable by humans and could impact tourism negatively. A dramatic change might imply that defining characteristics of the landscape would be lost (e.g., haymaking in the summer). This type of complementary information is mandatory for "putting things in perspective" especially for policymakers, as it is often not possible to value environmental impacts in monetary terms if some sort of valuation study has not been done.

Table 9 shows baseline landscape indicators for Jönköping County that have been calculated with the aid of AgriPoliS using the landscape modelling methodology described in Section 9. Only baseline indicators are presented in this Working Paper because AgriPoliS is still in the process of being validated for the region (i.e., realistic future scenarios are currently not possible to obtain). These statistics are for illustrative purposes only. A number of characteristics of the Jönköping landscape that become obvious from these statistics are, for example:

a) the landscape has a fine mosaic (many small fileds)

b) there are a large number of fields per farm

c) grazing land is the dominating patch type

d) arable fields are important for mosaic/heterogeneity

Table 9. Baseline patch/field statistics for Jönköping County, Year 2003

Field statistics	Barley	Triticale	Set-a-side	Arable grazing	Arable silage	Grazing Land	Region
Total fields	256	928	369	408	752	1,551	4,264
Total area	693	3,168	386	1,206	2,626	4,423	12,502
Mean fields/farm	1.12	4.07	1.62	1.79	3.30	6.80	18.70
Field size							
Mean	2.70	3.40	1.00	3.00	3.50	2.90	3.00
Standard Deviation	21.30	14.00	6.90	18.40	15.90	11.50	19.00
Coefficient of variation	7.90	4.10	6.60	6.20	4.60	4.00	6.40
Shannon Diversity index							
Farm average	1.17						
Region average	1.37						
Farms	228						

Source: AgriPoliS using actual structural data

An expected impact of decoupling in this region is that grain production will become less profitable and considerable areas of grain (e.g., 15-25 %) will be idled as grass or abandoned. In terms of national grain production this will have only a marginal impact, but for landscape mosaic it would have a considerable negative impact. This is because the aesthetic value of the landscape would be reduced.

This is just one example of the type of analyses that will be conducted with the proposed landscape indicators. Other negative impacts on the landscape that can be evaluated are (Armsworth *et al.* 2004);

- *Conversion* from one land-use to another (e.g., land abandonment)
- *Degradation*, reduced land quality without destroying completely (e.g., below optimal grazing pressure).

- *Fragmentation*, the process of altering the size, shape or connectivity of the mosaic of patches defining a landscape (i.e., changes in the size or distribution of a particular land use).

10.2 Biodiversity Indicators

To illustrate the usefulness of the proposed biodiversity indicators developed in Section 5.3 I compare baseline land uses in Jönköping County with a hypothetical decoupling scenario. The land use changes are therefore fictive and are for illustrative purposes only (but the changes are not unrealistic). Recall from Section 5 that a meaningful measure of the value of biodiversity must consider species that are unique in some way and not simply the total number of species associated with a particular land use. We adopt endangered species as our measure of uniqueness. A reduction in the area of suitable habitat would reduce the survival probability of an endangered species. If a species becomes extinct then the diversity of the set of species in the world would decline. Hence endangered species are a relevant measure of uniqueness.

Sweden has developed a database (ArtDataBanken, 2005) of endangered species that complies with the internationally accepted criteria for classifying species as endangered (i.e., red-listing) established by the International Union for Conservation of Nature (IUCN, 2001). ArtDataBanken (2005) holds a lot of information about species found in Sweden that are considered to be endangered (according to IUCN criteria), such as the type of habitat they rely on and in which regions they can be found. Table 10 provides an overview of relevant data from the ArtDataBanken of endangered species found in agricultural habitat in Jönköping County.

Table 10. Biodiversity value of agricultural habitat in Jönköping County

Habitat/Land use	Unique [1] Species	Current [2] Area (ha)	Productivity[3] C_i	Marginal Div.Value
Crop land	19	3,861	3.96	0.0009

Green Set-a-side	14	386	4.52	0.0069
Arable grazing	8	1,206	2.08	0.0013
Arable silage	5	2,625	1.12	0.0004
Grazing land	146	4,423	29.63	0.0063
All land	147	12,500	24.49	0.0022
Arable edges	32	1000	8.61	0.0061
Average				**0.0034**

Source: (1) ArtDatabanken (2005), (2) Note normalised values from AgriPoliS, (3) Derived using Equation (1.20) *(4) Derived using equation* (1.7)

To obtain the necessary parameters of the species-area relationship I have used the transformation shown in Equation (1.20). The implicit assumption made is that the observed number of endangered species and area of habitat represents an equilibrium. For example the 3,861 ha (these are normalised hectares) of crop land in Jönköping County contributes to the survival of 19 endangered species. The productivity parameter C_i in Table 10 is obtained by plugging 19 and 3,861 into Equation (1.20). The theoretical appropriateness of this transformation has been established in Section 5.4. Marginal diversity value is the implied reduction in species value if 1 ha of habitat is lost. Grazing land and field edges are shown to have very high marginal diversity values because they are very productive habitat (this is also consistent with current ecosystem knowledge). Green set-a-side also has high marginal value but this is simply because there is relatively little area of this habitat in the baseyear. However, the productivity of this land use is relatively low. This is a potential trap for policy analysis.

Table 11 shows the results of a hypothetical decoupling and associated land use change on the value of biodiversity in Jönköping County. Note first that any reduction in the area of a particular habitat results in reduced diversity value. This follows from the species-diversity relationship, which implies that the probability of species survival will fall if habitat is reduced. This result is consistent with the view of conservation ecologists that the current area of agricultural habitat in Jönköping must be maintained and the Swedish Environmental goal that 100 % of grazing land area should be maintained: any reduction in area will result in a loss in species!. The relevant question for our analysis is therefore to try and say by how much. The relevant question for the policymaker is then to decide whether the market gains of decoupling are sufficient to motivate the costs of lost environmental values.

Table 11. Impact of land use change on value of diversity

Habitat/Land use	New Area	Change in Area	Unique Species Before	After	Value of Change	% Value Change	Change Spec/ha	Marg.Div After
Crop land	2,498	-1,363	19	17.49	-1.51	-8%	0.0000	0.0064
Green Set-a-side	1,117	731	14	17.13	3.13	22%	0.0003	0.0090
Arable grazing	623	-583	8	7.06	-0.94	-12%	0.0002	0.0083
Arable silage	1,627	-998	5	4.57	-0.43	-9%	0.0000	0.0024
Grazing land	3,670	-753	146	140.91	-5.09	-3%	0.0000	0.0360
All land	9,536	-2,965	147	139.63	-7.37	-5%	0.0000	0.0167
Arable edges	400	-600	32	26.89	-5.11	-16%	0.0002	0.0474
Total			371	353.68	-15.25	-4%		0.0180

Note the implications of high "endangered species" productivity, which is illustrated in Figure 17. Set-a-side has increased by 189 % or 731 ha and the number of species by 3.13, on the other hand grazing land has declined by 17 % or 753 ha and the number of species by 5.09. Large reductions in relatively unproductive habitat (e.g., silage) will have a relatively small impact on diversity value. However, edge effects which are an indirect consequence of arable farming are a relatively productive habitat. Thus it is not necessarily the land use itself that is important but the contrast it provides to the surrounding landscape or matrix. What is important to note here is that the proposed value of biodiversity indicator captures the trade-offs, to some degree, that seem to be implied by the policy debate: all agricultural habitat is important for biodiversity but some is more important than others.

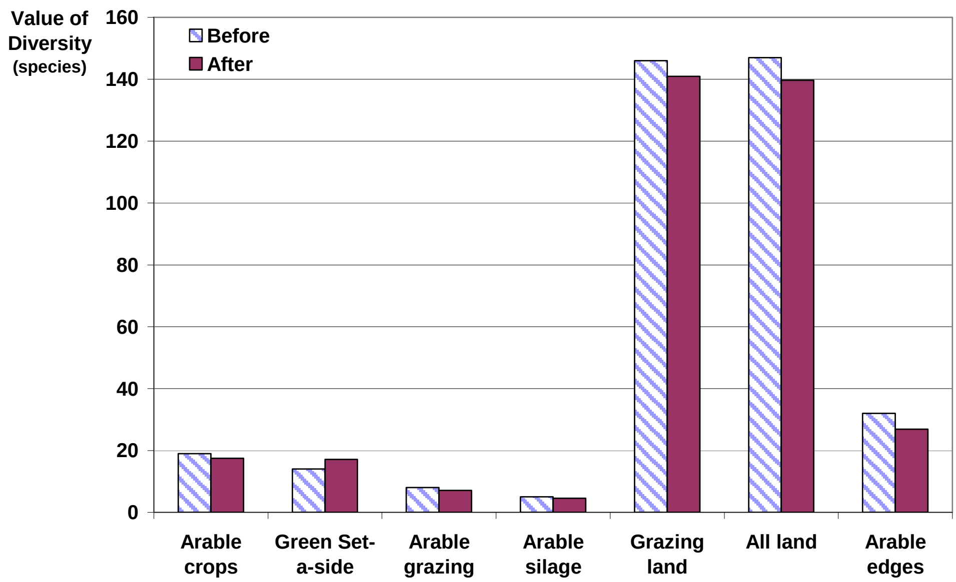


Figure 17. Comparison of diversity value of different land uses/habitat

This preliminary result, that we can expect a reduction in the value of biodiversity because of decoupling, however, seems to conflict with other studies of the phenomena. For example Lehtonen et al. (2005) state *"[due to decoupling] the amount of green fallow will increase considerably. As a result, the agricultural land will become biologically richer"*. The reason for this difference in conclusions can be attributable to the choice of indicator and definition of the value of biodiversity. Lehtonen et al. measure diversity value in terms of species richness and use a linear measure of the value of habitat. The advantage of our measure is that it is grounded in both diversity theory (Weitzman Diversity) and landscape ecology theory (species area relationship). Nevertheless will be important in the empirical evaluation to more closely compare and ciontrast the results obtained in IDEMA with other studies.

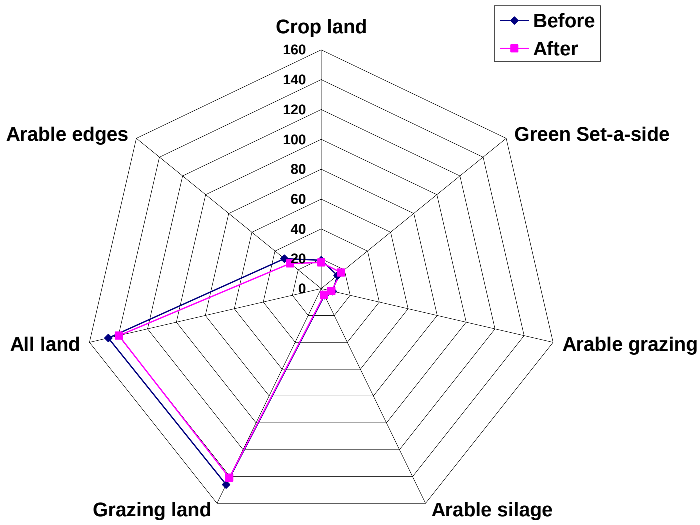


Figure 18. Impact of land use change on diversity value of species (1 = 1 species)

The diversity indicators should be evaluated independently of each other and not aggregated, as this would risk double counting. The aggregate impact can be illustrated in a radar diagram which is capable of capturing tradeoffs between different habitat. Figure 18 shows absolute changes in indicator values which might be used to obtain an indication of the total impact on biodiversity value (i.e., how many species are at risk?) whereas as Figure 19 shows relative changes (e,g,. what is the impact on a particular type of habitat). Given that we have descriptive information of the importance of a particular habitat for biodiversity value, a relative index (which requires no specific data and can be calculated by simply setting the productivity parameter to 1) can be calculated. This index can be used to say what the impact of decoupling will have on species associated with this type of habitat, but will not quantify the number of species at risk. Those with relevant ecosystem knowledge will though be able to make expert judgements.

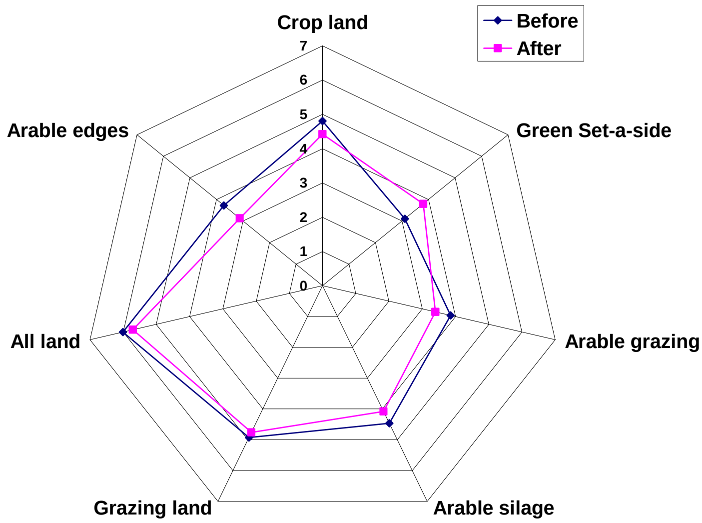


Figure 19. Relative impact of land use change on value of diversity

11 Summary

The aim of this Working Paper was to develop a methodology suitable for evaluating the regional environmental impacts of *decoupling* agricultural support from commodity production. Decoupling is a radical policy change that is expected to influence the regional structure of agricultural production, and as a consequence influence most environmental impacts associated with agriculture. However, the overriding social concern seems to be the loss of environmental benefits produced jointly with commodities, which has manifested in the concepts of the "European model of Agriculture" and "Multifunctionality". Further more, structural change implies that land use will change (e.g., through land abandonment or idling land) which is of course, the principle determinant of the value of a particular agricultural landscape. Environmental evaluation in IDEMA will therefore focus on the impact of decoupling on *landscape values*, principally the value of biodiversity, and the diversity and mosaic of the landscape (which are positively related to specific landscape values such as cultural heritage, amenity, recreation and knowledge value) . Pollution issues will also be considered but only through the use of rudimentary indicators such as nutrient and water balances at the farm level.

Environmental evaluation we will be done for a subset of representative regions:

1) Sweden─representative of marginal agricultural areas with high landscape values (two regions including an LFA region).

2) Czech Republic─new member agriculture (one region).

3) Italy─Mediterranean agriculture (two regions).

A general problem that is faced when attempting to do economic evaluation of the environment, is that environmental impacts (positive or negative) are not priced on markets, they are external effects (a form of market failure). In the absence of market prices it is necessary to use alternative indicators of environmental quality that can be related indirectly to the perceived social benefits or costs of the impact. This paper develops a number of indicators that can be used for evaluating changes in biodiversity and landscape value at the regional or landscape level.

To ensure that the proposed indicator of biodiversity value is relevant for policy anlaysis it is defined in terms of diversity theory (Weitzman, 1992), a branch of economic theory The empirical indicator was then constructed by drawing on the species-area relationship from landscape ecology theory. This new indicator seems to be superior to other indicators used at the landscape level because it a) measures diversity value rather than species richness and b) recognises important trade-offs between land use area and the value of biodiversity that is not captured in simple linear species "richness" indicators.

The choice of landscape indicators are based on Mosaic Theory from landscape ecology (Duelli, 1997) and the Weitzmen theory of diversity. To implement landscape indicators for empirical analysis the spatial modelling capacity of AgriPoliS has been further developed so that the synthetic landscape generated by AgriPoliS can be initialized to reflect the statistical characteristics of the real landscape.

AgriPoliS is not an explicit spatial modelling system like a GIS but creates a synthetic landscape from regional data on soil types. To date, the synthetic landscape has been a fairly crude representation of the real landscape and therefore not suitable for landscape analysis. Neither have production activities been spread across the landscape so that it could be seen where production was taking place. A development made in this paper is that it is now possible to initialise the synthetic landscape so that it has "landscape" characteristics similar to the real landscape being modelled (i.e., the distribution of field/patch size). A procedure has also been developed to allocate production activities to individual plots of land in the landscape. This procedure provides a consistent spatial representation of agricultural production to conduct landscape analysis.

As AriPoliS simulates the effects of policy on structural change, the developments made in this paper provide a tool to link structural change to changing patterns in the (synthetic) landscape and the value of biodiverdsity.

As a complement to the biodiversity and mosaic indicators, we will also calculate a range of standard indicators: nutrient\water balances, livestock densities, chemical intensity, etc. to draw conclusions about potential pollution impacts . Note that the relevant spatial scale for modeling chemical and nutrient pollution is the watershed, however, IDEMA focuses on homogenous agricultural regions. This reduces IDEMA's suitability for studying water quality issues. These issues (i.e., nonpoint source pollution) are however the focus of a similar research project (GENEDEC) which utilizes models more suitable for this type of analysis.

Summarizing, environmental analysis in IDEMA will be based on an abstract representation of the landscape, rather than a real landscape. However, this is consistent with the modelling approach in AgriPoliS (abstract representation of farm production and regional structure) and the regional scale of the project.

12 References

Anderson, K., & Blackhurst, R. (1992). The greening of world trade issues. In K. Anderson & R. Blackhurst (Eds.), *Trade, the Environment and Public Policy* (pp. 3-22). Ann Arbor: University of Michigan Press.

Andersson, F. (2004). *Decoupling: The Concept and Past Experiences* (IDEMA/SLI Working Paper No. 2004:1). Lund: Swedish Institute for Food and Agricultural Economics.

Andersson, F. (2005). *The Swedish 1990 Agricultural Reform* (IDEMA/SLI Working Paper No. 2005:1). Lund: Swedish Institute for Food and Agricultural Economics.

Armsworth, P. R., Kendall, B. E., & Davis, F. W. (2004). An introduction to biodiversity concepts for environmental economists. *Resource and Energy Economics, 26*(2), 115-136.

ArtDataBanken. (2005). *The 2005 Red List of Swedish Species.* Uppsala: ArtDataBanken, Swedish University of Agricultural Sciences (SLU).

Balmann, A. (1997). Farm-based modelling of regional structural change: A cellular automata approach. *European Review of Agricultural Economics, 24,* 85-108.

Berland, O. (1994). *Valuing Multidimensional Environmental Changes with Contingent Ranking* (Discussion Paper No. #D-15/1994). Ås: Agriculturasl University of Norway, Department of Economics and Social Sciences.

Björklund, J., Limburg, K. E., & Rydberg, T. (1999). Impact of Production Intensity on the Ability of the Agricultural Landscape to Generate Ecosystem Services: An Example from Sweden. *Ecological Economics, 29,* 269-291.

Bonnieux, F., Rainelli, P., & Vermersch, D. (1998). Estimating the supply of environmental benefits by agriculture: A French case study. *Environmental & Resource Economics, 11*(2), 135-153.

Brady, M. (2003). The Relative Cost-Efficiency of Arable Nitrogen Management in Sweden. *Ecological Economics, 47*, 53-70.

Brouwer, F. (2002). *Effects of Agricultural Policies and Practices on the Environment: Review of Empirical Work in OECD Countries*. Paris: Directorate for Food, Agriculture and Fisheries, Organisation for Economic Co-operation and Development.

Buller, H., Wilson, G. A., & Höll, A. (2000). *Agri-environmental Policy in the European Union.* Hampshire: Ashgate.

Chambers, R. G. (1988). *Applied Production Analysis: A Dual Approach.* Cambridge: Cambridge University Press.

Duelli, P. (1997). Biodiversity evaluation in agricultural landscapes: An approach at two different scales. *Agriculture Ecosystems & Environment, 62*(2-3), 81-91.

E. Goddard, A.Weersink, K. Chen, & C.G. Turvey. (1993). Economics of Structural-Change in Agriculture. *Canadian Journal of Agricultural Economics-Revue Canadienne d Economie Rurale, 41*(4755), 475-489.

Ekman, S. (2002). *Cost-Effective Nitrogen Leaching Reduction as Influensed by Linkages Between Farm-Level Decisions* [Ph.D. thesis]. Uppsala; Department of Economics, Swedish University of Agricultural Sciences.

FAO. (1996). *Control of Water Pollution from Agriculture* (Irrigation and Drainage Paper No. 55). Rome: Food and Agriculture Organization of the United Nations.

Forman, R. (1995). *Land mosaics: The ecology of landscapes and regions.* Cambridge: Cambridge University Press.

Forman, R., & Gordon, M. (1986). *Landscape Ecology.* New York: John Wiley & Sons.

Gustafson, E. J. (1998). Quantifying Landscape Spatial Pattern: What is the State of the Art? *Ecosystems, 1,* 143-156.

Hanley, N. (1991). *Farming and the Countryside: An Economic Analysis of External Costs and Benefits.* Oxon, UK: CAB International.

IDEMA. (2003). *The Impact of Decoupling and Modulation in the Enlarged Union: A sectoral and farm level assessment (IDEMA)* (Sixth Framework Programme Priority 8.1: Policy Orientated Research). . http://www.sli.lu.se/IDEMA/.

IUCN. (2001). *IUCN Red List Categories and Criteria version 3.1.* Gland, Switzerland and Cambridge: Species Survival Commission.

Kathrin Happe. (2004). *Agricultural policies and farm structures: Agent-based modelling and application to EU-policy reform.* Unpublished doctoral dissertation, Institute of Agricultural Development in Central and Eastern Europe (IAMO), Halle.

Kellermann, K., & Brady, M. (Forthcoming). *Documentation of new functions for landscape modelling in AgripoliS*. Halle: Institute of Agricultural Development in Central and Eastern Europe (IAMO).

Kimberly A. With. (1997). The application of neutral landscape models in conservation biology. *Conservation Biology, 11*(52), 1069-1080.

Lankoski, J., & Ollikainen, M. (2003). Agri-Environmental Externalities: A Framework for Designing Targeted Policies. *European Review of Agricultural Economics, 30*(1-2), 51-75.

Lehtonen, H., Lankoski, J., Niemi, J., & Ollikainen, M. (2005). *The Impacts of Alternative Policy Scenarios on Multifunctionality.* Presented at the XIth International Congress of the European Association of Agricultural Economists (EAAE), August 24-27, Copenhagen, Denmark.

Lekakis, J. N., & Pantzios, C. (1999). Agricultural Liberalization and the Environment in Southern Europe: The Role of the Supply Side. *Applied Economics Letters, 62*(7), 453-58.

Miettinen, A., Lehtonen, H., & Hietala-Koivu, R. (2004). On diversity effects of alternative agricultural policy reforms in Finland: An agricultural sector modelling approach. *Agricultural and Food Science, 13*(3), 229-246.

Nehring, K., & Puppe, C. (2002). *Econometrica, 70*(3), 1155-1198*A Theory of Diversity*.

Noss, R. (1990). Indicators for monitoring biodiversity: A hierarchical approach. *Conservation Biology, 47*(4), 355-364.

OECD. (1997). *Environmental benefits from agriculture: issues and policies (The Helsinki Seminar)*. Paris: Organisation for Economic Cooperation and Development.

OECD. (1998a). *Agriculture and the Environment: Issues and Policies*. Paris: Organisation for Economic Cooperation and Development.

OECD. (1998b). *Agriculture in a Changing World: which policies for tomorrow?* Paris: Organisation for Economic Cooperation and Development.

OECD. (2001a). *Decoupling: A Conceptual Overview*. Paris: Organisation for Economic Cooperation and Development.

OECD. (2001b). *Multifunctionality: Towards an Analytical Framework*. Paris: Organisation for Economic Cooperation and Development.

Olson, R. K., & Francis, C. A. (1995). 1. In R. K. Olson, C. A. Francis & S. Kaffka (Eds.), *A hierarchical framework for evaluating diversity in agroecosystems* (pp. 5-34). Madison: American Society of Agronomy.

Peerlings, J., & Polman, N. (2004). Wildlife and landscape services production in Dutch dairy farming; jointness and transaction costs. *European Review of Agricultural Economics, 31*, 427-449.

R.V. O'Neill, R. Gardener, & M.G. Turner. (1992). A hierarchical neutral model for landscape analysis.' *Landscape Ecology, 7*(1-2), 55-66.

Riitters, K. h., O'Neill, R. V., Hunsaker, C. T., Wickham, J. D., Yankee, D. H., Timmins, S. P. et al. (1995). A Factor-Analysis Of Landscape Pattern and Structure Metrics. *Landscape Ecology, 10*(1), 23-39.

Romstad, E. (1999). *Policies for Promoting Public Goods in Agriculture* (Discussion Paper No. #D-20/1999). Ås: Department of Economics and Social Sciences, Agricultural University of Norway.

Romstad, E., A. Vatn, P. K. Rorstad, & V. Soyland. (2000). *Multifunctional Agriculture: Implications for Policy Design* (Report No. 21). Ås: Agriculturasl University of Norway, Department of Economics and Social Sciences.

Rosenzweig, M. (1995). *Species diversity in space and time.* Cambridge: Cambridge University Press.

Saura, S. (2003). *SimMAP 2.0 Landscape categorical spatial patterns simulation software: User's Manual*. http://www.udl.es/usuaris/saura (Accessed 25 Sept 2005).

Saura, S., & Martinez-Millan, J. (2000). Landscape patterns simulation with a modified random clusters method. *Landscape Ecology, 15*(7), 661-678.

Shortle, J. S., & Horan, R. D. (2001). The Economics of Nonpoint Pollution Control. *Journal of Economic Surveys, 15*(3), 255-289.

Urban, D., R.V. O'Neill, & H.H. Shugart. (1987). Landscape Ecology. *BioScience, 37*, 119-127.

Weitzman, M. L. (1992). On Diversity. *The Quartery Journal of Economics, 107*(2), 363-405.

Weitzman, M. L. (1998). The Noah's Ark Problem. *Econometrica, 66*(6), 1279-1298.

X. Li, H.S. He, X. Wang, R.Bu, Y. Hu, & Y. Chang. (2004). Evaluating the effectiveness of neutral landscape models to represent a real landscape. *Landscape and Urban Planning, 69*, 137-148.