DISSERTATION

Cost-effectiveness of organic farming for achieving environmental policy targets in Switzerland



Christian Schader

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Cost-effectiveness of organic farming for achieving environmental policy targets in Switzerland

A thesis presented for the Degree of Doctor of Philosophy

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Institute of Biological, Environmental and Rural Sciences

Aberystwyth University

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'*Rational policy appraisal requires the consideration of costs and benefits.* […] *While defining and measuring the costs of agri-environmental schemes appears difficult enough, the problems multiply with benefits.*'

D. Pearce

Summary

The aim of this PhD thesis was to calculate the cost-effectiveness of organic farming in achieving environmental policy targets compared to single-agri-environmental policies.

Using a theoretical model, it was demonstrated that financial support for organic farming does not in principle contradict the Tinbergen Rule, even if there are other targeted policy measures which are more cost-effective in achieving specific environmental goals. Hence, organic farming should be included as an option within a mix of other policies as long as its cost-effectiveness with respect to the overall set of policy goals is superior to that of a combination of other policy instruments.

The cost-effectiveness of agri-environmental policies can be understood as a function of policy uptake, environmental effects, and public expenditure. Taking the Swiss agricultural sector as an empirical case study, both the costs and effects of organic farming and other single agri-environmental measures were calculated at sector level. Therefore, the economic sector model FARMIS was extended by three modules encompassing a) life cycle assessments for fossil energy use, biodiversity and eutrophication, b) public expenditure, including policy-related transaction costs, and c) uptake of agri-environmental policies.

The calculations revealed a slightly higher abatement cost with organic farming of 14 CHF/ha for a 1 % average improvement in the environmental indicators, compared to a combination of three single agri-environmental policies (11 CHF/ha), including both extensification of arable land and meadows. In view of total public expenditure on agriculture of 2 to 3 kCHF per ha in Switzerland, these differences can be understood as marginal. Sensitivity analyses confirm that the cost-effectiveness of organic agriculture and combined agri-environmental policies is very similar. Thus it is concluded that financial support for organic farms in Switzerland is economically sound in view of the provision of public goods.

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List of abbreviations

ABC	Abatement cost
AEM	Agri-environmental measure
AEP	Agri-environmental policy
AES	Agri-environmental scheme
AHQ	Average habitat quality
ART	Agroscope Reckenholz-Tänikon
AWU	Average working unit
BPP	Beneficiary Pays Principle
BTS	Particularly animal-friendly stabling
C	Cost
CAP	Common European Policy
CBD	Convention on Biological Diversity
CC	Cross-compliance
CE	Cost-effectiveness
CED	Cumulative energy demand
CH	Switzerland
CH_4	Methane
CHF	Swiss Francs
CO_2	Carbon dioxide
E	Effectiveness
EAFRD	European Agricultural Fund for Rural Development
EC	European Community
ECA	Ecological Compensation Area
EEC	European Economic Community
eq	equivalents
EU	European Union
EVD	Eidgenössisches Volkswirtschaftsdepartement
FADN	Farm Accountancy Data Network
FDEA	Federal Department of Economic Affairs
FiBL	Research Institute of Organic Agriculture (Forschungsinstitut für biologischen Landbau, FiBL)
FOAG	Federal Office of Agriculture (*Bundesamt für Landwirtschaft, BLW*)
FSO	Federal Statistical Office (*Bundesamt für Statistik, BfS*)
FSS	Farm structure survey
FWU	Family working unit
GAI	Goal attainment index
GATT	General Agreement on Tariffs and Trade
GDP	Gross domestic product
GHG	Greenhouse gas
GMO	Genetically-modified organism
ha	Hectare
ISO	International Organization for Standardization
K	Potassium
kcal	Kilo calories
kCHF	1000 CHF
LCA	Life cycle assessment
LCI	Life cycle inventory

LCIA	Life cycle impact assessment
LP	Linear programming
LU	Livestock unit
LwG	Federal Law on Agriculture *(Landwirtschaftsgesetz)*
MEA	Millennium Ecosystem Assessment
MFA/RLU	Average main forage area per roughage-consuming livestock unit
MIPS	Material Input per Service Unit
N	Nitrogen
N_2O	Nitrous oxide
NH_3	Ammonia
NHG	Natur- und Heimatschutzgesetz
O_2	Oxygen
ÖA	*Ökologischer Ausgleich*
OECD	Organisation for Economic Co-Operation and Development
OFASP	Organic Farming Area Support Payments
ÖQV	*Öko-Qualitätsverordnung*
P	Phosphorus
PE	Public expenditure
PEP	Proof of Ecological Performance
PL	Payment level
PMP	Positive Mathematical Programming
PPP	Polluter Pays Principle
PRTC	Policy-related transaction costs
PU	Policy uptake
RAUS	Livestock with outdoor exercise
RDA	Röhm-Dabbert approach
RGVE	Roughage-consuming livestock units (*Rauhfutterverzehrende Grossvieheinheiten*)
RLU	Roughage-consuming livestock unit
SALCA	Swiss Agriculture Life Cycle Assessment
SILAS	Sector Information and Prognosis System for Swiss Agriculture (*Sektorales Informations- und Prognosesystem für die schweizerische Landwirtschaft*)
SP	Stated preferences
SR	Swiss Jurisdiction *(Schweizer Recht)*
TC	Transaction cost
TEEB	The Economics of Ecosystem and Biodiversity
TEP	Animal husbandry under adverse conditions (*Tierhaltung unter erschwerten Produktionsbedingungen*)
UAA	Utilised agricultural area
UN	United Nations
UNECE	United Nations Economic Commission for Europe
UNFCCC	United Nations Framework Convention on Climate Change
WTO	World Trade Organisation
WTP	Willingness to pay
ZA	*Zentrale Auswertung*, (Swiss ‚FADN equivalent')

1 Introduction

This introduction describes the background and the research gap which this thesis addresses and formulates the principal research question (Section 1.1). Second, the goal and the specific objectives of this thesis are formulated (Section 1.2). Finally, the structure of the document is explained (Section 1.3).

1.1 Problem statement

Organic farming has become a significant alternative farming system in many countries of the European Union (EU) and beyond. In Switzerland, organic farming has a strong institutional foundation (Moschitz and Stolze, 2009) and has been growing more rapidly and strongly than in most other countries (Willer *et al.*, 2008).

Since 1993, the Swiss federal agricultural policy has been providing financial support for organic farming via area payments (Padel and Lampkin, 2007). Like other voluntary agri-environmental programmes, these payments are intended as incentives for farmers to comply with defined production standards (Lampkin and Stolze, 2006). Such payments lead to better environmental performance, as compliance with organic production standards averts negative and provides positive external effects compared to conventional or integrated farming (CRER, 2002). For instance, organic farming is largely not dependent on external inputs. This minimises the resource use of the farming system and limits the nutrient loads in the system, which in turn leads to less overfertilisation and reduced eutrophication risks involving nitrogen and phosphorus (Haas *et al.*, 2001). Less intensive crop management, the ban on synthetic pesticides and the greater reliability on a functioning environment all serve to improve habitat quality for wild animal and plant species and their diversity (Köpke, 2002; Stolze *et al.*, 2000).

However, the environmental evidence stems largely from case studies relating to narrow system boundaries, such as field (Mäder *et al.*, 2002), a rotation (Alföldi *et al.*, 1999) or a farm (O´Riordan and Cobb, 2001). Structural differences between organic and conventional[1] farms (Schader *et al.*, 2008b) and differences in farmers' management skills have largely been disregarded (Alig and Baumgartner, 2009), the latter due to the difficulties capturing them. At the same time, there are often insufficient large-scale environmental monitoring data available, while existing data are imprecise (EEA, 2005). Thus it is difficult to draw general conclusions on the environmental impacts of organic farming.

Against the background of limited public budgets, considerations of cost-effectiveness play a fundamental role in the further development of direct payment schemes as the major agricultural policy in budgetary terms (Swiss Federal Council, 2009). The targeting and tailoring of policies to achieve maximum effectiveness with a given budget is essential (OECD, 2007d). It is therefore necessary to compare both environmental effects and the societal costs of agri-environmental policy measures with each other in order to provide a basis for economically sound policy design (Pearce, 2005).

Agricultural economists hold differing views on the cost-effectiveness of organic farming support payments: On the one hand, von Alvensleben (1998) and Mann (2005a) argue that the organic farming area support payments are not economically sound, as the policy goals could be achieved more efficiently using more flexible and targeted combinations of various agri-environmental measures. The economic rationale behind this argument was introduced by Tinbergen (1956), who theorised that an efficient policy requires at least as many specific instruments as there are specific goals. However, the Tinbergen Rule may not apply fully in this case due to interactions between policies, conflicting goals and the limited determinability of different aspects of environmental performance. Furthermore, the multi-purpose character of organic agriculture could increase its cost-effectiveness due to its potentially lower transaction costs compared to targeted agri-environmental measures (Dabbert *et al.*, 2004).

Despite its high political significance, the question as to whether organic farming can be cost-effective in providing environmental services remains unresolved, since there are no analyses

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[1] Henceforth the term 'conventional farms' is used for all non-organic farms. Non-organic farms cover both conventional farms (< 2 % of total farm in Switzerland, not eligible for direct payments) and integrated farms (> 85 % of total farms in Switzerland, eligible for direct payments).

available at agricultural sector level[2] based on empirical data. Thus, the principal research question addressed by this thesis is: **How cost-effective is organic farming in providing environmental services under the current Swiss agricultural policy scheme?**

Due to the complexity and the multitude of environmental impacts associated with organic agriculture, the subsequent quantitative modelling analysis focuses on a selection of environmental categories. This selection was based on the following criteria: a) the importance of the environmental category in the current policy debate, b) the importance of agriculture for the environmental category, c) the existence of systematic differences between organic and non-organic farming systems, d) the feasibility of modelling the environmental indicators at sector level and e) the availability of comprehensive, quantitative and widely accepted data for Switzerland.

As a result, the environmental impact categories have been limited to a) the use of fossil energy from a life-cycle perspective, b) impacts on biodiversity in terms of habitat quality, and c) eutrophication with nitrogen and phosphorus (see Section 6.3.7 for a detailed description of the selection of impact categories).

1.2 Aim and specific objectives

The overall aim of the thesis is **to compare the cost-effectiveness of organic farming with the cost-effectiveness of individual agri-environmental policies by developing and applying an economic modelling framework at sector level for the Swiss case.**

The specific objectives of the thesis are as follows:

1. To review current knowledge about economic evaluation and environmental impacts of organic farming at an international level as a basis for the development of an analytical framework and research hypotheses.

[2] For the benefit of brevity, the term ‘sector level’ is used with regard to the **Swiss agricultural sector** if not specified otherwise.

2. To design an analytical framework and economic model for analysing the cost-effectiveness of organic farming and other agri-environmental policy measures for the Swiss agricultural sector.

3. To assess the relative environmental impacts of organic farming with respect to fossil energy use, biodiversity and eutrophication with nitrogen and phosphorus.

4. To compare the cost-effectiveness of organic farming with the cost-effectiveness of agri-environmental measures.

1.3 Outline

Following this introduction (**Chapter 1**), which outlines the background, principal research question and objectives of this thesis, the available literature is presented along with some conceptual considerations in three chapters that follow:

Chapter 2 provides a conceptual overview of the economic theory and evaluation of agri-environmental policy instruments, thus establishing a methodological framework for the subsequent analysis. It shows that the Tinbergen Rule does not, in principle, contradict multi-objective policies such as financial support policies for organic farming. **Chapter 3** presents empirical results on the environmental impacts and costs of organic farming from Swiss and international studies. This is a prerequisite for developing a targeted model for analysing the cost-effectiveness of organic farming. In **Chapter 4**, the Swiss agricultural policy framework is reviewed in terms of its overall goals, specific targets, existing instruments and evaluations relevant to this thesis.

Based on the literature and ideas contained in Chapters 1 to 4, some working hypotheses are developed in **Chapter 5**. The core hypothesis is that organic farming is a more costly policy instrument if only single agri-environmental goals are pursued, but that it becomes competitive compared to individual agri-environmental policies if multiple environmental goals are to be achieved simultaneously.

Chapter 6 explains the methodological approach used in investigating the research questions and working hypotheses. The aim of the approach is to ascertain the cost-effectiveness of organic farming and agri-environmental measures by deriving the environmental impacts and public expenditure entailed at sector level. These parameters are generated on the basis of a sector-representative economic farm group model (FARMIS), which is extended by three modules encompassing a) life cycle assessments for fossil energy use, biodiversity, eutrophication, b) public expenditure, including policy-related transaction costs, and c) uptake of agri-environmental policies. To ascertain the costs and effects of organic farming, organic farms are compared to their conventional counterparts in their current state. The cost-effectiveness of agri-environmental measures is derived by calculating the impacts of abolition of the payments for the total sector. The difference between the situation without the payments and the situation with existing payments is interpreted as the additionality of the respective policy measure.

In **Chapter 7**, abatement and provision costs of organic farming and existing agri-environmental measures – namely 'extenso payments', 'payments for less intensive meadows', 'payments for extensive meadows', and the combination of the three measures – are derived and compared with each other. The comparison reveals a slightly lower but still competitive cost-effectiveness of organic farming compared to the combined agri-environmental measures.

Chapter 8 discusses the results of this thesis in a wider context against the methodological limitations of the approach and against existing literature. The results regarding the cost-effectiveness of organic farming show significant environmental benefits at competitive costs when compared with existing individual agri-environmental measures. The approach proved to be a useful tool for evaluating the cost-effectiveness of agri-environmental policies at sector level. However, the model results have only a limited applicability due to the assumptions underlying the model and the comparison of organic agriculture with a limited number of agri-environmental policies.

Chapter 9 draws conclusions within the methodological context and provides policy recommendations on the basis of the results obtained in this thesis. This thesis proved on a theoretical level that the Tinbergen Rule is not a sufficient reason for excluding organic agriculture policy support from a portfolio of agri-environmental policy instruments. Furthermore, this thesis contributes to knowledge, as it designed a framework for assessing the cost-effectiveness of agri-environmental policies. Besides public expenditure and environmental effectiveness, farm-structure, in particular policy uptake is identified as a major determinant of cost-effectiveness. The model FARMIS was expanded with a) life cycle assessment data for energy use, biodiversity and eutrophication with nitrogen and phosphorus, b) the calculation of public expenditure including policy-related transaction costs and c) an adaptation of the Röhm-Dabbert-Approach (RDA) for modelling the uptake of agri-environmental policies. The expanded model was applied in this study for assessing organic agriculture for the first time in both economic and ecological terms at sector level. Sector-representative figures for abatement cost were calculated for both organic farming and relevant agri-environmental policies in Switzerland. The results revealed a comparable cost-effectiveness of organic agriculture and the combination of agri-environmental measures. There is a large potential for further applications of the expanded FARMIS model.

2 Theoretical basis of the economic evaluation of agri-environmental policy

This chapter provides an overview of the economic evaluation of agri-environmental policy (AEP). First, the economic foundations of AEP evaluation are presented (Section 2.1). Second, the dimensions and economic concepts of evaluation are reviewed and principal instruments of AEP analysed qualitatively (Section 2.2). The conclusion section summarises the most important outcomes from the previous sections (Section 2.3), thus providing the basis for the structure of costs and benefits in the review of effects and costs of organic farming in Chapter 3.

This chapter introduces the terms agri-environmental policy (AEP), (agri-environmental) policy instruments, agri-environmental schemes (AES) and agri-environmental measures (AEM). These terms were defined in 1992 in the Common Agricultural Policy (CAP) with the introduction of the Council Regulation (EEC) No 2078/92[3] and its successor Regulations on rural development[4]. Their usage within this thesis is clarified briefly in the following paragraphs.

The aim of **'agri-environmental policy'** is to address agri-environmental problems by employing different types of **policy instruments**. These may be regulatory (e.g. command-and-control measures), economic (e.g. taxes or quotas), communicative (e.g. information campaigns, research projects), or a mix of these types (Horan and Shortle, 2001; Pearce, 2005; Tuson and Lampkin, 2007). Following Buller (2000), the term '**agri-environmental scheme**' – in contrast to the broader terms 'policy instrument' and 'agri-environmental policy' – refers to a specific type of instrument, which is widespread in both the EU Member States and in Switzerland. According to Frieder *et al.* (2004) and EC (2005) AES are defined as voluntary, agricultural area-related policy instruments which, on the one hand, impose a set of management restrictions on the farmer and on the other compensate him or her for the costs

[3] Council Regulation (EEC) No 2078/92 of 30 June 1992 on agricultural production methods compatible with the requirements of the protection of the environment and the maintenance of the countryside

[4] Currently: Council Regulation (EC) No 1698/2005 of September 2005 on support for rural development by the European Agricultural Fund for Rural Development (EAFRD)

arising. Thus AES include both a regulatory and a financial component. Commonly, agri-environmental schemes consist of different specific '**agri-environmental measures'**, which address different problems and are specified in the EU AEP regulations.

2.1 Economic theory as a basis for agri-environmental policy

This thesis draws on both the theory of ecological economics on the one hand and environmental and resource economics on the other. Resource economics is based mainly on positive utility economics, while environmental economics has the normative character of neo-classical welfare economics (Mankiw, 1998). Ecological economics evolved from the 1970s onwards, prompted by debate about the finiteness of natural resources and by the severe oil crisis (Constanza *et al.*, 1998).

Since the early 19th century, natural resources became more unimportant in classical economics. The classical economic theories recognised natural resources as a whole, in addition to the production factor 'labour', as the basis of economic production (Smith, 1776). However, theories of economic value replaced this view by focussing on commodities (Faucheux and Noël, 1995). According to Ricardo's definition, the value of a commodity is a function of its scarcity and the amount of labour needed to create it. This means that the economic value of natural resources is zero (Ricardo, 1821). Natural resources, in this view, do not fulfil the criteria for economic goods because they a) are not scarce, b) they cannot be produced industrially, and c) they are not exchangeable (Walras, 1952). Therefore, natural resources are excluded from the group of economic goods and declared to be 'free goods' (Faucheux and Noël, 1995). It should be stressed here, however, that land, being both a natural resource and a production factor, is excluded from this definition, either in classical or in neo-classical economics. As land is scarce, it has a price and can be traded on factor markets.

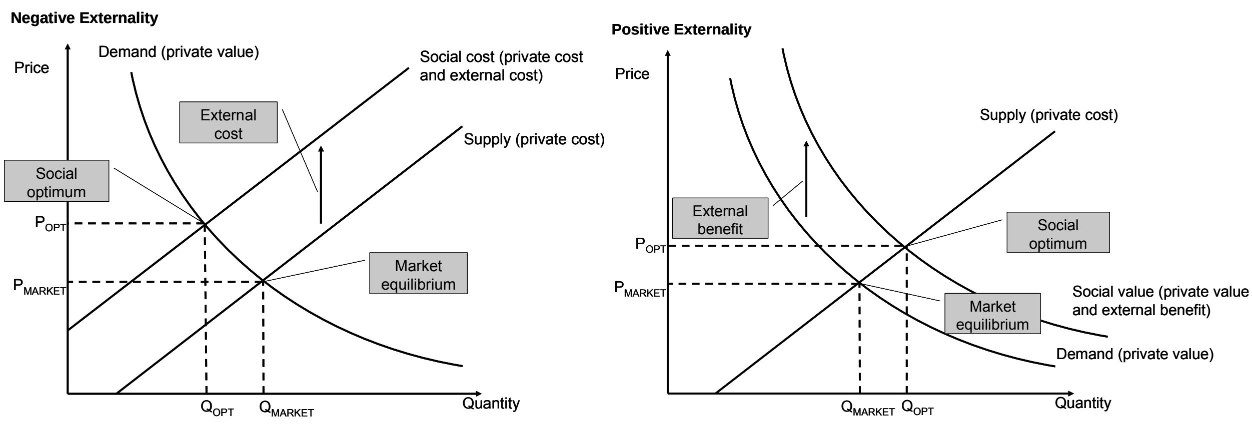
In neo-classical economic theory, this view of natural resources came to be perceived as too narrow, as the analysis of environmental problems came increasingly into focus during the 20^{th} century. In a broadening of the neo-classical view, three fundamental components were introduced through environmental economics:

- **Natural resources:** Resources which cannot be produced by man. Natural resources can be subdivided into renewable resources, such as timber, and non-renewable resources, such as coal and oil (Hotelling, 1931).

- **Externalities:** Interdependencies exist among economic actors which influence their objective functions without an unsolicited exchange (Pigou, 1932). Only if a) property rights are well defined, b) people act rationally, and c) transaction costs are minimal will individual bargaining solve the problem of externalities (Coase, 1960).

- **Collective goods:** Collective or public goods are consumed jointly with others (Samuelson, 1954). Public goods are characterised by non-rivalry and non-excludability. There are also impure public goods, which fulfil these criteria only partly (Buchanan, 1968).

Figure 1 illustrates the presence of a negative externality (left graph) and a positive externality (right graph) on a single-commodity market. In the case of a negative externality, *i.e.* if the production of a commodity involves societal costs which are not taken into account by the market agents, the market equilibrium leads to a higher output of the commodity than is socially desirable. In the case of a positive externality, *i.e.* if the production of a commodity leads to external benefits, the market mechanism leads to a lower commodity output than is socially desirable. While negative externalities of agriculture include, for instance, the pollution of ground water with nitrates, positive externalities cover landscape maintenance or the positive impacts of agriculture on species diversity (OECD, 2001a).



Source: Samuelson and Nordhaus (2007)

Figure 1 Representations of positive and negative externalities in a single commodity market

The consequence of including natural resources, external effects, and collective goods in the economic framework is that markets may fail to generate an efficient solution. This implies the need for market interventions. In fact, it was this realisation that triggered the birth of environmental policy and – in the context of agriculture – of agri-environmental policy (Hampicke, 1992). AEP offers a set of economic instruments (taxes, subsidies, tradable emission permits), non-economic instruments (laws, bans, bondages, norms), communicative policies (information campaigns, extension) and combinations of these three, such as agri-environmental schemes or action plans (Lampkin *et al.*, 2008) to compensate for market failure (Bateman, 1994; Faucheux and Noël, 1995; Pearce, 1999).

The concepts of producer and consumer surplus are essential for the economic evaluation of such policy measures. Producer surplus refers to the amount a seller is paid for a good minus the seller's cost of providing it. The producer surplus can be graphically derived, e.g. in Figure 1, as the area above the supply courve and below the dashed horizontal price line. The consumer surplus is the difference between the price a consumer is willing to pay and for a product and the price of a product. Graphically, the consumer surplus can be derived as the area below the demand courve and above the horizontal dashed price line (Figure 1).

Contrary to neo-classical environmental economics, ecological economics goes further in questioning the fundamental assumptions of economic theory and widens the approach into a complex multi-disciplinary framework (Ehrlich, 2008). The concept of ecological economics is based on four key elements (Constanza *et al.*, 1998):

1. The idea of the earth being a closed thermodynamic and non-growing system. Economy represents a sub-system of the ecosystem. This implies the existence of limited biophysical resource flows which go from the ecosystem through the economic system and finally back to the ecosystem.

2. The future model of a sustainable society with high quality of life for all inhabitants (humans and other species), within the limits mentioned above.

3. The acknowledgement of the fact that the analysis of complex systems faces both spatial and time-related uncertainties. Some processes are irreversible and therefore require a precautionary approach.

4. The need for proactive rather than reactive institutions. The outcome should be simple, flexible and feasible policy strategies, based on deep knowledge of the systems and acknowledging the fundamental uncertainties. This is the basis for sustainable policy.

Thus, from an ecological economist's point of view, Norgaard (1985) argues that expansions of the neo-classical theory are essential:

> '[...] *The basic assumptions of the neo-classical model do not fit the natural world. The model assumes that resources are divisible and can be owned. It acknowledges neither relationships between resources in their natural environments nor environmental systems overall. It assumes that both the economic and the environmental system can operate along a continuum of equilibrium positions and move freely back and forth between these positions. Markets fail to allocate environmental services efficiently because environmental systems are not divisible, because environmental systems almost never reach equilibrium positions, and because changes are frequently irreversible.'*

Consequently, ecological economics analyses three generic problems: the allocation problem, the distribution problem and the problem of scale. Both allocation and distribution are a component of standard economic theory. 'Scale' is a central concept within ecological economics and refers to the fact that the economic system is a part of a natural system, the Earth. Therefore, unlimited growth, as assumed by neo-classical economics, is impossible. Growth, measured economically as GDP, does not become a goal in itself, but rather a burden that leads us steadily closer to absolute limits. Usually, the resource use per head is used as an indicator for scale (Constanza, 1980). These types of indicators have attracted increasing attention during the last few years, driving forward concepts such as the ecological footprint (Wackernagel *et al.*, 1999), life cycle assessment (LCA) (Heijungs *et al.*, 1992) and material input per service unit (MIPS) (Schmidt-Bleek, 1994).

The existence of agri-environmental policies can be rationalised by two different economic theories. On the one hand, agri-environmental policy can be explained by public choice theory, which reasons that decision makers and parties act rationally by trying to maximise their votes. Different stakeholder groups seek to generate support for policies and organise voters. In this sense, the fiscal unimportance of agri-environmental policy may be traced back to the strength of farmers' associations, who favour market support policies rather than agri-environmental payments (Buchanan, 1972; Mann, 2002b; Olson, 1965).

On the other hand, public finance theory states that public policy is brought about by acknowledging the existence of public goods and external effects (Faucheux and Noël, 1995; Musgrave, 1959). Agri-environmental policies are issued to internalise external effects, *i.e.* to compensate for market failure (Pigou, 1932). Brandes *et al.* (1997) argue that not only individuals but also the government has to decide rationally how to spend existing resources, *i.e.* the agricultural budget. A rational decision maker, being aware of limited resources, will thus spend the available budget on efficient policies, which make it possible either to pursue policy goals with the least amount of budget spending or to achieve the goals to a maximum level within a given budget (Samuelson and Nordhaus, 2007).

Both public choice and public finance theory have their justification for analysing policies. However, public finance theory constitutes the main foundation for this thesis, as the thesis' primary aim is to analyse whether three externalities of agriculture can be compensated for by the support of organic agriculture in a cost-effective way.

2.2 Evaluating agri-environmental policies economically

This section first outlines dimensions of economic evaluation of agri-environmental policy (AEP) (Section 2.2.1). Second, analytical concepts of economic AEP evaluation are described (Section 2.2.2). After that, the welfare impacts of the principal instruments of AEP are analysed qualitatively (Section 2.2.3).

2.2.1 Dimensions of economic AEP evaluation

Since the early 1990s, both in Switzerland and the EU, direct payments and agri-environmental programmes have increasingly replaced market and price support schemes as the primary instrument for supporting agriculture (BLW, 2006; Bruckmeier and Ehlert, 1999). This change has had a significant influence on farming practices in Europe (OECD, 2004).

Evaluations of agri-environmental policy are an important means to improve programme operations, to adapt them according to changing environmental, economic, social, and political parameters, and thus to influence the way agriculture is practised (Jones, 2004). The notion of 'evaluation' has been defined in many different ways, depending on the context of the evaluation in question. Lampkin *et al.* (2008, Section A2-2.1) compares a number of relevant general definitions and summarises his findings as follows:

> *'In summary, policy evaluation involves the systematic gathering of information and assessment of a programme according to specific criteria in order to make judgements about the value of the programme, thus reducing uncertainty in decision-making about future actions. The assessment of value may relate to the goals of more than one specific interest group, including policy-makers, beneficiaries and third parties, and they may fulfil a range of purposes, from financial control and accountability to intervention improvement and knowledge advancement.'*

While some authors restrict the notion of 'evaluation' to *ex-post* assessments (Vedung, 2000), most authors include *ex-ante* or mid-term assessments as well (EC, 1999a; Pearce, 2005). Stockmann (2004) distinguishes between *ex-ante*, ongoing and *ex-post* perspectives. While *ex-ante* evaluations have a formative character and are mainly used when formulating policies in order to improve future policies, the summative *ex-post* evaluations are carried out after the policy has been implemented, for purposes of explanation and generalisation. Evaluations of ongoing policies can be both formative and summative (Table 1).

Table 1 Dimensions of evaluation research

Stages of the programme process	Analysis perspective	Perception interests	Evaluation concept
Formulating the programme/planning stage	*Ex-ante*	Analysis for policy Science for action	Reformative/formative: active designing, process orientated, constructive => improvement of future policy
Implementation stage	Ongoing	Both planning and impact stage perception interests possible	Formative and summative possible
Impact stage	*Ex-post*	Analysis for policy Science for knowledge	Summative, making up the balance, result orientated => explanation and generalisation

Source: Stockmann (2004)

Existing policy evaluation guidelines and frameworks highlight the multi-layered character of an evaluation (Weiss, 1998). Bussmann *et al.* (1997) state that the objects of evaluation of a policy concept are policy design (administration programme), institutional arrangement, action plans for implementation, outputs, impacts, outcomes and results of the policy. However, according to Drummond (2005) economic evaluation needs to deal principally with inputs and outputs, *i.e.* costs and effects of the evaluation subject. Due to the scarcity of resources, choices have to be made. These are made on the basis of multiple criteria, sometimes explicit but often implicit as well. Thus economic evaluation can be defined as the '*comparative analysis of alternative courses of action in terms of both their costs and consequences*' (Drummond, 2005, p. 9).

In recent years, there have been major efforts to progress methodologically in practical economic evaluation of agri-environmental policy (Cahill and Moreddu, 2004; EC, 1999b; OECD, 2001b). Databases with ecological indicators which can be used for economic analysis have been established and enlarged (EEA, 2005). Nevertheless, the results of most evaluations do not meet the criteria necessary for upscaling to regional or sector level, such as the inclusion of an extrapolation or scaling procedure, or the expression of impacts according to several reference units (Herzog, 2005; Payraudeau and van der Werf, 2005).

2.2.2 Economic concepts for AEP evaluation

In fact, economic evaluation concentrates on contrasting the costs of the programme with its effectiveness, *i.e.* its outcomes, results and impacts, or benefits. There are three basic economic concepts for comparing costs and effects (benefits) (Drummond, 2005; Pearce, 2005): Cost-Benefit Analysis (CBA), Cost-Effectiveness Analysis (CEA) and Multi-Criteria Analysis (MCA). These concepts are explained in the following sections.

Cost-Benefit Analysis

A CBA necessarily involves the quantification of costs and benefits (units and weights) in monetary terms. Money as a unit reflects the strength of individuals' preferences, which are measured by the willingness to pay (WTP). In its simplest form the CBA equation appears thus:

$$S_i = \sum_{j,n} WTP_{ijn} - C_i \quad \forall i \tag{1}$$

where S_i is the overall score of the i^{th} option, WTP_{ijn} is the willingness to pay for the i^{th} option, for the j^{th} attribute or criterion and the nth individual, and C_i is the cost of the i^{th} policy option (Pearce, 2005).

Economists have developed various techniques to place a value on non-commodity outputs (NCOs), public goods and cultural amenities consistent with the microeconomic valuation of marketed goods; *i.e.* based on individual preferences (OECD, 2002). These techniques are based upon either observed behaviour (revealed preferences, RP) or stated preferences (SP) in surveys dealing with the public good. Furthermore, direct and indirect approaches are distinguished. While direct methods deal straightforwardly with the non-commodity concerned, indirect methods derive values for the non-commodity from related aspects or commodities (Table 2). For a detailed description of these approaches, see Christie *et al.* (2008) and Navrud (2000).

Table 2 Classification of approaches to measure willingness to pay (WTP)

	Indirect	Direct
	Techniques based on individual preferences	
Revealed preferences	Household Production Function Approach: - Travel Cost method - Averting Cost method Hedonic Price analysis	Simulated markets Market prices Replacement cost
Stated preferences	Contingent Ranking Choice Experiments/ Conjoint Analysis	Contingent Valuation Method
	Techniques based on collective preferences	
Revealed preferences	Implicit Valuation	
Stated preferences	Citizens' Juries Delphi Method Market stall Valuation workshop Expert Valuation Method Budget game	Multi-Criteria Analysis[5]

Source: Schader *et al.* (2009a) modified from Navrud (2000)

Cost-Effectiveness Analysis

CEA compares a single indicator of effectiveness E_i to cost C_i. In this way a cost-effectiveness ratio (CER_i) is obtained according to Equation 2 (Pearce, 2005):

$$CER_i = \frac{E_i}{C_i} \quad \forall i \qquad (2)$$

Whilst E_i is measured in an environmental unit, C_i is measured in monetary units. Due to this difference in units of E_i and C_i, this ratio does not reveal whether the benefits of the scheme exceed its costs. Hence it cannot be judged whether the scheme is worth conducting. This question could only be answered if C and E were measured in the same units (as for the CBA above).

In avoiding the monetisation of benefits, the CEA cannot help in deciding whether a policy (option) should be chosen or not, it can help only in comparing different options. Furthermore, CEA necessitates aggregating effects in a non-monetary unit, if more than one effect

[5] Being a major appraisal tool for policy evaluation in itself (see page 16), multi-criteria analysis can also be used to elicit WTP statements within a CBA setting.

has to be related to the costs. This is particularly relevant for the research question, as it stipulates that more than one environmental category has to be taken into account.

Multi-Criteria Analysis

The MCA is similar to the CEA in many respects but involves multiple indicators (criteria) for effectiveness, however, cost should always be chosen as an indicator in the MCA (Pearce, 2005). Finally, the indicators are weighted according to their importance. In theory, the weighting of criteria should be done in accordance with public preferences, in practice, the MCA tends to work with experts' knowledge. The equation for the final score for an option is:

$$S_i = \sum_j w_j \; CR_{ij} \qquad \forall i \tag{3}$$

where S_i is the score of the i^{th} option, w_j is the weight of the j^{th} criterion and CR_{ij} is the score for criterion j of the i^{th} option.

If criteria are weighted on the basis of public preferences, the monetisation of criteria, as in the CBA, can be bypassed. Nevertheless, an assessment of the relative importance of the different criteria is necessary unless all criteria are accorded the same weight. These weights can be determined on the basis of either individual or collective preferences using the methods described in Table 2. Further information on weighting procedures is provided by Dodgson *et al.* (2001) and Lampkin *et al.* (2008).

Comparison of economic evaluation tools

All three techniques, regardless of their inclusion of a monetary valuation, have advantages and disadvantages in practice. There are general objections to the use of WTP techniques to derive economic values for environmental goods. Potential problems and biases that can occur are (Fischer *et al.*, 2003; Hampicke, 2003; Hanley *et al.*, 1995; Randall, 2002):

- **Information bias:** Individuals may not have enough information to state their WTP. This applies especially to very complex questions, e.g. multifunctional outputs, environmental benefits or rural amenities or other non-commodity outputs (OECD, 2001a) of agriculture.

- **Strategic bias:** Respondents may purposely give incorrect answers because they hope to achieve other aims (e.g. free-rider behaviour).
- **Interviewer bias, starting point bias:** As in any other method of empirical social research the interviewer or the formulation of the questionnaire might influence the respondent.
- **Hypothetical bias:** A hypothetical market is not comparable with a real one. Respondents are not used to valuating non-commodities or public goods. Some might even refuse to do so. Others might simply state an arbitrary amount of money as their WTP.
- **Embedding effect:** As a special case of hypothetical bias, the embedding effect leads to the fact that the respondents value the quality, but not the quantity, of a good. In the case of biodiversity, some studies report that the same respondents stated a higher WTP for the conservation of a single species than for a set of species.
- **Warm glow effect:** When asked for issues with a moral value, the interviewees state sums which they personally are willing to pay for charity purposes. This results in a valuation which is not according to the micro-economic theory, where individuals think of personal welfare maximisation only.

As a further problem in the context of public goods, Hampicke (2003) emphasises the non-microeconomic thinking of the respondents (which also causes the warm glow effect). It is the exception rather than the rule that respondents are able to state their exact demand for a public good and, even if they could, public goods are often indivisible. For instance, if a respondent was able to state the exact amount of fresh air he or she demands, it is most likely impossible to provide this exact amount (Ahlheim and Frör, 2003). Therefore, the microeconomic idea of a marginal WTP differs significantly from reality concerning environmental public goods. Furthermore, ethical values and normative conceptions are attached to public goods, making valuation in a strictly microeconomic sense difficult. This critique is shared by Bateman (1994), who questions the appropriateness of individual preferences as a basis for judging the environmental and other values associated with a particular site or environmental benefit. He argues that the assumption that values can be measured on the basis of current income distributions may be wrong. Mann (2002a) brings forward the argument of merit goods in this context, arguing that expert-based valuations could be justified when complex issues are concerned.

Many researchers stress that if SP studies are conducted correctly these biases could be avoided (Christie and Azevedo, 2008; Hampicke, 2003; Hanemann, 1994; Kontoleon *et al.*, 2002), particularly if the information bias is addressed (Christie *et al.*, 2006). Nevertheless, the above list of potential biases and general objections of stated-preference studies demonstrates that these studies have to be conducted very accurately and that their results need to be interpreted with special care. Pearce (2005) strongly advocates that the cost-benefit analysis (CBA) approach is used and thus sees the expression of all external costs and benefits in monetary terms as a necessity for '*rational appraisal of agri-environmental policy*'.

Against the objectives of this study a complete valuation of the benefits of the policy measures and farming systems in Switzerland, which would be necessary to conduct a full CBA, is not undertaken in this thesis, because the question whether the costs of the agri-environmental payments exceed their benefits is of secondary importance. The primary question in this thesis is not whether or not to support governmentally, but how to support with agri-environmental measures. Furthermore, former CBA-studies indicate that agri-environmental payments show very high benefit-cost ratios. Therefore, '*The degree of exaggeration would have to be substantial to conclude that agri-environmental schemes are anything other than good social investments*' (Pearce, 2005). Hence, the added value of a CBA would not help to answer the research question of this study but would include the valuation as a further normative element in the analysis.

As an alternative, multi-criteria analysis (MCA) seems to be more appropriate than cost-effectiveness analysis (CEA) for policies with multiple effects (EC, 1999b), however, MCA is less appropriate, if one wants to compare cost-effectiveness with respect to individual criteria. The advantage of CEA, on the other hand, is that it creates a direct relation between effectiveness in relation to a criterion and costs, while in MCA costs are integrated as only one criterion among many. Therefore, a CEA seems to be the most favourable option for each of the criteria without aggregating the different effects. An ultimate ranking of policy instruments is not possible, however, without a weighting procedure, unless the indicators are influenced homogenously by the different policies.

2.2.3 Qualitative analysis of the applicability of agri-environmental policy instruments

Environmental policy instruments are commonly evaluated comparatively against the criteria 'environmental effectiveness' and 'economic efficiency' (OECD, 2004). In addition, political economic and societal framework conditions as well as the nature of the environmental problem influence the applicability of specific instruments (Horan and Shortle, 2001).

This section provides, first, an overview of the rationale behind the most common environmental policy tools in the agricultural context. Second, it analyses beneficial and detrimental preconditions for the application of the instruments. Third, policy mixes and multi-objective policies are discussed, and the implications of the Tinbergen Rule for multi-objective policies are highlighted using a simple linear programming model for the case of organic farming.

Environmental effectiveness and economic efficiency of AEP instruments

Environmental policy offers a wide variety of policy instruments to deal with environmental problems. Depending on the type of environmental problem and the specific circumstances, certain policy instruments and designs are more or less favourable from an economic standpoint. An overview of commonly used agri-environmental instruments, based on Pearce (2005) and Mickwitz (2003), is provided in Table 3.

The most apparent environmental policy instrument is a **standard regulation** which either bans the use of certain inputs or management practices or prescribes the use of precautionary instruments following the 'polluter pays principle' (PPP). An example of this type of instrument would be a quantity restriction on a particularly toxic pesticide. Such a regulation leads to higher costs for farmers due to a loss of producer surplus and/or a loss in output. The cost to taxpayers are restricted to administration, monitoring and enforcement costs, commonly referred to as policy-related transaction costs (PRTC) (OECD, 2007b). There is no consumer-borne cost if product prices remain unaffected.

Regulations are generally highly effective, provided they are properly enforced. From an economic viewpoint, however, a regulation will be inefficient if marginal abatement costs vary among farmers, since farmers who have high abatement costs would rather compensate others for being allowed to bypass the regulation (Coase, 1960).

From a neo-classical economics perspective, efficient policy requires employing the market mechanism. **Environmental taxes** (Pigou, 1932), e.g. a nitrogen-fertiliser tax, are more efficient because farmers who are able to make profits from applying fertilisers despite taxation are still permitted to do so. Thus despite pursuing the polluter pays principle, the total loss of producer surplus will – in theory – be smaller compared to a fixed-quantity regulation. The cost to taxpayers comprises only the transaction costs of maintaining the scheme. However, taxes are relatively cheap to administer due to the large quantities of involved units (hectares, tonnes of products), which entail low transaction costs per environmental effect (Rørstad, 2007). In principle, taxes could also be output-related, e.g. taxing the quantity of an unintended output. But, input-related taxes are more common in the agri-environmental context, as environmental effects are often related to purchased inputs such as mineral fertilisers or pesticides (Daugbjerg and Pedersen, 2004).

Cap-and-trade schemes, also referred to as **tradable quotas**, are primarily related to taxes. A market is created for trading in certificates. Prices for the certificates are influenced due to the capping mechanism which creates a scarcity of permits. Similar to taxing, a cap-and-trade scheme is economically efficient in the sense that those producers with low abatement costs will sell their quota to producers with higher abatement costs. Thus, theoretically all farmers will continue to extend production until their farm-specific marginal production costs exceed their marginal revenues, *i.e.* until the marginal profits of all farms are zero.

The generic difference to taxes, however, is that the price remains unregulated, whereas the quantity is regulated (Hepburn, 2007). Quotas can be either grandfathered[6] according to the beneficiary-pays-principle (BPP), or auctioned, according to the polluter pays principle (PPP). If the quotas are grandfathered, the producer surplus will stay unaffected; if auctioned, the producer surplus will sink. But as those producers who have relatively high abatement costs will purchase more quota, the quota system is still economically efficient, provided the quota is tradable (Cramton and Kerr, 2002).

[6] That is quotas are distributed for free to the economic units concerned

Table 3 Typology of costs and benefits of agri-environmental policy instruments

Type	Instrument	Principle	Costs to farmers	Costs to taxpayers	Costs to consumers	Benefits
Regulatory	Command-and-control regulation	PPP	Loss of producer surplus via cost increases	Administration, monitoring and enforcement costs	Loss of consumer surplus, if market prices are affected	Environmental, safety and health. Use and non-use values
Economic incentives	Environmental taxes	PPP	Loss of producer surplus due to cost increases	Administration monitoring and enforcement costs	Loss of consumer surplus, if (part of) cost increases shifted forward	As above, but with potential for re-cycling tax revenues to environmental uses
	Auctioned tradable quotas	PPP	Loss of producer surplus via cost increases	Tax revenues, administration, monitoring and enforcement costs	Loss of consumer surplus, if (part of) cost increases are shifted forward	As above, plus the re-cycling of quota revenue is possible
	Grandfathered tradable quotas	BPP	Depends on the tightness of the cap and the reactions to the quota prices	Quota revenues, administration, monitoring and enforcement costs	None, however, consumers are expropriated because a former public good has become privately owned	As above, but no revenue re-cycling possible
	Environmental auctions	BPP	None, but depending on the context of the auction	Cost for service, Administration, monitoring and enforcement costs	None, but depending on the rationale of the auction	Environmental, safety and health. Use and non-use values
Commun-icative	Education, extension, research, labelling	none	None or small gains due to increased market transparency	Cost of provision of information and research, if publicly funded	None or small gains due to increased market transparency	Indirect effects through changes in farm practice and consumer behaviour
Mixed: Regulatory/economic incentives	Agri-environmental schemes and measures	BPP	Zero, if payments offset output losses. Practically negative due to overcompensation of some farms	Public cost of payments Administration, monitoring and enforcement costs	None unless output reductions affect prices	Direct environ-mental effects of AESs. Use and non-use values
	Cross-compliance (CC)	PPP/ BPP (de-pending on the baseline)	Loss of producer surplus (for some elements of CC) compared to baseline of unrestricted subsidy	Administration, monitoring and enforcement costs. Additional or less cost depending on the baseline	None, unless output reductions affect prices	Direct environ-mental effects of compliance obligations. Use and non-use values
	Community-based schemes	BPP	Zero, if payments offset output losses. Practically positive due to overcompensation of some farms	Cost of funding initiatives plus administration, monitoring and enforcement costs	None unless output reductions affect prices	Direct and indirect environmental effects. Use and non-use values

Source: based on Pearce (2005), adapted. PPP: Polluter pays principle, BPP: Beneficiary pays principle

Normally, the consumer surplus remains unaffected by grandfathered tradable quotas. However, there will be a theoretical expropriation of the citizens if quotas of a natural resource are grandfathered to an economic sector. If the quota is auctioned, quota-purchasing farms shift forward parts of their cost increases.

Environmental auctions are discussed being an efficient and effective solution in the agri-environmental context on a smaller scale. Since the contractor is responsible for completing the task, there is a success-oriented element underlying this instrument. Auctions also have economic appeal due to their ability to provide an efficient solution at moderate transaction costs. An institutional prerequisite is, however, that enough bidders are on the market and collusion is prevented (Cason and Gangadharan, 2005).

Communicative policies follow a very different approach. Rather than paying producers for environmentally-friendly production or penalising environmentally-harmful production methods, this type of instrument leads indirectly to higher uptake levels of agri-environmental schemes on the production side. If applied on the demand side, improved market transparency and potentially higher demands for environmentally-friendly produce can be achieved. Both consumer and producer surplus can be assumed to be zero. However, if market transparency is improved, gains in consumer and producer surplus may occur. Costs arise for the taxpayer from the implementation and administration of the measure. In Switzerland, measures target ‘production and sales’, e.g. sales promotion of sustainable or regional products (Chapter 4). Furthermore, several measures contained in organic action plans (Lampkin and Stolze, 2006) can be classified as information policies, including research and extension.

Agri-environmental schemes (AES) and measures (AEM) are most frequently used in Swiss and EU agri-environmental policy to address both positive and negative externalities of agricultural production. Conceptually, agri-environmental schemes are a blend of a regulatory instrument with an economic incentive. Thus, AEM follow the BPP. According to Frieder *et al.* (2004) agri-environmental measures in the EU are characterised as voluntary measures related to a specific area. AES compensate farmers for yield and income losses[7] as well as

[7] In the EU, an additional 20 % additional payment could be used as a further incentive to boost the uptake of AES. After the mid-term review of the Rural Development Plans this incentive permission was replaced by policy-measure-specific compensation for farm-level transaction costs. In Switzerland, according to Mann (2003a), by trend, all agri-environmental measures are overcompensated.

higher production costs. Furthermore, farms are obliged to maintain the relevant management practices over a defined period of time (EC, 2005). AEM relate to either the whole farm (e.g. for organic farming area support payments, OFASP), the farm branch (e.g. for extensive production of grains), a certain production activity (e.g. high-stem fruit trees), or single fields, plots or elements (e.g. hedges) (Frieder *et al.*, 2004).

In Switzerland, these characteristics apply also to ecological direct payments, although the mandatory time period and the size of surplus incentive payments involved are different. Thus the Swiss ecological direct payments can be classified overall as an agri-environmental scheme.

Agri-environmental schemes have a positive effect on the producer surplus since farmers will not take up agri-environmental measures as long as the benefits do not exceed the costs for individual farmers. However, since the payment levels are not adapted to each farm, there will be a considerable producer surplus for many farmers, due to overcompensation. There will be no effect on consumer surplus in relation to purchased products, if consumer prices are unaffected by the output reductions. Economically, this is fair to assume as in the current market situation domestic output reductions will be off-set by imported products of at least the same price.

Cross-compliance regulations were introduced as a policy instrument in Switzerland in 1998 and in the European Union in 2005. In contrast to the voluntary agri-environmental schemes, cross-compliance rules are obligatory. In Switzerland, farms which do not comply with the Proof of Ecological Performance (PEP) are not eligible to receive direct payments. In the EU, cross-compliance rules can be specified at member state or regional level. The evaluation of this instrument depends, however, on the baseline with which the cross-compliance regulation is compared. If this baseline situation is compared with a situation involving direct payments without cross-compliance, this instrument follows the PPP, as cross-compliance urges producers to change their farm management practices. Thus cross-compliance makes farmers incur higher production costs. But if the instrument is compared with a situation without any other policies, the cross-compliance follows the BPP, since the payments linked to the cross-compliance would have to be taken into account.

Thus the question arises whether a second-best (Lipsey and Lancaster, 1956) or a first-best solution is sought. While Pearce (2005) argues that the policy should always be compared with a situation without any other policy, Henrichsmeyer and Witzke (1994b) argue that in the context of agricultural policy it is not always useful to look for a first-best solution when evaluating a policy reform. If the existence of other policies is inevitable, e.g. for political reasons, a policy reform leading to a second-best solution is preferable. This consideration also applies in the context of AES. Since many AES would not qualify as a first-best solution, they constituted a viable second-best solution when they were introduced (Henrichsmeyer and Witzke, 1994b).

Community-based schemes follow a bottom-up approach, the idea being to fund local initiatives aimed at pursuing policy goals at regional or local level. Such measures do exist both in Switzerland and the EU, but they constitute a negligible budget share relative to total agri-environmental expenditure. From a welfare-economic perspective, these schemes are comparable to standard agri-environmental schemes. However, since they operate at a local level, including discussions and negotiations with farmers, these programmes tend to have a lower transfer efficiency than standard agri-environmental schemes (Vatn, 2002).

With regard to **benefits**, most of the instruments give rise to societal impacts in the form of environmental safety, health or aesthetic and cultural values. Additional monetary benefits are generated by taxes and auctioned quotas for the government. Apart from achieving the imposed market correction, these revenues can be spent on paying PRTC and direct payments on further environmental policies. If licences are grandfathered instead of being auctioned, firms may raise an indicator variable such as energy use prior to the application of the instrument in order to receive more licences. Furthermore, applying the PPP instead of the BPP seems appropriate for most environmental problems in terms of equity (Hepburn, 2006). An auction will direct the attention of the agent implementing the measure, e.g. the farmer, to the environmental problem. The instruments which mix regulatory and economic elements have various different benefits. While cross-compliance and standard AES and AEM work on a very broad level, community-based schemes frequently take into account local circumstances and work in a result-oriented way. This may often lead to better outcomes compared to broad-brush AES and AEM (Eggers *et al.*, 2004).

Favourable and detrimental settings for the application of instruments

Economic instruments, *i.e.* taxes or **tradable quotas**, are especially useful in situations in which the regulator does not have sufficient information on the abatement costs of farms or if large-scale environmental problems need to be addressed. The larger the geographic scale of the problem, the more diverse the circumstances. While a regulation would have to be adapted for each of the regional conditions, economic instruments are capable of producing an efficient solution by employing the market mechanism. However, only quantitative policy targets can be addressed through economic instruments (Cramton and Kerr, 2002).

Taxes are preferable in cases where there is no precisely defined quantitative policy target since there is empirical evidence that taxes deliver the same effects as other instruments more efficiently, *i.e.* at a lower administrative cost (Hepburn, 2006). However, taxes are less preferable in cases where farmers, are net subsidy receivers. This is because taxes imply a further financial burden for farms which needs to be compensated for if the effectiveness of the subsidising policies should be maintained. Otherwise the tax would interfere with other non-environmental policy goals. However, if quotas are auctioned and not grandfathered, the same problem as with the tax occurs.

Command-and-control regulations are more appropriate when the regulator has good quality information, when the risk of government failure is low, and when the desired goal is best achieved by imposing similar requirements upon different firms and individuals. Moreover, command-and control regulations perform well when qualitative policy targets, such as a complete ban on a substance, need to be achieved.

Environmental auctions are discussed for situations where large gaps in knowledge among the agents exist and/or complex causal region-specific circumstances need to be tackled. Auctions require viable entrepreneurial behaviour on the part of farmers and make it possible to address problems in situations where farmers need to cooperate or network (Cason and Gangadharan, 2005).

Information policies are applicable in many circumstances. Large-scale environmental problems can be addressed effectively by information policies, because the larger the scale of the problem, the higher the impact per unit cost of the information campaign, labelling effort, or research effort. Information policies should be implemented if there is a high level of certainty about impact mechanisms and if the risk of government failure is low. Large knowledge gaps and complex causal relations can be addressed by information policies, and cooperation among farmers can be facilitated by them. In particular, a low level of policy uptake can be attributed to an information gap or to ideological constraints, so that information policies targeted to producers may be an effective accompanying measure. However, information policies lack effectiveness as a sole policy instrument if a specific environmental externality needs to be internalised. Nevertheless, information policies can be a good complementary measure in many circumstances, providing indirect benefits and enhancing the effects of other policies.

Favourable conditions for **agri-environmental schemes** include the existence of qualitative policy goals, a high certainty about the impact mechanisms and if farms are net-subsidy receivers. **Cross-compliance** regulations are preferable to AES if an even impact over all farms needs to be achieved and if exact targets need to be reached. Furthermore, cross-compliance is a policy option when large-scale environmental problems need to be dealt with, as the instrument has a binding character.

Community-based schemes are particularly effective in relation to small-scale problems, where networking among farmers is required. At the same time, a particular entrepreneurial understanding by farmers is not essential, since the mediation is often undertaken by a private or public service provider by order of the agricultural department or office. As the costs are borne by the policy maker, this instrument is especially useful if the farmers concerned are already net subsidy recipients.

In terms of tailoring and targeting policies (OECD, 2007d), it can be concluded on the basis of the above considerations that, with regard to the environmental problems analysed in this thesis, **energy use** as a predominantly global impact category may be most efficiently addressed by economic incentives. One reason for this is that the total energy consumption is important, while equal distribution of energy use is not relevant. Thus market-based mechanisms such as environmental taxes and tradable quotas seem to be preferable to command-and-control regulations, agri-environmental schemes or cross-compliance. Market instruments enable policy makers to take advantage of the potentially highly variable abatement costs and thus to achieve significant gains in economic efficiency. Furthermore, taxes may be preferable to tradable quotas, as the administrative burden will be much lower (Hepburn, 2006; Schleef, 1999).

In contrast to energy use, **biodiversity** is a qualitative environmental impact category which needs to take account of local conditions. For example, a tradable quota for Switzerland would lead to high effectiveness of the measure in the mountain regions but to low effectiveness in the lowlands. However, it is important for biodiversity to maintain a certain level in all regions. Furthermore, the application of market-based instruments would be difficult, as it would be difficult to establish units to attach the taxes to, e.g. fertiliser input, while measuring the impacts for quotas in a robust way would entail a high administrative burden. Therefore, command-and-control measures, cross-compliance and agri-environmental measures are appropriate instruments for addressing biodiversity.

Eutrophication is difficult to tackle using only one policy instrument. On the one hand, nitrate leaching and phosphorus eutrophication are predominantly local impacts. Ammonia eutrophication, on the other hand, has regional impacts. Both command-and-control mechanisms and cross-compliance are needed for ensuring an area-wide minimum eutrophication standard, particularly regarding nitrate and phosphorus eutrophication (Osterburg and Runge, 2007). However, employing market-based instruments for combating ammonia emissions and further reducing nitrate and phosphorus fertilisers would induce efficiency gains (Schleef, 1999).

Multiple policy goals and policy mixes

In the previous paragraphs the ability of single policy instruments to address single environmental problems was discussed. Theoretically, however, policy mixes could be also applied in pursuing single policy goals. Further, in reality multiple policy goals often have to be achieved simultaneously. In this situation, policy mixes and/or multi-objective policies can be applied. Hence the following paragraphs discuss the most salient considerations regarding a) policy mixes and b) multi-objective policies. Following this, the implications of the Tinbergen Rule for multi-objective policies are demonstrated using a simple linear programming model for the case of organic farming.

Policy mixes

Tinbergen (1956) found that at least one separate policy instrument per policy goal is needed to design efficient policy. However, there is no good reason to *a priory* limit the attention to only one type of instrument (Weizmann, 1974). The OECD (2007c) stresses that opportunities for mutually enhancing instruments should be exploited. Moreover, instrument mixes provide the possibility of responding flexibly to a changing environmental problem. Finn (2005) describes circumstances in which policy mixes are applied in policy practice in order to pursue a policy goal: Sometimes policy mixes are implemented because of either a poor scheme design or a lack of clarity about the goals. In other cases, the complexity of the environmental problem – *i.e.* when clear causal linkages cannot be established – results in the implementation of policy mixes. Furthermore, if several key mechanisms are known to be at the simultaneous driving forces of an environmental problem, these might be addressed by a mix of different policy instruments.

Yet, according to Hepburn (2007), implementing multiple instruments is problematic if these are incompatible with each other. If the interactions between different policies are not carefully considered, there may be adverse results. Since additional transaction costs are associated with each policy instrument, no additional policy instruments should be implemented if their existence is not justified by pursuing a certain goal (Dabbert *et al.*, 2004). The OECD (2007c) emphasises that the risk of mutually conflicting policies increases the more policy instruments are involved. Additionally, overlapping policy instruments ought to be avoided, since these tend to hamper flexibility and generate unnecessary transaction costs (OECD, 2007c).

These general rules can function as guiding principles for practical agri-environmental policy, yet policy making has to take account of more than just economic efficiency criteria. From a rational policy maker's point of view, creating a further policy instrument makes sense only provided additional overlap and transaction costs do not outweigh benefits gained from a more precise policy.

Multi-objective policy instruments

Tinbergen (1956) proved mathematically that there should be at least as many specific policy instruments as there are policy objectives. However, the Tinbergen Rule is applicable only on the assumption that there are no conflicting goals, co-benefits of policies and no transaction costs (Hallett, 1989; Stolze *et al.*, 2000).

Looking at the reality of agri-environmental policy instruments, these assumptions are hardly ever given. Particularly co-benefits and/or detrimental side-effects exist for almost every instrument in agri-environmental policy. Even if policies are designed especially to deal with a single environmental problem, they may have substantial effects on other environmental categories. For instance, buffer strips along streams and river banks are usually implemented to pursue the goal of both to reducing the transfer of nutrients into waterways and increasing the diversity of farmland wildlife. Thus buffer strips could be conceptualised either as a multi-objective policy or a single-objective policy with a co-benefit (Feng and Kling, 2005).

Organic farming area support payments (OFASP) are a more complex example of such a policy measure. Theoretically, OFASP could be implemented to address a single agri-environmental problem, namely to reduce the influx of toxic substances into water bodies. However, as discussed in more detail in Chapter 3, organic farming generates multiple environmental benefits, thus qualifying the measure as a multiple-objective policy.

Von Alvensleben (1998), Mann (2005a), and the Swiss Federal Council (2009) referred to the Tinbergen Rule (1956) when they concluded that multi-objective policies such as the organic farming area support payments are not economically sound, as the policy goals could be achieved more efficiently by more flexible and targeted combinations of various agri-environmental measures. Ahrens and Lippert (1994) also conclude on the basis of the Tinbergen model that *'a link of policy instruments should be avoided.* […] *A link* [of policy instruments] *results in the simplest case in merging two policy instruments in to one'* (Ahrens and Lippert, 1994, p. 152, translated).

At the same time, the multi-purpose character of organic agriculture could increase its cost-effectiveness due to potentially lower transaction costs compared to targeted agri-environmental measures (Dabbert *et al.*, 2004). Mann (2003c) even discussed whether organic farming should receive governmental support due to 'merit good' characteristics (Erlei, 1992; Musgrave, 1959) of organic food.

Implications of the Tinbergen Rule for organic farming area support payments

In the following it is shown that the Tinbergen Rule is not a sufficient criterion for excluding multi-objective policies, such as OFASP, from the policy mix for efficiency reasons. In order to do this, a simple theoretical linear public expenditure allocation model is used, which excludes transaction costs, merit goods, and conflicting goals. The other assumptions of the model are kept very simple, but do in principle comply with the current state of knowledge according to the scientific literature.

Suppose that a government has the environmental policy goals A, B and C. Suppose, further, that the government has set specific quantitative targets for each goal and is able to measure the exact level of goal attainment. Moreover, let us assume that it was possible to pursue each of the goals using a specific agri-environmental measure. Additionally, organic farming was able to be used to meet all three policy goals simultaneously but was only half as cost-effective with respect to individual policy goals. These assumptions correspond with the main body of literature as outlined in Section 3.

The question of the optimal combination of policy measures turns out to be a linear optimisation problem with the following specification: The objective of the government is to minimise public expenditure (PE) as a function of the parameter 'payment level' (PL) and the variable 'policy implementation' (PI), *i.e.* adoption (Equation 4).

$$\min PE = \sum_i PI_i \ PL_i \qquad (4)$$

where i is the index of policy instruments.

A goal attainment index (GAI_j) for goal j ranging from 0 to 100 % is assumed. GAI_j is a function of the initial state of goal attainment (IS_j) and the cumulative impact of the policy instruments (IM_j) on goal j (Equation 5). The objective function (Equation 4) is subject to the constraint that each environmental target must be achieved (Equation 5).

$$GAI_j = IS_j + IM_j = 100\% \quad \forall j \qquad (5)$$

with

$$IM_j = \sum_i PI_i \ E_{ij} \quad \forall j \qquad (6)$$

where parameter E_{ij} is the environmental effect of the policy instrument i regarding policy goal j (Equation 6). PI_i is defined as non-negative. Parameter IS_j is set to 50 %. The environmental effects and the payment level of each policy instrument are presented in Table 4.

With this specification, OFASP is the most efficient solution for addressing the three policy goals. The other AEMs which are targeted specifically to one of the policy goals are not part of the least-cost solution. In the case of three policy goals, organic farming is chosen instead of the individual agri-environmental payments, provided that the cost-effectiveness of the targeted AEMs regarding the single goals does not exceed three times the cost-effectiveness of OFASP.

Table 4 Simplified assumptions on effect and cost parameters for policy instruments

Policy instrument	Effect on goal A	Effect on goal B	Effect on goal C	Payment level
AEM_A	2	0	0	1
AEM_B	0	2	0	1
AEM_C	0	0	2	1
OFASP	1	1	1	1

Source: own assumptions

The above assumption that all indices of goal attainment are at the same state before the policies are introduced corresponds to reality only in exceptional situations. What is more common is that the distance between initial state and policy target differs for each policy goal. Furthermore, a policy mix has to be flexible in order to respond to changes in the target setting or the environmental state. To take this into account, different initial states are modelled in relation to a single goal, *ceteris paribus*. Assuming the initial state regarding energy use (policy goal A) varies between 0 and 100 %, the efficient solution results in a policy mix of OFASP and the targeted AEMs (Figure 2).

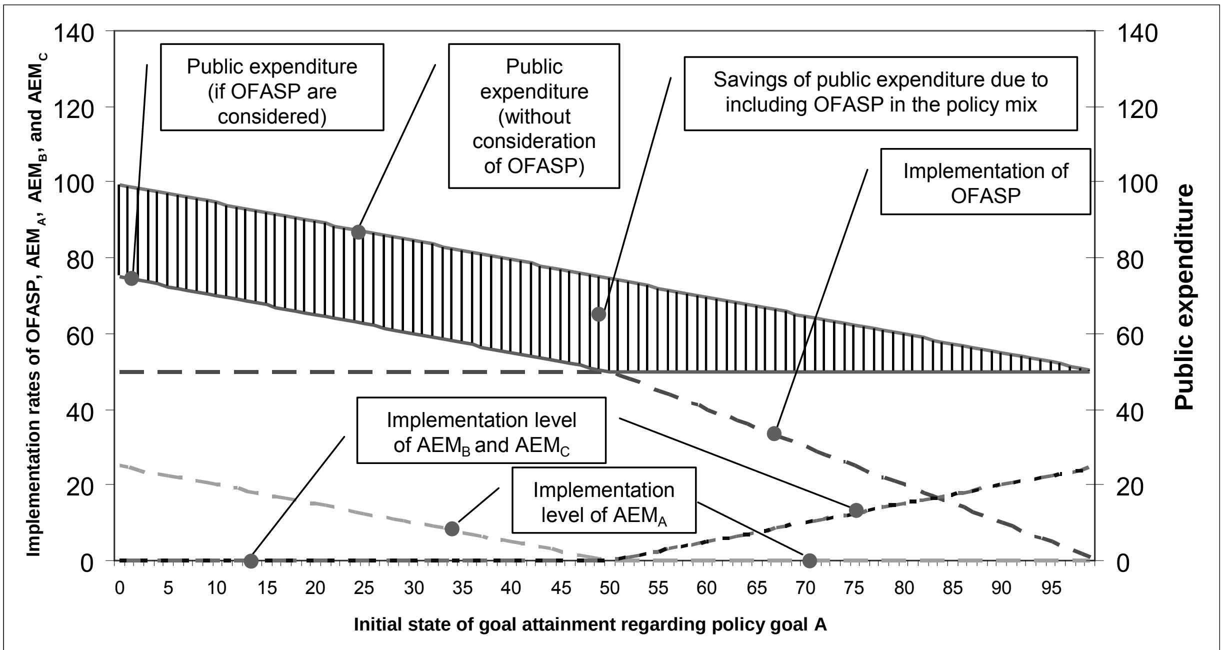


Figure 2 Public expenditure and optimal combination of policies depending on different initial environmental states

While the question of whether or not OFASP are part of the optimal solution depends on the cost-effectiveness ratio, the question of the extent to which OFASP is part of the optimal solution depends on the relation between the distances between the initial state and the policy target for the different policy goals. If the distance-to-target is equal for each policy goal, then the model results in the sole implementation of OFASP (Figure 2).

If the distance-to-target becomes greater for one goal than for the others, the gap in goal attainment left by OFASP is closed by implementing the relevant targeted AEM addressing the respective policy goal (in Figure 2 this is AEM_A). At the same time, the budget share of OFASP goes down if policy goal A has a lower distance-to-target than the others. In this example, OFASP are implemented to achieve environmental impacts until goal A has a goal attainment index of 100. The remaining gaps in goal attainment of goal B and C are filled by specific policy measures (AEM_B and AEM_C) as these are more efficient. With decreasing distance-to-target, the share of OFASP in total public expenditure goes down. The difference between the two public expenditure curves can be interpreted as the public expenditure saved by including OAFSP in the portfolio of agri-environmental measures (hatched area in Figure 2).

Thus the model results demonstrate that OFASP can in fact be a part of an efficient solution for addressing environmental problems, not as a sole instrument but as a complementary instrument alongside other measures. The modelled optimal budget share allocated to OFASP depends on the number of policy objectives taken into account, the effectiveness and costs of the OFASP relative to the targeted payments, and the relative distance-to-target of environmental categories before applying the instruments. In the modelled example with three policy goals, OFASP payments can be up to 66 % less effective than targeted AEMs and still provide the most efficient solution. Furthermore, OFASP payments – even assuming only 50 % of cost-effectiveness in pursuing single objectives compared to targeted payments – may be up to 49 % more expensive per unit (e.g. ha of land under the agri-environmental policy) and still be more efficient.

If more than three organic farming-oriented agri-environmental policy goals are included in the model and the cost-effectiveness relation between organic farming and the other policy measures is kept constant, the optimal share of OFASP as a proportion of total public expenditure increases.

At this point it should also be emphasised that, in order to reduce the complexity of the question and to stay in line with Tinbergen's (1956) original theory, the condition of *'GAI equals 100 %'* was established (fixed target approach (Hallett, 1989)). However, theoretically the constraint could also be defined as '*GAI should equal at least 100 %'*, if further environmental effects are not perceived as adverse (flexible targets). In this case, different results would be obtained, as the higher cost-effectiveness of a multi-objective policy may outweigh the costs of exceeding the set target. A model formulation with flexible targets would also be justified if no clear quantitative target was defined.

Thus from this modelling exercise new insights could be gained. It demonstrated that OFASP can be an efficient policy instrument in a policy mix, provided the above assumptions are met. Only under exceptional circumstances, however, will OFASP be sufficient as a sole agri-environmental policy measure. Thus, the Tinbergen Rule cannot be used as a knock-out criterion against OFASP and multi-objective policies. This theoretical insight also applies to other policy contexts where multi-objective policies could be applied. Consequently, the question of the efficiency of OFASP needs to be answered for each specific situation on the basis of empirical quantitative economic analysis.

2.3 Summary and conclusions

This thesis applies concepts derived from both neo-classical environmental economics and ecological economics. The theories of ecological economics are a) employed as ecological indicators referring to physical resources, thereby acknowledging the absolute limits of resources within an economic model and b) are utilised within a multi-dimensional approach with different environmental and economic categories, instead of a single-scaled, CBA-based, welfare-economical approach. Concepts of neo-classical welfare economics were used to classify and qualitatively analyse the policy instruments. Furthermore, the ‘FARMIS’ model employed later on is based on neo-classical production economics.

The above comparison of the economics concepts used in agri-environmental policy evaluation clarified that both the costs and environmental effects of each policy instrument need to be contrasted for an economic analysis. For the main research questions of this thesis, the CBA approach is inappropriate (for the reasons outlined in Section 2.2.2). Thus the evaluation approach used in this thesis (explained in Chapter 6) combines elements of the CEA and MCA to create an evaluation tool for addressing the research question.

The analysis of policy instruments in Section 2.2.3 revealed that it is complex to determine the cost-effectiveness of agri-environmental policies because many types of costs and benefits are involved. Therefore, a qualitative analysis cannot provide adequate answers for the problems in question. This is why the problem is addressed using a quantitative modelling approach, outlined in Chapter 6.

The appraisal of single policy instruments shows that a reduction in energy use can be pursued by both economic and regulatory measures, although the economic instruments are superior in terms of economic efficiency. Biodiversity, as a primary qualitative category, can be addressed by instruments containing regulatory elements. Nitrate and phosphorus eutrophication are probably best reduced by command-and-control measures, as regionally specific management strategies need to be controlled to avoid ‘dirty zones’ (Bader, 2005). Ammonia eutrophication, by contrast, is a more wide-scale problem which can be addressed by economic instruments in order to produce a more efficient solution. Economic instruments are appropriate in general, depending on which kind of eutrophication problem needs to be dealt with. However, economic instruments may not be suitable as a single solution, since they do not guarantee an even compliance of all farms with the same environmental standards.

From a rational policy maker’s point of view, creating a further policy instrument makes sense only as long as the additional overlap and transaction costs do not outweigh the gained benefits of a more precise policy.

Using an own theoretical linear programming model it was shown that the Tinbergen Rule does not in principle contradict multi-objective policies as long as the cost-effectiveness ratio of the multi-objective policy is not lower than a certain level. This level depends on the environmental effects, the public expenditure and the number of policy goals considered. The optimal budget share depends also on the relative distance-to-target of the different policy goals.

3 Environmental performance and costs entailed by organic farming

In this chapter, the rationale behind organic farming is outlined, as well as its environmental impacts (Section 3.1). There then follows a review of international studies that identify, quantify, and monetise the general environmental effects of organic farming. A general overview of the impacts is given first, after which the impact categories energy use, biodiversity and eutrophication[8] are discussed in detail (Section 3.2). Studies analysing the monetary costs of organic farming are also investigated (Section 3.3), and a review of studies on cost-effectiveness is conducted (Section 3.4). The concluding section of this chapter (Section 3.5) highlights the most important facts emerging for the subsequent analysis.

3.1 Rationale of organic farming systems

Organic farming is a distinct form of agriculture that has emerged in the course of the 20th century as an environmentally-friendly alternative to conventional agriculture (Niggli, 2007; Vogt, 2007). In the course of rapid technical innovations, conventional agriculture became increasingly capital intensive, input dependent and specialised. Bound to strict rules, organic farming did not follow this path, which led to a significant gap between the two farming systems over time. Lampkin (1990) stresses, however, that several misconceptions exist regarding organic farming: commonly, organic farming is conceived as farming in the pre-1939 style or a production method that does not use chemicals, substitutes mineral fertilisers with organic fertilisers, and bans pesticides. The role of agro-ecosystem management and other positive management practices is often ignored in such conceptions.

The international umbrella organisation of organic agriculture, the International Federation of Organic Agriculture Movements (IFOAM), defines organic agriculture as:

[8] These environmental categories have been selected according to the criteria discussed in Section 1.1. Further details are provided in Section 6.3.7.

> '[…] *a production system that sustains the health of soils, ecosystems and people. It relies on ecological processes, biodiversity and cycles adapted to local conditions, rather than the use of inputs with adverse effects. Organic agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved'* (IFOAM, 2009).

This definition highlights the largely self-sustaining nature of organic farming as a farming system (Köpke *et al.*, 1997). Detailed standards, principles and aims are set out by IFOAM in the periodically revised 'IFOAM Norms'. These contain the 'IFOAM Basic Standards' as an international guideline for national standards in organic agriculture. Council Regulation (EC) No 834/2007[9] was also based on the IFOAM Basic Standards and provides a binding framework for EU Member States (IFOAM, 2009). More detailed rules for the implementation of organic farming in the Member States are set out in Commission Regulation (EC) No 889/2008[10]. In Switzerland, the federal (country-wide) standards[11] have been developed according to Council Regulation (EEC) No 2092/91[12] and are due to be updated in 2009 and 2010 according to Regulation 834/2007.

3.2 Environmental impacts of organic agriculture

This section provides a general review of the evidence on environmental impacts of organic farming systems (Section 3.2.1). The environmental categories energy use, eutrophication

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[9] Council Regulation (EC) No 834/2007 of 28 June 2007 on organic production and labelling of organic products, repealing Regulation (EEC) No 2092/91, O.J. L 189/21 2007. This regulation was amended by Council Regulation (EC) No 967/2008 of 29 September 2008, O.J. L 264/1 (2008).

[10] Commission Regulation (EC) No 889/2008 of 5 September 2008 laying down detailed rules for the implementation of Council Regulation (EC) No 834/2007 on organic production and labelling of organic products O.J. L 250/1 which was amended by Commission Regulation (EC) No 1254/2008 of 15 December 2008, amending Regulation (EC) 889/2008 laying down detailed rules for implementation of Council Regulation (EC) No 834/2007, O.J. L 337/80.

[11] Ordinance on Organic Farming. Verordnung des EVD vom 22. September 1997 über die biologische Landwirtschaft (SR 910.181)

[12] Council Regulation (EEC) No 2092/91 of 24 June 1991 on organic production of agricultural products and indications referring there to on agricultural products and foodstuffs

with nitrogen and phosphorus and biodiversity are reviewed, as these are directly relevant to the focus of this study. These categories are described in relation to a) their general importance for society, b) the general contribution of agriculture compared to other activities and c) the differences between organic and non-organic systems.

3.2.1 Overview of environmental impacts of organic agriculture

This section provides an overview of environmental impacts of organic agriculture. It aims neither at completeness nor at differentiating environmental impacts for specific cases. For more detailed information on the environmental impacts of organic agriculture see Stolze *et al.* (2000), Shepherd *et al.* (2003), Lampkin (2007), and Mondelaers *et al.* (2009).

The environmental impacts of human activities have been increasing with growing populations and industrialisation (Meadows *et al.*, 1972). In recent decades, these environmental impacts have received greater attention from policy makers, scientists and the public (Aeschenbacher and Badertscher, 2008).

As indicated by the above-mentioned standards and production restrictions, the management of organic farming systems differs systematically from that of non-organic farming systems. These differences in management imply multiple and systematic effects on the environment (Morris *et al.*, 2001). In economic terms, environmental effects can be characterised as positive or negative externalities (see Section 2.1), since these effects are not taken into account by market actors (Pigou, 1932).

As will be shown below, there are many studies identifying the positive and negative effects of organic agriculture. The impact categories most commonly analysed are biodiversity, abiotic resource use efficiency, food quality, soil fertility, climate change and animal welfare (Köpke, 2002; Lampkin, 2007). In general, environmental effects are grouped according to the types of natural resources concerned. A widespread classification and overview of the environmental impacts of organic farming, based on Stolze *et al.* (2000), is shown in Figure 3. This review has been adapted to the terminology used in this thesis and updated on the basis of the most relevant studies published since 2000. In this qualitative figure, ‘X’ indicates the most frequent result in literature, while the grey spread indicates the range of different outcomes of studies found. Due to regional differences, farm and management-specific impacts, and gaps in scientific measurement methodologies, there is a range of uncertainty.

However, several impacts can be determined relatively precisely, since their systematic influence dominates regional or farm-specific differences (Stolze *et al.*, 2000).

As a more detailed review of biodiversity impacts below will show (Section 3.2.3), the impacts of organic farming on **biodiversity** range from much better to equal compared to non-organic agriculture. According to most studies, organic agriculture clearly performs better for faunal and floral species diversity (Bengtsson *et al.*, 2005). Thus, Stolze's evaluation (2000) has been confirmed by recent studies. Concerning **landscape** and habitat diversity organic farming may perform better due to more diverse crop rotations (Norton *et al.*, 2009) and higher implementation rates of structural elements such as hedges and fruit trees (Schader *et al.*, 2009b). However, landscape effects are very farm and site-specific. Therefore, no general trend can be determined (Steiner, 2006).

According to Stolze *et al.* (2000) organic agriculture has no general effect in terms of **greenhouse gas emissions**. However, recent studies suggest that this depends largely on the assumptions and system boundaries of the analysis. Furthermore, there are indications of better performance regarding CO_2 sequestration (Niggli *et al.*, 2009; Olesen *et al.*, 2006). Thus recent studies suggest a change in the appraisal that Stolze *et al.* made in 2000 from 'equal' to 'better'.

Ammonia emissions and pesticide emissions into the **air** are much lower in organic systems (Stolze *et al.*, 2000). There is no recent empirical evidence that might call into question the appraisal made by Stolze *et al.* (2000) (See Section 3.2.4 for further details on ammonia eutrophication).

Organic farming performs much better in terms of **soil** biological activity (Mäder *et al.*, 2002) than non-organic farming. Soil erosion and organic matter content are also affected positively by organic practices, although soil structure remains unaffected (Fliessbach *et al.*, 2007).

Eutrophication of ground and surface **water** is very much dependent on what exactly is the subject of comparison. The impacts of nitrate leaching from organic farming can range from better to worse (Kirchmann and Bergström, 2001; Kramer *et al.*, 2006) compared to conventional agriculture. However, most of the studies analysed found that organic farming performs better (Köpke, 2002; Stolze *et al.*, 2000) (see Section 3.2.4 for more details). Regarding pesticide emissions into ground and surface water, organic agriculture performs much better due to the ban on artificial pesticides (Nemecek *et al.*, 2005).

Organic agriculture is	much better	better	equal	worse	much worse
Biodiversity and Landscape		X			
Genetic diversity			X		
Floral diversity		X			
Faunal diversity		X			
Habitat diversity			X		
Landscape			X		
Climate change			X		
CO_2		X			
N_2O			X		
CH_4			X		
Air quality		X			
NH_3		X			
Pesticides	X				
Soil fertility		X			
Organic matter		X			
Biological activity	X				
Soil structure			X		
Soil erosion		X			
Ground and surface water pollution		X			
Nitrate leaching		X			
Phosphorus runoff		X			
Pesticide emissions	X				
Resource depletion		X			
Nutrient resources		X			
Energy resources		X			
Water resources			X		

Source: based on Stolze et al. 2000

Figure 3 Classification of environmental impacts and relative performance of organic farming compared to conventional farming

Regarding **resource-use efficiency**[13], organic farming performs better regarding nutrients and energy (Nemecek *et al.*, 2005), which confirms the evaluation done by Stolze *et al.* (2000). Water consumption as an environmental indicator is of secondary relevance in Switzerland. Compared to conventional farming, the input is not substantially affected by organic farming systems (Haas *et al.*, 1995; Stolze *et al.*, 2000).

3.2.2 Fossil energy use

Fossil energy use has a two-fold role as an indicator of environmental pressure. First, it is the major driver of greenhouse gas accumulation in the atmosphere and thus leads to climate

[13] Further details on the impacts of organic agriculture on fossil energy use are provided in Section 3.2.2.

change (IPCC, 2007). Second, it leads to resource depletion due to the fact that most of the primary energy sources used stem from fossil non-renewable resources such as oil, gas, coal and uranium (Constanza *et al.*, 1998; Pervanchon *et al.*, 2002).

Societal relevance

Climate change has been perceived for decades as a significant global environmental problem. Over the last years in particular the environmental awareness has increased markedly within the general population, partly due to reporting on the international negotiations in relation to the Kyoto Protocol, and partly because of more visible impacts of climate change on ecosystems such as glaciers or Polar Regions. The higher incidence of natural disasters such as hurricanes, droughts and floods has also contributed to the growing awareness of climate-change over the last few years (IPCC, 2007). Recent studies estimate that the cost of current and projected levels of greenhouse gas emissions and the climate change caused by them exceeds their abatement costs (Stern, 2007). As agricultural production has an impact on all the three major greenhouse gases (CO_2, CH_4 and N_2O), it is perceived as a crucial and potentially cost-effective lever for mitigating climate change (Smith *et al.*, 2007).

Resource depletion is a problem of similar magnitude against the background of industrial civilisations' growing dependency on fossil fuels (Meadows *et al.*, 1972). The debate about peak oil (*i.e.* the point in time when the maximum rate of global oil extraction has been reached) is currently increasing in intensity (Zittel and Schindler, 2007). Agriculture, once a net energy producer has today become a net energy consumer for some commodities (Leach, 1976). Given the need for efficient resource use, energy use has become a standard environmental indicator (Frischknecht *et al.*, 2007; Pimentel *et al.*, 2005).

Contribution of agriculture

About 12 to 14 % of global greenhouse gas emissions stem from the agricultural sector (Smith *et al.*, 2007). In Switzerland, agriculture and forestry comprise 11 % of the total greenhouse gas emissions. However, while CH_4 and N_2O emissions are predominantly attributable to agriculture, the share of the agricultural sector in terms of CO_2 emissions caused mainly by burning fossil fuel is disproportionately small. Nevertheless, many studies suggest that food consumption is a major contributor to total societal energy use (Ziesemer, 2007). Studies comparing crop and animal products conclude that crop products have much higher energy efficiency per unit of digestible energy than produce from animal production (Pimentel and Pimentel, 2005).

Effects of organic agriculture

The impacts of organic agriculture on energy use can be analysed on the basis of different functional units (Halberg, 2008). While some studies use 'area' as a unit (Haas *et al.*, 2001), others take the weight of output from the farming system as a reference (Grönroos *et al.*, 2006). Although the latter approach is in line with the standard procedure within life cycle assessments (Heijungs *et al.*, 1992) and illustrates energy use per unit of food produced, it still has weaknesses when it comes to analysing agricultural systems. Often, research on farming systems encompasses consideration of multiple outputs. Either these outputs need to be expressed in a single unit, or an allocation of the energy use has to be performed, or again, by-products need to be deducted to enable a comparison across all products (Schader *et al.*, 2008a). A product-related assessment additionally involves the determination of the functional unit. However, the scorings related to weight, volume, calories or protein might produce highly varied results.

Stolze *et al.* (2000) concluded that organic farming systems perform better than conventional ones in terms of energy use per ha. According to Lampkin (2007) most product and area-related energy use assessments of organic farming to date show a lower energy use per ha. Due to the generally lower productivity of organic farming, per-ha comparisons reveal higher differences than product-based comparisons. Unlike Stolze *et al.* (2000), Lampkin's (2007) review also identified that, for most products, energy use was also lower per unit food produced, except in the case of potatoes.

Haas *et al.* (2001) compared organic and conventional grassland farms in southern Germany. They found a 44 to 46 % lower energy use per ha and per tonne of milk. Thomassen *et al.* (2008) also analysed milk production and found that the energy efficiency of organic production was significantly higher compared to conventional production. Thomassen *et al.* concluded that the use of concentrates in particular is a major driver and has potential for reducing energy use.

Grönroos *et al.* (2006) calculated that fossil energy use for organic rye bread and milk was lower by 13 % for rye bread and 31 % for milk – compared to conventional products. From a cradle-to-(farm)gate perspective, the difference is even higher, with organic products consuming only 50 % of energy use of conventional products. Similar results were generated by Hoeppner *et al.* (2005) who compared the energy use throughout a rotation. Energy use and energy output were 50 % and 30 % lower respectively over organic rotations in a long-term field experiment. This results in higher energy efficiency (energy use per product) of organic farming compared to conventional farming.

Nemecek *et al.* (2005) demonstrated a lower energy use per ha and per product unit overall in organic systems for all major crops in Switzerland. This was done by analysing data from long-term field experiments and generating subsequent calculations aimed at generalising the results for Switzerland. An exception to this is potatoes, where a slightly higher energy use was calculated per tonne of organic potatoes.

In summary, organic farming has a lower energy use per ha and, in most cases, higher energy efficiency – *i.e.* input/output ratio – than conventional farming. There are only a few exceptions on the crop-production side, notably potatoes, with organic systems displaying lower energy efficiency due to low relative productivity levels. While milk production is more efficient on organic farms, poultry production has shown slightly lower energy efficiency. Thus, the quantitative advantage of organic farming depends crucially on the product, the geographic region, and the assumptions of the study.

3.2.3 Biodiversity

Biodiversity can be described according to four different levels. First, it is expressed at species level, most simply by monitoring the species in selected groups, such as birds or plants in a certain area. Second, biodiversity can be expressed in terms of the regional diversity of habitats and ecosystems in which species live. Third, the functional diversity describes the complexity of ecological processes and interactions. Fourth, diversity within species (genetic diversity) includes the diversity of farm animals and crops. Genetic diversity enables species to adapt to changing environments, e.g. due to climate change (Christie *et al.*, 2006). One of the key messages of a major study on 'The Economics of Ecosystems and Biodiversity' (TEEB) was the '*inextricable link between poverty and the loss of ecosystems and biodiversity*'(ten Brink *et al.*, 2009)

Societal relevance

Biodiversity has a substantial influence on human society. According to the Millennium Ecosystem Assessment (MEA), *'Biodiversity benefits people through more than just its contribution to material welfare and livelihoods. Biodiversity contributes to security, resiliency, social relations, health, and freedom of choices and actions.'* (MEA, 2005). Apart from the fact that a robust nature *per se* enhances human welfare, ecosystems provide distinct environmental services to society, which have a measurable economic benefit. These ecosystem services encompass, for example, the provision of clean drinking water and pollination by bees. Furthermore, animal, plant and fungal species provide the potential to cure illnesses that either exist already or may develop in future. Despite the importance and the direct and potential use values of biodiversity, ecologists and economists alike emphasise the fact that biodiversity valuation comprises both use and non-use values (Christie *et al.*, 2008).

Nowadays, the decline in biodiversity continues due to the destruction, damage or fragmentation of habitats. This in turn is caused by a combination of factors, including high consumption of land used for residential areas and traffic infrastructure, intensive agriculture, abandonment of areas of marginal productivity, changed forestry practices, invasive species, unsustainable leisure-time activities, climate change and emitted pollutants (MEA, 2005).

Contribution of agriculture

Agriculture has an important influence on biodiversity as the major user of land in Switzerland (Aeschenbacher and Badertscher, 2008). Due to agricultural activities, a great variety of ecosystems have been created which, overall, have enhanced biological diversity. Agriculture affects biodiversity directly through cultivation practices. Furthermore, it affects biodiversity indirectly through nitrogen emissions into the air and CO_2 emissions into the atmosphere. On land under intensive agricultural cultivation, biodiversity decreases significantly due to the high nutrient influx, high cutting frequencies on meadows, high stocking rates, use of pesticides, and modern methods of processing cut grass (Knop *et al.*, 2006). In the lowlands, many diverse agricultural ecosystems have disappeared, while in the mountain regions two parallel trends are apparent: the intensification of productive areas and the abandonment of unproductive but ecologically valuable areas (Aeschenbacher and Badertscher, 2008).

Effects of organic agriculture

Alongside eutrophication effects, biodiversity effects are among the most frequently studied environmental impacts of agriculture. Recent meta studies (Bengtsson *et al.*, 2005; Fuller *et al.*, 2005; Hole *et al.*, 2005) show clear differences between organic and conventional farming systems. In very rare cases, organic production was found to have negative impacts, although this was outweighed by studies showing positive impacts. The differences vary among *taxa*, but for each species group large differences were found (Table 5). On average, an about 50 % greater species diversity was achieved on organic farms (Niggli *et al.*, 2008).

Apart from differences at species-group level, structural differences at farm level are prevalent between organic and non-organic farms (Gibson *et al.*, 2007; Schader *et al.*, 2008b). In addition, Boutin *et al.* (2008) identified higher species richness in semi-natural habitats on organic farms compared to conventional farms.

Table 5 Number of studies analysing the impacts of organic farming on biodiversity with respect to various *taxa* on the basis of 76 comparative studies[14]

Taxa	Impacts of organic farming		
	Positive	No difference	Negative
Plants	16	2	0
Birds	11	2	0
Mammals	3	0	0
Arthropods			
Beetles	15	4	5
Spiders	9	4	0
Butterflies	2	1	0
Bees	2	0	0
Other arthropods*	8	3	1
Bacteria, fungi and nematodes	12	8	0
Earthworms	8	4	2
Total	**87**	**28**	**8**

* mites, bugs, millipedes, flies, and wasps

Source: Hole *et al.* 2005, updated by Niggli *et al.* (2008)

[14] Updated using studies from 2004 to 2008

Organic farming practices are most beneficial for birds, predatory insects, spiders, soil organisms and the arable weed flora, while pests and 'indifferent organisms'[15] do not show different levels of abundance in the farming systems. Furthermore, differences in arable land between the farming systems are more pronounced than on grassland (Table 5).

Studies attribute the higher biodiversity in organic systems to the following factors: a) ban on herbicides and artificial pesticides, b) ban on mineral fertilisers, c) more diverse rotations, d) lower organic fertilisation e) careful tillage f) a higher share of semi-natural habitats in total UAA (Niggli *et al.*, 2008).

3.2.4 Eutrophication

Eutrophication is defined as nutrient enrichment in sensitive ecosystems (UNECE, 1999).

Societal relevance

Eutrophication entails various environmental impacts that cause both the loss of biodiversity and negative impacts on human health. Eutrophication leads to excessive growth of algae and excessive oxygen demand, with anaerobic conditions leading to foul smelling surface waters and the fish death. These effects of eutrophication can be understood as societal costs, either in terms of abatement, purification or restoration costs, or as damage costs if the negative impacts of eutrophication are not abated or fixed.

Contribution of agriculture

The main environmental risk entailed by agricultural production in relation to nutrient enrichment involves nitrogen and phosphorus. The leaching of mobile nitrates into ground and surface water and gaseous emissions such as ammonia (NH_3) from organic fertilisers are the major contributors to nitrogen eutrophication. Ammonia emissions affect ecosystems like forests, swamps and diverse meadows, which require low nitrogen levels. Furthermore, ammonia emissions into ecosystems cause acidification and the release of toxic substances including heavy metals.

[15] i.e. organisms which are neither a pest nor have a beneficial role for the crops

Nitrate pollution in the lowlands has been the most severe environmental problem resulting from post-war policies (Gruber, 1992). These policies provided incentives to run intensive, highly-yielding agricultural production involving heavy nutrient surpluses.

Phosphorus is relatively immobile in soils but can be emitted from agricultural systems to surface waters by erosion and run-off processes. While phosphorus rarely represents an environmental problem in rivers, it causes algae growth in lakes and seas, as normally, phosphorus is the limiting nutrient for algae growth. The decomposition of this additional plant material reduces the amount of oxygen. Finally, fauna die due to anaerobic conditions. Phosphorus emissions from agriculture give rise to high societal costs due to bad odours, costs of treatment, and the hindrance of recreational activities,.

Effects of organic agriculture

The reduction of nitrogen and phosphorus eutrophication demands efficient use of these nutrients (Herzog and Richner, 2005). Evaluations have identified that the problem of nitrate leaching occurs predominantly in arable farming systems, although leaching can also occur from grassland receiving high fertiliser inputs. Therefore, Herzog and Richner (2005) suggest that farms should no longer be permitted to have a 10 % nutrient surplus, as this is a factor in surpluses of total nutrient. Apart from systems that rely heavily on imported manures, e.g. horticultural systems, there is no nutrient surplus in organic systems, as nutrient import onto the farm is restricted for both feedstuffs and mineral fertiliser.

Several studies show that nitrogen leaching can be reduced by 40 to 64 % through organic farming. There are three facts underlining a lower eutrophication potential (Auerswald *et al.*, 2003; Condron *et al.*, 2000; Edwards *et al.*, 1990; Eltun, 1995; Goulding, 2000; Haas *et al.*, 2001; Kirchmann and Bergström, 2001; Mäder *et al.*, 2002; Osterburg and Runge, 2007; Pacini *et al.*, 2003; Shepherd *et al.*, 2003; Stolze *et al.*, 2000; Stopes *et al.*, 2002; Younie and Watson, 1992):

- Organic farming systems have lower nutrient levels, which reduces the absolute quantity of nutrients loads that can be emitted from the system.
- The quantity of directly available nitrogen is much lower in organically managed soils.

- As the nutrients cannot be imported easily into the systems, the opportunity cost of nitrogen losses is higher for organic farms than for conventional farms. This implies a need for more efficient nutrient management in organic systems, although this does not eliminate losses. In addition, nitrate leaching can be high at the point of transition from the fertility building phase of the rotation to the cropping phase.

In contrast, Nemecek *et al.* (2005) found higher eutrophication impacts for some organic crops compared to their conventional counterparts. In places, these higher nutrient loads on arable land are attributed to the greater use of organic fertilisers in the organic system, since the life cycle assessments used by Nemecek *et al.* (2005) assume relatively high fertilisation rates for organic farms.

Nevertheless, most international studies show lower per ha N and P losses on organic farms (Auerswald *et al.*, 2003; Condron *et al.*, 2000; Edwards *et al.*, 1990; Eltun, 1995; Goulding, 2000; Haas *et al.*, 2001; Kirchmann and Bergström, 2001; Pacini *et al.*, 2003; Stopes *et al.*, 2002).

3.3 Costs of organic agriculture as a policy option

This section outlines, first, a number of different perspectives for analysing the costs of organic farming as a policy option. Second, studies on the single cost components of organic farming are discussed.

Perspectives on policy-relevant cost of organic agriculture

In examining the costs associated with organic agriculture, three different views can be distinguished:

First, cost can be interpreted from a **farm-level perspective**, taking into account farm level costs only. Usually, such a perspective would distinguish between technical costs, *i.e.* the need to purchase new machinery, and costs caused by loss of production. Farmers also bear administrative costs, mainly as a result of their additional labour input for administrative tasks related to policy implementation and certification and paying for private certification. To the extent that certification is an eligibility requirement for policy support, these can be subsumed under policy-related transaction costs at farm level (OECD, 2007b). Where organic farming area support payments (OFASP) are implemented, e.g. under Council Regulation (EC) No 1698/2005, these are typically aimed at compensating farmers' additional costs and income forgone. In practice, however, there are substantial cost differences between individual farms, which give rise to differences in the profitability of organic farms. Thus the cost of organic farming from a farm-level perspective can be conceptualised as the sum of technical costs (TEC), the costs of production loss (PLC) and farm-level transaction costs ($PRTC_{FARM}$) minus the additional direct payments (OFASP)[16] (Equation 7). It should be noted that PLC in particular can also have a positive influence for some farms if the loss in production of physical units is outweighed by higher producer prices. This farm-level perspective was the one most frequently taken in the studies reviewed. For example, it is used to analyse the profitability (Nieberg and Offermann, 2002; Offermann *et al.*, 2005) and farm-level economic efficiency (Lansink *et al.*, 2002; Larsen and Foster, 2005) of organic farming.

$$COST_{FARM-LEVEL} = PLC + TEC + PRTC_{FARM} - OFASP \qquad (7)$$

Second, policy costs can be interpreted from a **budgetary perspective**. This perspective comprises the costs associated with payments to farmers (PC), including policy-related transaction costs ($PRTC_{PUBLIC}$), which occur during the decision-making, implementation and evaluation phase of the policy (Equation 8). This view implicitly conceptualises organic farming as an agri-environmental measure and at the same time neglects the other impacts. Marggraf (2003) employed such a perspective in a comparative analysis of agri-environmental measures in the German *'Länder'*. Osterburg and Runge (2007) also calculated the cost-effectiveness of different agri-environmental policies in Germany using this framework for the cost side. However, both studies excluded public transaction costs.

$$COST_{BUDGETARY} = PC + PRTC_{PUBLIC} \qquad (8)$$

Third, the **welfare economics perspective** comprises the total of all monetary and non-monetary costs a policy entails (Pretty *et al.*, 2000). This view also includes producers' forgone revenues and external costs that might occur due to negative environmental cross impacts. Cost is classified as a change in producer surplus (ΔPS), consumer surplus (ΔCS) and public expenditure (ΔPE) (Henrichsmeyer and Witzke, 1994a). External costs and

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[16] For reasons of simplification it is assumed that no direct payments other than OFASP are involved.

benefits are also added (ΔEX). From an economic point of view, this perspective is the soundest one, as it includes all costs and benefits. The welfare economic perspective is the foundation of cost-benefit analysis, which has been discussed in Section 2.2.2.

$$COST_{WELFARE} = \Delta PE + \Delta PS + \Delta CS + \Delta EX \qquad \textbf{(9)}$$

A comparison of the different perspectives (farm-level, budgetary, and welfare economics) shows that different results will be generated depending on which perspective is opted for. Like other agri-environmental payments, the aim of OFASP is to compensate farmers for the costs incurred by them. Therefore, the sum of TEC, PLC, and $PRTC_{FARM}$ equals OFASP (Equation 7) and PC (Equation 8) if the policy is 100 % targeted. However, from the farm-level perspective the agri-environmental payments (OFASP) appear on the benefit side. For this reason, this perspective is not suitable for calculating societal costs. The budgetary perspective additionally includes public transaction costs ($PRTC_{PUBLIC}$).

The welfare-economist's perspective represents a further enlargement of the perspective, since it includes both all costs and benefits in monetary terms. As in the budgetary perspective, both payments and public transaction cost of the policy are subsumed under ΔPE (Equation 9). However, as the agri-environmental payments are transferred from taxpayers to producers, payments have to be added again to the producer surplus. Therefore, a positive net change of the producer surplus can be anticipated, as windfall profits are common for agri-environmental measures (Section 2.2.3). The consumer side is also included by quantifying the net change in consumer surplus, e.g. changes in market prices and transfers from consumers to producers. Furthermore, the economist's perspective covers external costs, which is usually analysed on the basis of the willingness to pay for environmental benefits or by quantifying the costs of damages to ecosystems or the costs for fixing these damages.

Review of literature on different cost components

In the following, the literature available on the costs of organic farming by single cost categories are discussed. The focus here is on public expenditure and transaction costs as the most relevant categories in the analysis to follow later (Chapter 6). **Externalities** are excluded from this review, as these have been discussed in Section 3.2 as environmental impacts.

Technical costs comprise mainly investments, e.g. in livestock housing or various machines, which have to be made predominantly during conversion (Hollenberg, 2001; Kerselaers *et al.*, 2007). In addition, technical costs arise from the higher demand for labour on organic farms (Morison *et al.*, 2005).

Production losses are caused by the prohibition of most pesticides and mineral fertiliser use, as well as by restrictions regarding the intensity of animal husbandry and other constraints. These restrictions often entail losses in production quantity (Offermann *et al.*, 2009) and, in most cases, reductions in physical yields compared to intensive agriculture (Badgley *et al.*, 2006).

Most quantitative studies adopt the budgetary perspective and discuss **public expenditure** or payment levels (Nieberg and Kuhnert, 2006; Nieberg and Kuhnert, 2007). In a European comparison of payment levels associated with conversion to and continuing organic farming, Stolze and Lampkin (2009) and Nieberg and Kuhnert (2006) found substantial variations between Member States (Table 6). Ranges of payment rates reflect regionally targeted payments.

Table 6 shows that in 2004/05, support for the maintenance of organic farming was paid in all countries except France. In all countries except Austria, the Czech Republic, Estonia, Portugal, Slovenia, Spain, Sweden, and Switzerland, conversion payments are higher than payments for continuing organic farming.

Particularly high maintenance payments for grassland were found in Austria (122-324 €/ha) and Belgium (55-275 €/ha), while the lowest payment rates were found in the UK (20-51 €/ha). For arable land, Switzerland (530 €/ha), Italy (110-600 €/ha), Greece (261-327 €/ha) and Belgium (240-350 €/ha) provide the highest payments, whereas UK, Spain and Estonia provide less than 100 €/ha. In most countries, vegetable growing receives substantially higher payments than arable or grassland crops. The highest rates for arable land are paid in Austria (545-690 €/ha), Belgium (380-750 €/ha), and Switzerland (795 €/ha), while the UK again has the lowest payment rates, at 20-51 €/ha. Finally, payments for permanent crops reveal a similar pattern, with 872 €/ha in Austria, 555-750 €/ha in Belgium, up to 924 €/ha in Germany, up to 900 €/ha in Italy, 788 €/ha in Sweden, and 795 €/ha in Switzerland. At the other end of the scale, the UK pays 20-44 €/ha for continuing organic farming. The high variations in payment rates among Member States are unlikely to be due to farm-economic reasons.

Instead, the differences in payment rates are most likely an indication of political priorities and budgetary constraints.

The net change in **producer surplus** equals the net change in farm income minus the compensatory direct payments (Schleef, 1999). There is empirical evidence of a positive impact of organic farming on producer surplus. The profitability of organic farms is higher compared to conventional farms in most EU Member States (Offermann *et al.*, 2009). Sanders (2007) found that organic farms had a higher level of profitability than conventional farms in Switzerland as well, and if Swiss agricultural policy is liberalised, this will lead to an increase in the relative profitability of organic farms.

Table 6 Financial support for conversion to and maintenance of organic production in different EU countries, 2004/2005 (in €/ha)

Country	Grassland		Arable		Vegetables		Permanent crops	
	Conversion	Maintenance	Conversion	Maintenance	Conversion	Maintenance	Conversion	Maintenance
Austria	122–324	122–324	363	363	545–690	545–690	872	872
Belgium	252–335	55–275	410–456	240–350	810–894	380–750	788–810	555–750
CzechRepublic	34	34	110	110	344	344	381	381
Denmark	187	117	187	117	187	117	187	117
Estonia	74	74	97	97	241	241	241	241
Finland	147	103	240	196	480	436	631	587
France	107	0	244	0	305	0	305–701	0
Germany	130–255	130–255	153–255	150–255	251–576	255–410	501–1,440	560–924
Greece	0	261–327	335–600	261–327	600	261–327	400–900	261–327
Hungary	59	59	178	127	329	202	400	281
Ireland	261–327	228–291	261–327	228–291	261–327	228–291	261–327	228–291
Italy	85–525	85–525	140–600	111–600	302–600	295–600	400–1,080	298–900
Lithuania	118	59	416	208	551	275	734–752	367–376
Luxemburg	180	150	180	150	360–510	300–450	510	450
Netherlands	136	136	147	136	147–737	136	885	136
Poland	72	57	149	131	215	206	394	337
Portugal	167–193	167–193	147–400	147–400	600	600	183–750	183–750
Slovenia	230	230	460	460	544	544	795	795
Spain	40–266	40–266	63–180	55–180	105–600	105–600	119–600	71–600
Sweden	53	53	137–231	137–231	525	525	788	788
Switzerland	133	133	530	530	795	795	795	795
UnitedKingdom	101–113	20–51	101–173	44–51	101–209	20–51	131–539	20–44

Source: Nieberg and Kuhnert (2006) in Stolze and Lampkin (2009), adapted

Reasons for this higher profitability are the higher levels of direct payments and higher producer prices, which over-compensate farmers for the costs they incur and their income forgone. Further details on the profitability of organic farming can be found in a recent dissertation by Sanders (2007).

In contrast to the large body of evidence regarding the impact of organic farming on the producer surpluses and on the profitability of organic farms, no study examining the net change in **consumer surplus** has been found.

Finally, the **transaction costs** of agri-environmental payments need to be addressed as a cost component. Within the rise of new institutional economics, transaction costs have been discovered as a relevant parameter that needs to be taken into account by policy evaluation (Williamson, 1989). Transaction costs are, however, more difficult to quantify, since they occur at different levels (farmer, certification body, or different levels of public administration) and are often difficult to link directly to the policy itself (OECD, 2001b). There have been various – in some cases contradictory – studies aimed at quantifying or estimating the transaction costs of organic farming and agri-environmental measures (Buchli and Flury, 2006; Cahill and Moreddu, 2004; Hagedorn *et al.*, 2003).

Organic farming generates relatively low transaction costs for public administration, whereas the costs related to certification incurred by farmers are notably high. The high producer transaction costs are attributable mainly to the strict 100 % annual inspections required for organic certification rather than the 5 % inspections required under EU regulations for agri-environmental measures carried out in Germany (Hagedorn *et al.*, 2003). Another study compared the cost-efficiency of organic area payments and taxes related to pesticides or fertiliser, using an Applied General Equilibrium model. The results indicate that taxes have a clear advantage over organic farming area support payments in terms of cost-efficiency due to their far lower transaction costs (Jacobsen, 2002). According to Vatn *et al.* (2002), this is not surprising given that the share of transaction costs as a proportion of total public expenditure becomes higher the more specific the policies are and the less frequently a transaction occurs. Therefore, taxes normally show evidence of higher transfer efficiency than agri-environmental measures, since the items the taxes are linked to, e.g. mineral fertilisers, are traded frequently.

Calculations of the transaction costs of Swiss direct payments in the cantons Zürich and Grison have been conducted recently. The results show that transaction costs account for a 9.7 % share of total costs associated with organic farming payments, while the agri-environmental measures range from 5 to 16.6 % of their total cost (Buchli and Flury, 2006). However, as calculated by Hagedorn *et al.* (2003) for two German *Länder*, farmers have to bear the bulk of the transaction costs. A Norwegian study identified policy characteristics which influence the amount of transaction costs and which may be also valid in the Swiss environment: The higher the asset specificity and the lower the number of farms adopting a policy, the higher the transaction costs become. Furthermore, commodity-oriented policies show significantly lower transaction costs than policies which are bound to land, since there are no transaction costs at farm level (Rørstad *et al.*, 2005).

Thus the conclusion that can be drawn from the existing literature is that organic farming is likely to incur higher costs at farm level. These costs are generally overcompensated for through higher producer prices and additional direct payments. The higher public expenditure on organic farms than on conventional farming entails higher societal costs, unless environmental benefits are monetised. The existing literature suggests that the private transaction costs are substantially higher in organic systems due to the 100 % inspections which farmers have to pay for. At the same time, public transaction costs of organic farming are lower if the government pays for the costs of the private inspections, compared with agri-environmental schemes requiring individual producer agreements (as opposed to tax policies).

3.4 Comparison of costs and environmental impacts of organic agriculture

As shown above, the calculation of the cost-effectiveness of organic agriculture is a complex undertaking, as multiple effects have to be taken into consideration. Furthermore, multiple cost components play a role in organic agriculture. Finally, due to the numerous options for combining of agri-environmental measures with which organic agriculture could be compared, the assumptions and system boundaries of the study constitute a significant factor in the results obtained. The following is a review of the literature which compares the costs and environmental effects of organic agriculture.

In a British farm-level study using a cost-benefit approach, the external benefits of organic agriculture were estimated at between £75 and £125 per ha and year (Cobb, 1998). O´Riordan and Cobb (2001) calculated the costs of conventional fields at between £25 and £40, while those of organic fields were between only £10 and £15 per field and year. Other studies, for instance by Pretty *et al.* (2000), suggest that current intensive agricultural production is economically unsound if external costs are taken into account and if societal savings are possible through agricultural production being extensified or reduced.

Studies addressing the cost-effectiveness of different policy instruments for such extensification concentrate mostly on specific environmental impact categories. An exception to this is Ziolkowska (2008), who included three environmental objectives (natural resources, biodiversity and cultural landscape) in her study. Using an analytical hierarchy process, she identified organic farming and extensive meadow farming as generating the greatest environmental benefits. The cost-effectiveness of these measures was, however, rated as comparatively low by different stakeholders (Ziolkowska, 2008).

Studies focussing on the abatement costs for **fossil energy use** were rare and did not include organic agriculture (Kränzlein, 2008). Similarly, studies focussing on cost-effectiveness regarding the provision of **biodiversity** did not include organic farming as a policy option (Julius *et al.*, 2003; Zgraggen, 2005). Only Sipiläinen *et al.* (2008) compared economic efficiency in food production and the production of the non-commodity 'biodiversity' between conventional and organic farms. They concluded that farm-level economic efficiency is lower on organic farms if biodiversity is excluded. However, if biodiversity is taken into account, the efficiency of organic farms is either higher (pooled farms) or differs to a non-significantly degree (single farm data).

While only a few studies deal with the cost-effectiveness of organic agriculture regarding energy use and biodiversity, many studies were found that deal with **eutrophication**, in particular nitrogen eutrophication. Most of these studies used a typical CEA framework with a budgetary perspective (Marggraf, 2003; Osterburg and Runge, 2007).

Jacobsen *et al.* (2005) calculated a medium cost-effectiveness of organic farming for reducing nitrate leaching at 3.8 €/ kg N. Other measures ranged from 0.7 €/ kg N (increased utilisation of N in animal manure) to 10.4 €/ kg N for lower livestock density. Osterburg and Runge (2007) provide ranges of abatement costs rather than specific values, as the reduction potential and the costs per ha are variable. They found costs of 0.7 to 6.7 € for lowering the nitrogen surplus by 1 kg N, generating an average of 2.8 €[17]. Similar values have been calculated by Böhm (2002), but within a smaller range, between 0.85 and 2.25 €/ha. Böhm (2002) anticipated a mean reduction in N surplus of 60 kg N at a cost of 200-330 €. Comparing the

[17] Values result from a reduction potential of 30 to 120 kg N and cost of 80-200 € per ha and year.

abatement costs of organic farming with other measures, Holländer *et al.* (2008) attest that organic farming has a very high cost-effectiveness, due to its low costs overall.

Based on the limited empirical evidence found in literature it was not possible to draw unequivocal conclusions. However, empirical evidence does exist to indicate that organic agriculture is not always more costly, as theorised by von Alvensleben (1998). Nevertheless, as shown theoretically in Section 2.2.3, it has also become evident in this section that the cost-effectiveness of organic farming may depend substantially on the specific geographic and policy context in which it occurs.

3.5 Conclusions

Organic farming entails a wide variety of environmental effects achieved by a range of measures, including the avoidance of non-renewable resource use, greenhouse gas mitigation, protection of ground and surface water, improvements in species and habitat diversity, and to landscape aesthetics. Furthermore, ethological benefits such as animal welfare and socio-economic benefits, including increased net added value in rural areas and rural employment have been proved by scientific studies and policy evaluations.

According to the studies reviewed above, fossil energy use is generally lower in organic systems both per ha and per tonne of product. However, there are individual products, such as potatoes and meat, which show a higher energy use on organic farms compared to conventional farms. The relative differences between the farming systems are highly variable, depending on the general setting and the geographical context. The major contributor to lower energy use per ha is the reduced use of inputs in organic systems; concentrate feedstuffs and mineral fertilisers in particular are mentioned as major drivers of higher energy use in the literature.

Current meta-studies show significantly higher biodiversity levels in organic farming systems. In particular, arable weeds, birds, predatory insects and spiders benefit from organic practices such as the ban on chemical inputs, lower rates of organic fertiliser and higher shares of semi-natural habitats.

Eutrophication is about 50 % lower in organic systems than in conventional systems, according to international literature. This is attributable above all to lower nutrient loads in the system and in the soil, resulting from the restricted purchase of inputs into the system.

The majority of studies dealt with the environmental impacts of organic farming, whereas there were fewer studies focussing on costs. Furthermore, no single framework is used to determine the types of costs which must be taken into account. The existing body of literature suggests that organic farming imposes a higher burden on public budgets than conventional farms due to higher direct payments, unless external benefits are considered.

Studies comparing the costs and effects of organic farming with those of alternative environmental policies or farming systems are rare. The current literature suggests that organic farming involves the same or higher costs compared with specific agri-environmental measures on conventional farms. However, significant contradictions are apparent among the various studies. Thus the current literature on the cost-effectiveness of organic farming does not provide sufficient information to draw general conclusions about whether organic farming is superior to other farming systems with regard to cost-effectiveness. Moreover, the performance of organic agriculture needs to be analysed in a specific geographical and political context.

4 The Swiss agri-environmental policy framework

This chapter contains an overview of the Swiss agri-environmental policy context. On the basis of the conclusions drawn from the previous two chapters, it reviews the development of the direct payment system, prevalent policy goals and targets, policy instruments and existing evaluation studies. In the first section, the general development of agri-environmental policies in Switzerland is outlined (Section 4.1). Section 4.2 presents the goals of Swiss agricultural policy in terms of the fundamental reasons for its implementation. Furthermore, agri-environmental goals, quantitative targets and the degree of target attainment are discussed. Third, existing agri-environmental policy instruments are examined as a means to attain the envisaged goals and targets (Section 4.3). Fourth, current evaluations of these policy instruments are reviewed in order to generate an overview of causal relations between policy instruments and policy goals (Section 4.4), this constituting an important backdrop to the original analyses that follow.

Hence, this chapter serves as a brief synopsis of the agri-environmental policy context for those readers unfamiliar with the Swiss Direct Payment System. Due to substantial differences between this system and the Common Agricultural Policy (CAP) in terms of policy goals, targets and instruments, this overview is required to contextualise the subsequent modelling analysis.

4.1 Development of agri-environmental programmes in Switzerland

Since 1951, when the Law on Swiss Agriculture came into effect (Moser, 2006), Swiss agricultural policy has been dominated by production incentives with the primary goal of increasing food supply (Gruber, 1992). Growing environmental concerns emerging in the 1970s and 1980s were addressed by scattered prohibitive measures. Organic farming evolved as a niche system constituting the forerunner of environmentally friendly agriculture (Anwander Phan-huy, 2000).

However, as the trade-off between economics and ecology in agriculture became increasingly obvious, a paradigm shift in agricultural policy took place in the early 1990s. The Swiss Direct Payment Scheme was put in place in 1992 as a response to the common international shift from market support policies, which were linked to the output of production via price subsidies, to direct support of farmers, which was tied to land and livestock. Internationally, this change was triggered by the inclusion of agriculture in the GATT[18] negotiations and the substantial amount of surplus production (Henrichsmeyer and Witzke, 1994a; Sciarini, 1996). According to Anwander Phan-huy (2000) and Curry and Stucki (1997), this fundamental change was also pushed by the development and growing occurrence of environmentally friendly alternative farming systems, by general public support in Switzerland for environmentally friendly agriculture, and by the availability of sufficient state funds to manage the transformation process in a socially acceptable way.

With the introduction of cross-compliance[19], the direct payment scheme underwent its first significant reform in 1998. As a consequence of a referendum on the newly defined constitutional mandate setting out the multifunctional character of Swiss agriculture (see Section 4.2.1 below), obligatory minimum standards for farms were implemented. A further change took place in 2001, when the 'Ordinance on Regional Promotion of Quality and Networking of Ecological Compensation Areas in Agriculture'[20] was enacted, introducing an additional result-oriented remuneration scheme for agricultural and nature conservation practices (Moser, 2006). Since 2001, Swiss agricultural policy (AP) has been changing in line with a 4-year policy cycle. However, the agricultural policy reforms 'AP2002', 'AP2007' and 'AP2011', named according to the year of their phasing out, contained only minor adjustments regarding the direct payment scheme[21]. At the same time, existing elements of market support measures and milk quotas were successively cut (Figure 5, page 71) (Mack and Flury,

[18] The General Agreement on Tariffs and Trade (GATT), later transformed into the World Trade Organization (WTO), approved only those subsidies for agriculture which were not or not substantially trade distorting.

[19] Proof of ecological performance (PEP) see Ordinance on Direct Payments, 7 December 1998 (Direktzahlungsverordnung, DZV, SR 910.13 http://www.admin.ch/ch/d/sr/c910_13.html)

[20] Ordinance on Regional Promotion of Quality and Networking of Ecological Compensation Areas in Agriculture 4 April 2001 (Öko-Qualitätsverordnung, ÖQV, SR 910.14, http://www.admin.ch/ch/d/sr/c910_14.html)

[21] Except for an increase in direct payments linked to livestock under AP2011.

2006). Despite this, the OECD reported a high percentage Producer Support Estimate (PSE) for Switzerland of 63[22] in 2006 compared to an OECD mean of 32 (OECD, 2007a).

There is currently an ongoing public debate about the further development of the direct payment system beyond AP2011 (Swiss Federal Council, 2009). As a response to the results of an evaluation of the direct payments system, public authorities have been considering the effectiveness and efficiency of the system (Flury, 2005; Mann and Mack, 2004). In particular, the question of targeting (OECD, 2007d) the policy measures was addressed. The Tinbergen Rule (Tinbergen, 1956) has been discussed as a guiding economic principle for reforming the direct payment system (Mann, 2005a).

4.2 Agri-environmental policy goals and targets

In this section, agri-environmental policy targets are reviewed using legal documents, scientific and grey literature. The first section presents the general goals of Swiss agriculture as formulated in the federal constitution. The second section delineates the quantitative agri-environmental policy targets for energy use, biodiversity and eutrophication in order to investigate the political relevance of potential improvements in each environmental impact category.

4.2.1 Constitutional mandate of Swiss agriculture

Political goals and targets for agriculture can be interpreted as the societal demand for services from the agricultural sector, apart from the production of food and fibres (OECD, 2001a; Olson, 1965). This demand also includes environmental services and the mitigation of negative environmental effects (Schader *et al.*, 2009a). In 1996, federal constitution article 104[23] on an '*ecological and market-orientated agriculture*' was accepted by voters. The article states the following explicit policy targets in paragraph 1 (translated):

[22] The percentage PSE for Switzerland has declined from 77 to 63 between 1986 and 2006.

[23] Swiss Federal Constitution, 18 April 1999 (Bundesverfassung der Schweizerischen Eidgenossenschaft, SR 101, http://www.admin.ch/ch/d/sr/101/index.html)

- *'Contribution to the conservation of natural resources and the maintenance of scenic rural landscapes*
- *Contribution to a decentralised inhabitation of the country*
- *Contribution to a secure food supply for the population'*

Furthermore, paragraph 4 sets out specific criteria for federal government policy measures to enable agriculture to fulfil its tasks (translated):

- *'The government supplements the farm incomes with direct payments for an appropriate remuneration of the services delivered by agriculture, provided that the proof of ecological performance (PEP) is given.*
- *The government supports methods of nature-orientated and animal-friendly production through economic incentives.*
- *The government decrees regulations on declarations of origin, quality and method of food production and processing.*
- *The government protects the environment against environmental damage from fertilisers, chemicals and other substances.*
- *The government may support agricultural research, advice and training and may foster investments.*
- *The government may pass regulations on the stabilisation of farmers' real property.'*

A comparison of the formulations contained in the pre-1999 version and the current constitution shows how they have changed and which additional aspects have been considered post 1999 (Table 7). In particular, two aspects concerning natural resources have been added namely '*Preservation of natural resources and living conditions*' and '*Cultivation of the rural landscape*' (Hediger, 2006). Furthermore, the subsidiary principle was formulated as a separate goal in the 1999 constitution.

Table 7 Overview of goals of Swiss agricultural policy before and from 1999 onwards

Federal constitution (1947-1999)	Federal constitution (1999 onwards)
Regulations in an overall societal interest	Multifunctional tasks of agriculture
Preservation of a viable peasantry	Soil and land cultivating small-farm enterprises
Preservation of productive agriculture	Sustainable and market-oriented agricultural production
Strengthening of peasant property	Strengthening of peasant property
Protection of economically endangered regions	Decentralised settlement of the country
Precautionary measures for war periods	Food security and food safety
	Preserving of natural resources and living conditions
	Cultivation of the rural landscape
	Self-support with subsidiary policy measures

Source: Hediger (2006), adapted

The importance of these relatively novel constitutional goals of agriculture was confirmed by the referendum in 1996. Recent studies also show that these environmental and ethological goals of agriculture still play a prominent role in Swiss society (Tutkun *et al.*, 2007). Further, of 27 items in total, '*compliance with strict animal welfare regulations*', '*particularly strict environmental standards*' and '*upgrading and maintaining ecologically valuable areas*' were ranked 1st, 3rd and 7th respectively (4hm AG and FBM-HSG, 2007).

Of relevance to agri-environmental policy, apart from article 104 on agriculture, are articles 73 on sustainability, 74 on environmental protection, 75 on spatial planning, 76 on water and 78 on nature and homeland protection. They constitute Swiss ecological goals which are not restricted solely to the agricultural sector (Swiss Federal Council, 1999).

The question of agri-environmental policy goals has been discussed in the course of two federal initiatives. First, in the context of planned reforms to the direct payments a system of environmental and socio-economic targets was defined (Vogel *et al.*, 2008). Second, environmental goals for Switzerland are currently defined for each economic sector. For the agricultural sector, this process has already been completed (Aeschenbacher and Badertscher, 2008). The general and agriculture-related policy goals relevant for the environmental impact categories energy use, biodiversity and eutrophication are summarised in Table 8.

Table 8 General and agriculture-related goals of Swiss environmental policy

Environm. category	General environmental goal	Agriculture-related goal
Energy use (in the context of climate change)	Stabilisation of the concentration of greenhouse gases in the atmosphere at a level that will prevent dangerous disturbance of the climate system.	Reduction in carbon dioxide, methane and nitrous oxide emissions from agricultural activities (the elaboration of a climate strategy for agriculture is planned for 2010 by the FOAG).
Biodiversity	Conservation and development of native species and their habitats.	Agriculture contributes substantially to the conservation and promotion of biodiversity. There are three aspects to biological diversity: 1. species and habitat diversity, 2. genetic diversity within species, and 3. functional biodiversity. 1. Agriculture safeguards and promotes those native species and habitat types in their natural range that occur mainly on land used for agricultural purposes or depend on agricultural use. Efforts are made to conserve and foster populations of target species. Efforts are made to conserve and foster populations of character species by making suitable habitats available with sufficient surface area and of the required quality and spatial distribution. 2. Agriculture conserves and fosters the genetic diversity of native wild species found mainly on land used for agricultural purposes. It also makes a substantial contribution to the conservation and sustainable use of native crop varieties and native farm animal breeds. 3. Agricultural production maintains the ecosystem services provided by biodiversity.
Eutrophication with ammonia	1. Precautionary limitation of emissions to the extent that this is technically and operationally possible and economically viable. 2. No excessive ambient pollution, *i.e.* no exceedance of load limits such as ambient air quality limit values, critical loads, critical levels and air quality guidelines. Stricter emission limits to be imposed if, despite precautionary emission control, excessive ambient pollution occurs.	Ammonia emissions from agriculture amount to a maximum 25 000 tonnes of nitrogen per annum.
Eutrophication with nitrates	1. Maximum 25 mg of nitrate per litre in waters that serve as a source of potable water or whose use as such is intended. 2. Reduction in nitrogen input to waters by 50 % from the 1985 baseline.	1. Maximum 25 mg nitrate per litre in waters that serve as a source of potable water, or whose use as such is intended, in cases where the inflow is mainly from agricultural land. 2. Reduction in nitrogen input of agricultural origin to waters by 50 % from the 1985 baseline.
Eutrophication with phosphorus	The oxygen (O_2) content of lakes must not be less than 4 mg per litre at any time and at any depth. It must be sufficient to allow less sensitive organisms to occupy the bottom of the lake all year round and in the most natural possible density (unless there are exceptional conditions of natural origin).	The phosphorus concentration in water bodies should be at a natural level. The total phosphorus content of lakes, in cases where the input is mainly of agricultural origin, is less than 20 µg P per litre (unless there are exceptional conditions of natural origin).

Source: Aeschenbacher and Badertscher (2008); Flury (2005), adapted

As stated in the first line of Table 8, energy use is not explicitly conceived of a goal in terms of its impact on resource depletion but is of indirect relevance in the context of climate change. A more specific agriculture-related goal is expected to be elaborated by 2010. Biodiversity-related goals include species, habitat and genetic diversity. Furthermore, there is a reference to the promotion of functional biodiversity, *i.e.* the maintenance of ecosystem services such as pollination. Concerning eutrophication, distinctions are drawn between ammonia, nitrate and phosphorus nutrient enrichment. The formulation of agriculture-related goals includes quantitative targets, which are discussed in the next section.

4.2.2 Quantitative agri-environmental policy targets

Several public institutions have formulated policy targets in relation to the environment (Aeschenbacher and Badertscher, 2008). An overview of environmental policy targets formulated by the Swiss Federal Office of Agriculture (FOAG) is given in Table 9. It lists indicators for the environmental categories eutrophication, toxicity and biodiversity. The indicators are linked to either responses, such as the amount of ecological compensation area as a proportion of total UAA, or state indicators, such as nitrate levels within drinking water. Baselines, referring to the year 1990-1992, prior to the introduction of the direct payment scheme, are also presented. Finally, quantitative targets, which were set in 2002, are specified (BLW, 1999).

However, the public authorities did not develop any further quantitative targets for the post-2005 period (economiesuisse, 2006). Consistent with the principle of fiscal equivalency (Olson, 1969; Rudloff, 2002; Urfei, 1999), Vogel *et al.* (2008) point out the need to formulate regionalised targets if an externality has an impact at regional level and the demand for this externality differs according to region. More strongly focussed regionalised targets have been called for from the scientific players and are expected to be set for more regionalised policies in the future (Zingg, 2008).

Table 9 Agri-environmental targets of Swiss policy to be achieved in 2005

Issue	Measurement	Baseline	Target 2005
Agricultural process: ecological compatibility	N-balance	96,000 tonnes N (1994)	74,000 tonnes N (23 % reduction)
	P-balance	20,000 tonnes P (1990/1992)	10,000 tonnes P (50 % reduction)
Agricultural practice	Pesticides	2,200 tonnes active ingredient (1990/1992)	1,500 tonnes active ingredient (32 % reduction)
Effects of agriculture on the environment	Ammonia	53,500 tonnes N (1990)	Reduction by 9 %
	Biodiversity	1,080,000 ha agricultural area (1990/1992)	10 % set as ecological compensation areas, including 65,000 ha in the valley region
	Nitrate		90 % of catchments for drinking water with agricultural used watershed below 40 mg/l
Farmer's behaviour	Use of agricultural area	1,080,000 ha agricultural area (1990/1992)	98 % of the area used according to the proof of ecological performance or organic farming

Source: Badertscher (2005), Flury (2005)

Targets for fossil energy use

The Federal Law on Agriculture[24], the Federal Law Relating to the Protection of the Environment[25] and on reducing CO_2 emissions[26] are relevant in the context of formulating targets on energy use. The 'United Nation Framework Convention on Climate Change (UNFCCC)' (UN, 1992b) and the Kyoto Protocol to the UNFCCC (UN, 1998) are relevant international agreements in this context. However, these documents address energy use only indirectly via climate change.

According to the Kyoto Protocol, Switzerland agreed to reduce its greenhouse gas emissions by 8 % compared to the level in 1990 by 2012. CO_2, as the most important greenhouse gas across all sectors, is to be reduced by 10 % by 2010[27]. As there are no specific aims formu-

[24] Landwirtschaftsgesetz (LwG), 29 April 1998, SR 910.1

[25] Umweltschutzgesetz (USG), 7 October 1983, SR 814.01

[26] Bundesgesetz über die Reduktion der CO_2 Emissionen (CO_2-Gesetz), 8 October 1999, SR 641.71

[27] A current revision of the law on reducing CO_2 emissions discusses achieving targets of 20-30 % reduction or 50 % using compensation mechanisms by 2020.

lated for the agricultural sector, the target of the law of reducing CO_2 emissions also applies to agriculture (Aeschenbacher and Badertscher, 2008). Since 1990 the CO_2 emissions from agriculture and forestry have declined by 3 %, while other greenhouse gas emissions have dropped by 7.5 and 13.5 % respectively during the same period.

Quantitative targets for the agricultural sector are, however, of limited applicability, since a) there are no specific policy measures addressing the energy consumption or the energy efficiency of agricultural production, b) even tax relief is available for fuel consumption in agriculture and c) a quantitative target would not take into account the great need for action and the high uncertainty of target concentration of CO_2 in the atmosphere (Hepburn, 2007). For this reason, the Swiss Federal Council (2009) also stresses that the general quantitative target is irrelevant for the agricultural sector. However, a climate strategy for Swiss agriculture, which is expected to specify targets, is currently being developed by the FOAG.

Targets for biodiversity

At national level, the Federal Law on Agriculture, the Federal Law relating to the Protection of the Environment and the Law on Protection of Nature and the Landscape[28] are relevant in the biodiversity context. In addition to national legislation, relevant international treaties in the context of biodiversity are the Convention on Biological Diversity (CBD) (UN, 1992a) and the Bern Convention[29].

There are both regional and national area-related and quality-related goals addressing genetic, species and habitat diversity. Table 10 shows the targets formulated by public authorities. These targets have been only partly met. While the overall amount of ecological compensation areas (ECAs) was already achieved in 2000, the share of high-quality areas, particularly in the lowland region, is still below target. As the low number of endangered species on ECAs indicated, the decline in the number of red-list species could not be stopped (Herzog and Walter, 2005).

[28] Natur- und Heimatschutzgesetz 1 July 1966, NHG, SR 451

[29] Council Decision 82/72/EEC of 3 December 1981 concerning the conclusion of the Convention on the conservation of European wildlife and natural habitats (Bern Convention)

Table 10 Environmental targets for 2005 with respect to biodiversity

Target formulation by	Target	Target attainment
Federal Chancellery (2002)	10 % of total Swiss utilised agricultural area (1,080,000 ha) represents ecological compensation area	Target attained in 2000 In 2003: 116,000 ha
Federal Chancellery (2002)	65,000 ha ecological compensation areas in the lowland region	Target not attained In 2003: 57,000 ha
BUWAL (1998)	In the foreseeable future 65,000 ha utilised agricultural area in the lowland region is managed as **high quality** (BUWAL, 1998) ecological compensation area	Goal not yet attained, estimate for 2003: 20,000 ha
BUWAL (1998)	This promotes the conservation of native species diversity	Generally, more species and more demanding species on ecological compensation areas than on intensively managed land, though quality of ECA is often inadequate
BLW (1999)	Promotion of natural species diversity	
BLW (1999)	No further species losses (Red List), spread of endangered species	Only few endangered species on ecological compensation areas

Source: Herzog and Walter (2005)

Targets for N and P eutrophication

Relevant national laws in the context of eutrophication with nitrogen and phosphorus include the Federal Law on Agriculture, the Federal Law relating to the Protection of the Environment and the Federal Law on Water Pollution Control[30]. Internationally, the Gothenburg Protocol (UNECE, 1999), the 'OSPAR Convention' (OSPAR Commission, 1992), and the 'Convention on the Protection of the Rhine' (ICPR, 1992) are relevant for this environmental impact category.

Table 11 lists quantitative targets that have been formulated for the category of eutrophication in the context of evaluation of ecological direct payments. The targets address the national input-output inventories of nitrogen and phosphorus, the actual pollution of ground (nitrogen) and surface waters (phosphorus) and the total amount of ammonia emissions (Herzog and Richner, 2005). While ammonia emissions were drastically reduced during the period under evaluation, nitrogen budget surpluses could be reduced by only 15 %, meaning that the target of a 33 % reduction could not be met. Targets for reducing nitrate concentrations were achieved, however, partly due to closing down the most problematic wells instead of reducing the nitrate leaching from agricultural fields. The phosphorus surplus in the national input-

[30] Gewässerschutzgesetz, 24 January 1991, GSchG, SR 814.20

output inventory was lowered significantly. However, the reduction in P pollution caused by agriculture was below the target reduction of 50 % (Herzog and Richner, 2005).

Table 11 Environmental targets for 2005 with respect to eutrophication with nitrogen and phosphorus

Target formulation by	Target	Target attainment
BLW (1999)	33 % reduction in excess nitrogen in national input-output inventory	Only approximately 15 % reduction by 2004
Federal Chancellery (2002)	9 % reduction in ammonia emissions compared with 1990, *i.e.* a reduction of around 4,800 t N	20 % reduction already in 2000
BLW (1999)	5 mg/l cut in NO_3 pollution of selected, overall representative ground- and spring-water wells	Reduction in 3-4 mg NO_3/l by 2002/2003
Federal Chancellery (2002)	Nitrate content less than 40 mg/l in 90 % of the drinking water wells fed by areas used for agriculture	Goal attained 2002/2003
BLW (1999), Federal Chancellery (2002)	50 % reduction in excess phosphorus in national input-output inventory	Goal attained 1996, 2002 65 % reduction
BLW (1999)	50 % reduction in agriculturally caused P pollution of surface waters	Reduction only 10 to 30 % at most

Source: Herzog and Richner (2005)

4.3 Swiss agri-environmental measures

Despite the wide variety of types of policy instruments available (Section 2.2.3), agri-environmental problems in Switzerland are addressed predominantly using agri-environmental measures (OECD, 2004) and a cross-compliance regulation akin to arrangements in the EU (Nitsch and Osterburg, 2005).

Figure 4 gives an overview of the instruments employed within the Swiss direct payment scheme as part of the agri-environmental focus. While all of the policy instruments are discussed in the analysis that follows, the instruments marked in grey were selected explicitly for the cost-effectiveness comparison[31].

If farms fulfil the criteria of the proof of ecological performance (PEP), they are eligible to receive general direct payments. If further voluntary criteria specific to certain measures are fulfilled, farms are eligible to receive further payments on top of the general payments. All policy measures can be adopted cumulatively, *i.e.* the uptake of a measure does not impair

[31] Criteria for the selection of these policy measures are discussed in Section 6.3.6

eligibility for taking up other measures and the payment rates are added for each hectare and for each livestock unit (LU).

The following sections describe especially those agri-environmental measures that are relevant to the subsequent analysis. Each instrument is presented in terms of its eligibility criteria, payment levels, uptake levels and total public expenditure, while only those measures relevant to the later subsequent analysis are described in detail.

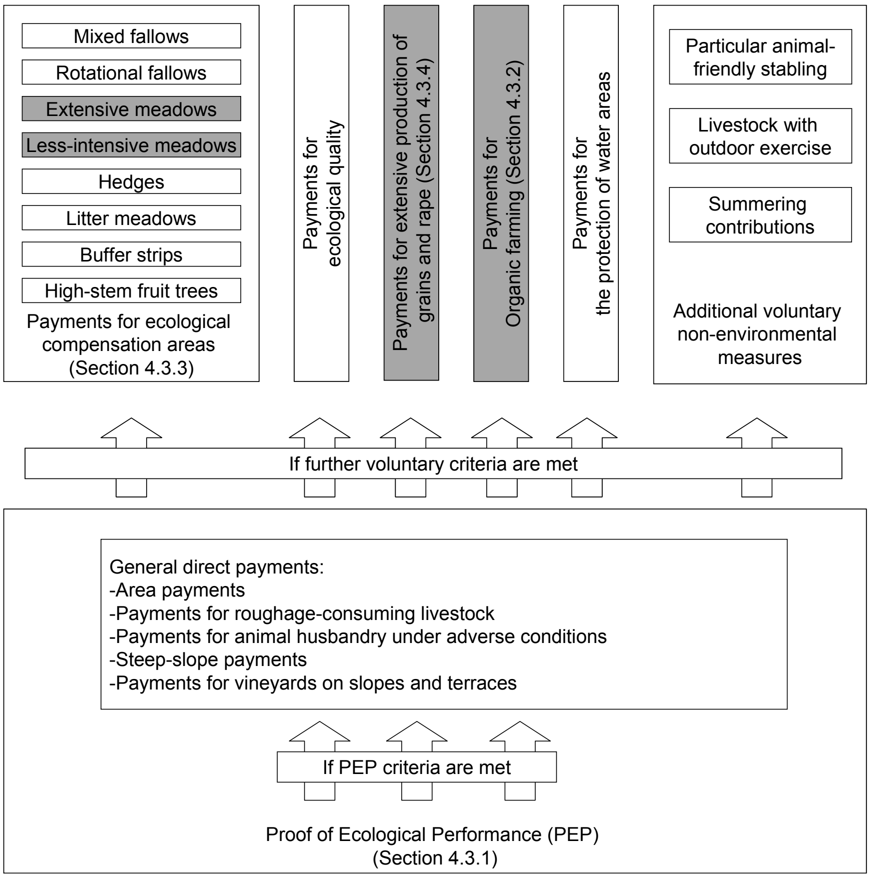


Figure 4 Design of Swiss ecological direct payments

4.3.1 Proof of ecological performance

The Swiss general direct payments are granted as compensation to farmers for rendering multifunctional services to society (Swiss Federal Council, 2009). The proof of ecological performance (PEP) is the cross-compliance regulation for Swiss agriculture.

Eligibility criteria

PEP rules are defined in Article 70 of the Federal Law on Agricultural as a general entry-level standard for obtaining direct payments. Thus the PEP rules follow a pure 'Red Ticket' approach (Baldock and Mitchell, 1995). They comprise the following main criteria:

- Animal-friendly husbandry: Compliance with the animal protection ordinance.
- Balanced nutrient budget: Nutrient balance may show a surplus of only 10 % with respect to nitrogen and phosphorus.
- Appropriate share of ecological compensation areas (ECA): Minimum share of ecological compensation areas in utilised agricultural area (UAA) 3.5 % for speciality crops (fruits, vegetables, vine) and 7 % for other UAA.
- Organised rotation: If the arable land of a farm exceeds 3 ha, at least four different crops have to be cultivated per year; the maximum amount of crops or fallows has to be adhered to.
- Suitable protection of the soil: This covers the use of winter crops, catch crops, or green manure, if arable land of a farm exceeds 3 ha. No periodical soil erosion is permitted.
- Selection and targeted application of pesticides: Including restrictions for pre-emergence herbicides, granulates and insecticides, consideration of early-warning systems and pest forecasts and a four-year test interval for sprayers.

Thus compared to cross-compliance in most EU Member States, the Swiss PEP requirements can be regarded as being even stricter. One exception is the obligatory maintenance of grasslands in the EU, which is not included in Swiss PEP requirements (Nitsch and Osterburg, 2005).

Apart from the PEP requirements, there are eligibility criteria regarding farm size, farmer's age, income, property and minimum workload. Although the PEP rules are officially voluntary, *de facto* they are binding for those farms which depend economically on direct payments. Only very small and very large farms do not come under the PEP rules (Mann and Mack, 2004).

Payment levels

Fulfilling the cross-compliance rules permits farmers to receive general and ecological direct payments, although the latter have further obligatory criteria. The payment levels of the general and ecological direct payments are stated in the ‘Ordinance on Direct Payments’ (DZV). The payments are linked to either UAA or livestock units (LU). The development of payment levels for general direct payments is listed in Annex A, Table 67 and Table 68.

Uptake

The percentage of farms compliant with the PEP has increased from its introduction up until 2007 to nearly 100 %, while a figure share of more than 90 % was reached in 1999 (BLW, 2008).

Public expenditure

An overview of public expenditure on direct payments and other policy instruments for the period between 1990 and 2007 is presented in Figure 5. Total expenditure rose up until 1996 and has remained since then on a relatively constant level, ranging from 3.5 (in 1990) to 4 billion CHF (in 1999). Between 1990 and 2007, the amount of direct payments as a proportion of the budget increased substantially, while expenditure on production and sales decreased. Expenditure on basic improvements remained at a constant level. Expenditure on administration is relatively minor. It should be stressed that from a welfare economic point of view, the reduction in expenditure on production and sales and the tariff reduction should have led to substantial efficiency gains, due to an increased consumer surplus (Henrichsmeyer and Witzke, 1994b).

Table 12 shows the development of direct payment expenditure from 2000 to 2007. During this period, both general and ecological direct payments increased. Among general direct payments, the largest increase was found for RGVE payments, while among ecological direct payments, public expenditure on OFASP, ÖQV, and ethological direct payments in particular rose noticeably. Public expenditure on ‘extenso payments’ (see Section 4.3.4 for more details) decreased marginally.

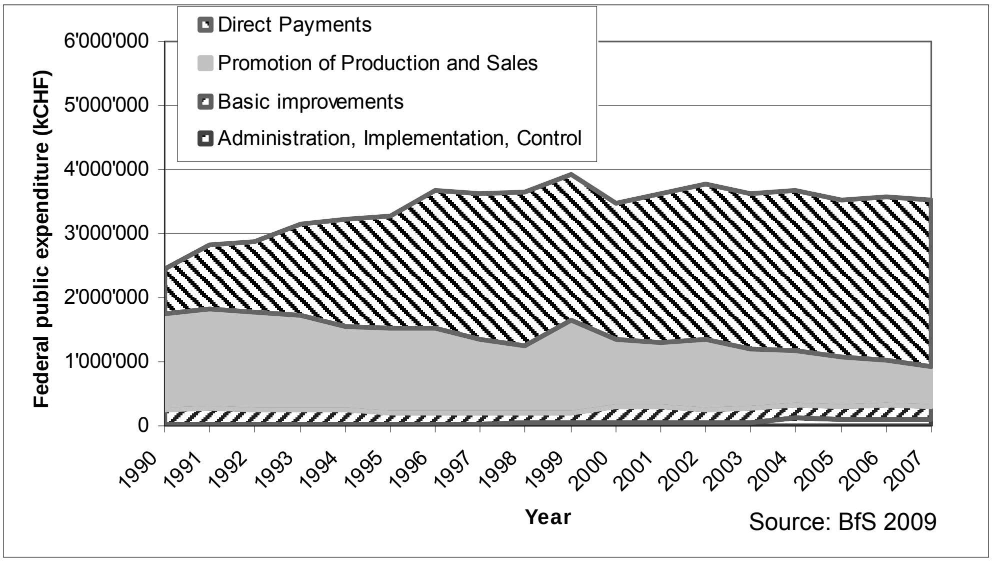


Figure 5 Federal budget spending on the general and ecological direct payments, basic improvements, and production and sales between 1990 and 2007

Table 12 Public expenditure on direct payment measures between 2000 and 2007

Policy measures	2000	2001	2002	2003	2004	2005	2006	2007
General direct payments	**1,803,658**	**1,929,094**	**1,994,838**	**1,999,091**	**1,993,915**	**1,999,606**	**2,007,181**	**2,070,357**
Area payments	1,186,770	1,303,881	1,316,183	1,317,956	1,317,773	1,319,595	1,319,103	1,275,681
RGVE payments	258,505	268,272	283,221	287,692	286,120	291,967	301,213	412,813
TEP payments	251,593	250,255	289,572	287,289	284,023	282,220	281,258	277,786
Hillside payments	96,714	96,643	95,811	95,630	95,308	94,768	94,227	92,671
Hillside payments for vineyards	10,076	10,043	10,051	10,524	10,691	11,056	11,380	11,407
Ecological direct payments	**361,309**	**412,664**	**452,448**	**476,724**	**494,695**	**506,895**	**518,211**	**523,533**
Eco payments	278,981	329,886	359,387	381,319	398,109	409,348	420,245	425,533
ECA payments	108,130	118,417	122,347	124,927	125,665	126,023	126,976	126,928
ÖQV payments	-	-	8,934	14,638	23,007	27,442	30,256	32,107
Extenso payments	33,398	32,526	31,938	31,255	30,824	31,516	31,094	30,529
OFAS payments	12,185	23,488	25,484	27,135	27,962	28,601	28,672	28,074
Ethological direct payments	108,118	155,455	170,684	183,363	190,651	195,767	203,247	207,796
Summering payments	81,238	80,524	89,561	91,381	91,066	91,610	91,696	92,110
Water protection contributions	1,090	2,254	3,500	4,024	5,521	5,936	6,270	5,890
Buget cuts	22,542	16,763	21,143	17,138	18,120	20,378	25,820	18,851
Total direct payments	**2,142,425**	**2,324,995**	**2,426,143**	**2,458,677**	**2,470,490**	**2,485,758**	**2,499,572**	**2,575,039**

Source: FOAG 2008

4.3.2 Organic farming area support payments

Organic farming area support payments are the object of this thesis. Despite the existence of a wide variety of other instruments for supporting the development of organic production, area payments are the most prevalent policy measure both in the EU (Stolze and Lampkin, 2009) and in Switzerland (Tuson and Lampkin, 2007). For this reason, this policy measure is explained in more detail than the other instruments. Further statistics regarding the structural differences between organic and non-organic farms are analysed in detail in Section 7.2.1 as a basis for interpreting the results of the subsequent modelling analysis.

The organic support scheme is based on the Federal Law on Agriculture, Articles 70, 72-76 and 177 and on the 'Ordinance on Direct Payments for Agriculture'[32], Articles 57-58 as well as 'Ordinance on Organic Farming[33]', Articles 3, 6-16, 38-39.

Eligibility criteria

Since 1999 all organic farms have led to cultivate their holding in compliance with the PEP criteria (see 4.3.1). In addition to the PEP, organic farms have to comply with the 'Ordinance on Organic Farming' (see Table 13) (BLW, 2004; EVD and BLW, 2004), which is equivalent to the original Council Regulation (EEC) No 2092/91 defining organic farming.

Payment levels

The payment levels for organic agriculture are delineated in the 'Ordinance on Organic Farming' (SR 910.18). Since 2001 annual payment levels have been 200 CHF/ha for grass-land, 800 CHF/ha for arable land, 1200 CHF/ha for speciality crops (fruits, vegetables and vine) (EVD, 1997). In the period from 1998 to 2000 the payment levels were slightly lower (arable land 800 CHF; forage and fodder crops 100 CHF; speciality crops 1000 CHF). However, between 1996 and 1998 payments levels were substantially higher (arable land 1400 CHF; forage and fodder crops 530 CHF; speciality crops 1800 CHF). According to Schmid (1999) and Schader and Schmid (2005), additional cantonal payments were granted during several years.

[32] Direktzahlungsverordnung, SR 916.344

[33] Bioverordnung, SR 910.18

Table 13 Selected eligibility criteria for organic producer payments in 2009

Farmers eligible to participate	Both converting and continuing farms are eligible.
Organic certification requirement	By a certification organisation accredited according to Swiss criteria and also to ISO 14000.
Fertilisation	Mineral fertiliser banned, except rock phosphate. Maximum organic fertilisation: corresponding to 2.5 livestock units (LU) per ha.
Ecological compensation areas	For speciality crops, 7 % ECA is required, instead of 3.5 % according to PEP
Seeds	Only organic seeds allowed. Exceptions are made if organic seeds are not available on the market
Synthetic fungicides and insecticides	Banned, only natural substances allowed
Herbicides	Banned
Eligible crop restrictions	All crops and types of livestock are eligible, whole farm approach
Sewage sludge	Banned
Feedstuffs	No GMO fodder allowed. In contrast to the EU legislation, ruminants have been allowed to be fed up to 20 % conventional feed. Full compliance with EU legislation is required since May 2009.
Organic management of livestock	In the three relevant private standards in Switzerland (Bio Suisse, Migros-Bio and Demeter), animal husbandry is an integrated part of certification. However, the 1997 regulation contained only a small section about animal husbandry, referring to generally recognised organic standards (without details). According to Bio Suisse and Migros-Bio, as of 1999 all organic farms have to keep their livestock according to the DEA Ordinance on Regular Outdoor Access for Livestock. Rabbit husbandry has to comply with payments for particularly animal-friendly stabling.
Staged (gradual) conversion possible	Max. 5 years – Bio Suisse, Migros-Bio, and national regulation from 1998. Gradual conversion is possible for animal husbandry only.
Staged conversion possible	None of the certification programmes (Bio Suisse, Migros-Bio, and Demeter) allow staged conversion By contrast, the current federal ordinance from 1998 (adapted in 2008) does allow staged conversion, except in two cases: a) Vineyards can be converted to organic without converting the whole farm; b) An orchard or vineyard (not restricted to a certain minimum size) may not be converted to organic farming.
Training and/or advice provided	Training in organic agriculture is optional, with two exceptions: 1. Some cantons (like Grison) with conversion subsidies require attendance at an introductory course in organic agriculture. 2. Since 1997, Bio Suisse demands that a new organic farm follows a 2-day introductory course in order to be certified.
Other restrictions	Compliance with environmental law (e.g. maximum 4 kg/ha copper use per year) and the soil protection regulation.

Source: own compilation based on EVD (1997), Schmid (1999) and Schader and Schmid (2005)

Uptake

The number of farms and the area under organic farming are illustrated in Figure 6. The graph shows the area and number of privately labelled farms and the number of farms and hectares receiving support under OFASP. Before 1993, when the OFASP were introduced, less than 1,000 farms were working according to organic standards. The increase from 1980 to 1993 was even flatter than displayed, because the statistics prior to 1989 cover only Bio Suisse farms. In 1989 and 1992 farms from other labels were added to the statistics. After 1993, both the number of organic farms and hectares under organic farming rose quickly until they peaked in 2004 (private-standards farms) and 2005 (OFASP farms). This steep rise was induced by several cantonal support payments, which had already started prior to 1996, and the federal OFASP, which started in 1996. A strong increase in demand for organic produce also accelerated the sharp growth, which continued until 2004. After this peak, the number of farms declined moderately, which can be attributed to general structural changes in Switzerland (Bio Suisse, 2009), as the number of non-organic farms also declined. The area cultivated organically is currently stagnating (BfS, 2009).

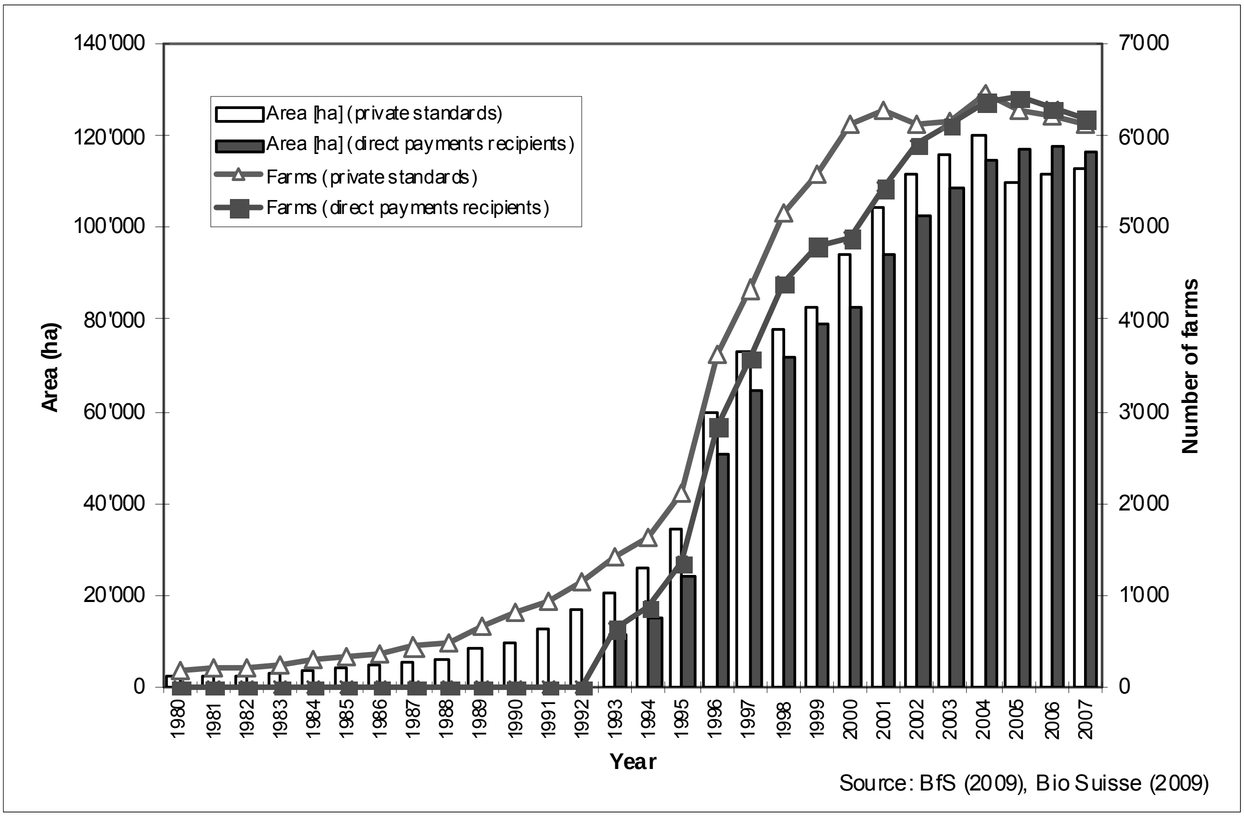


Figure 6 Uptake levels of organic farming area support payments compared to privately labelled farms indicated by area size and number of farms between 1980 and 2007

The spatial distribution of organic farms is heterogeneous. The number of organic farms in mountain regions is much higher than in the lowlands and has been increasing more dynamically since 1997 (Figure 7). Moreover, dairy and beef farms are represented above average.

Public expenditure

The direct payments dedicated to OFASP amounted to 28,074 kCHF in 2007 and represented a share of 1.15 % (2006) of total public expenditure on direct payments. The share varied during the period from 1999 to 2007 between 0.6 and 1.15 % (3.67 (1999) and 5.69 % (2003)) in total ecological direct payments.

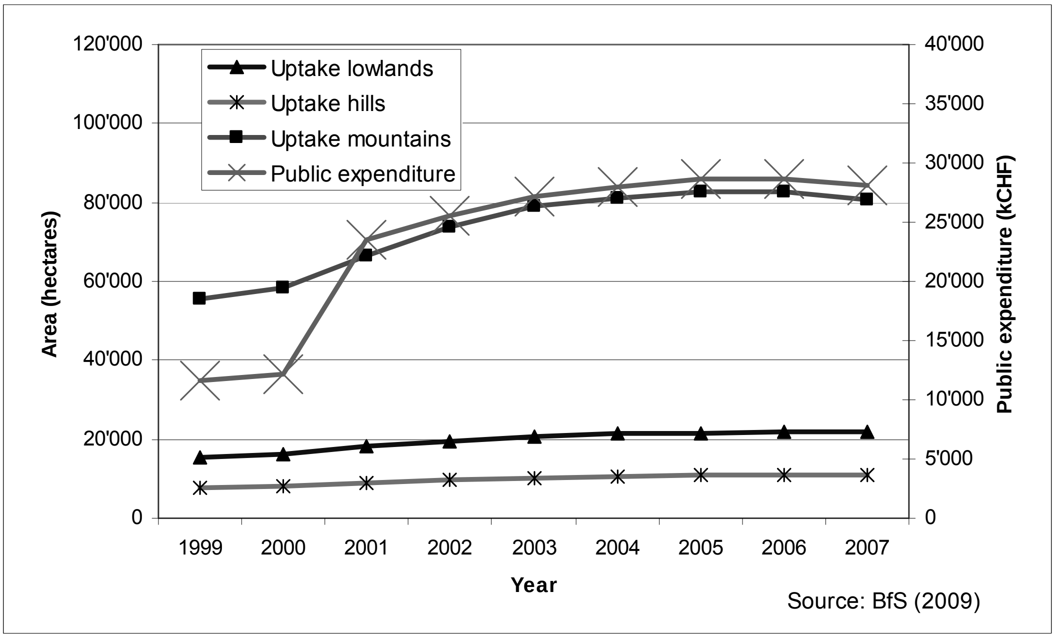


Figure 7 Regional uptake of and public expenditure on organic farming area support payments between 2000 and 2007

4.3.3 Ecological compensation areas

Ecological compensation areas comprise a set of 16 measures. An overview of ecological compensation areas shows the wide portfolio of measures (Annex A, Table 70). The measures are grouped into ECA-A and ECA-B measures. All ECA measures are applicable to the PEP requirement of 7 % of ECA in UAA. The implementation of ECA-B measures leads to additional direct payments. In terms of area, ECA-A measures are much less prevalent than ECA-B measures. ECA-A measures do not give rise directly to additional taxpayer costs, they entail no system-relevant differences between organic and non-organic farms and they are thus less relevant to the research questions of this thesis. For this reason, only ECA-B measures are discussed in the subsequent analysis.

Eligibility criteria

Table 14 summarises the eligibility criteria for the ECA-B measures.

While buffer strips, rotational and mixed fallows are applicable on arable land, the other measures can be taken up on grassland only. Extensive meadows, hedges and ‘extensive meadows on wet sites’ need to be implemented on the same area for at least 6 years, while the locations of other measures can be chosen flexibly. The principal distinction between less intensive and extensive meadows lies in the fertilisation restrictions. On extensive meadows, fertilisation is strictly banned, while on less intensive meadows organic fertilisation is permitted to a limited extent.

Payment levels

The payment levels for ECA measures are listed in Table 14. The payment levels have been determined by FOAG according to farm-level opportunity costs arising from the implementation of the measures. Farm-level opportunity costs comprise decreased production values, higher costs (e.g. labour) and additional administrative work. Due to the regional differences in opportunity costs, payment levels are differentiated regionally for extensive meadows, less intensive meadows, hedges and ‘extensive meadows on wet sites’. The highest payment rates can be found in the lowlands. The lowest payment rates can be found in the mountain region. Mann (2003a) estimated that payment levels are, by trend, higher than farm-level costs, even if farm-level transaction costs (additional administration) are taken into account.

Table 14 Eligibility criteria and payment levels for ecological compensation areas in 2007

ECA-B measure	Land type	Payment level (CHF/ha)	Restriction
Extensive meadows	Grass-land	Lowlands: 1500 Hills: 1200 Mountains: 450-700	• Area must be cultivated according to the standards for at least 6 years • Harvest must be used as fodder • No fertilisers or crop protection measures (except for single crop treatments) • At least one cut per year not before 15 June (low-lands), 1 July (lower mountains), 15 July (higher mountains) • Autumn grazing between 1 September and 30 November
Less intensive meadows	Grass-land	Lowlands and hills: 650 Mountains: 300-450	• No crop protection measures (except for single plant treatments) • Nitrogen fertilisation with manure and compost • Slurry only after first cut (max. 15 kg N/ha and application; max. 30 kg N/ha per year) • Cutting dates as for extensive meadows
Hedges	All	Lowlands: 1500 Hills: 1200 Mountains: 450-700	• Area must be cultivated according to the standards for at least 6 years • Harvest must be used as fodder • No fertilisers or crop protection measures • 3 m wide buffer strips on both sides (must be mowed at leased every 3 years) • Cutting dates like for extensive meadows
Extensive meadows on wet sites	Grass-land	Lowlands: 1500 Hills: 1200 Mountains: 450-700	• Area must be cultivated according to the standards for at least 6 years • Must be cut, but only once every 1-3 years • Harvest must be used as fodder • No fertilisers or crop protection measures • First cut not before 1 September
Mixed fallows	Arable land	3000	• After arable or permanent cultures in valley regions • Minimum 3m wide • No fertilisers or crop protection measures • Minimum 2, maximum 6 years on the same plot • Ploughing up not before 15 February
Rotation fallows	Arable land	2500	• Minimum 6 m wide • Sowing between 1 September and 30 April • Cultivation until 15 February (for 1-year fallow) and 15 September (for 2-year fallow) • No fertilisers or crop protection measures • Cutting only between 1 October and 15 March
Buffer strips	Arable land	1000	• Minimum 3 m maximum 12 m wide • Minimum 2 years • No insecticides and no nitrogen-containing fertilisers • Sown on the whole long side of arable crops • Grains must be harvested when ripe
High-stem fruit trees	All	15 CHF per tree	• Fruit, chestnut or nut trees • Minimum stem length for stone fruits 1.2, for other trees 1.6 m • No herbicides around the stem (exception for trees younger than 5 years) • At least 20 trees on the farm • Trees in fruit plantations are not eligible

Source: 'Ordinance on Direct Payment for Agriculture' (EVD, 2008)

Uptake

Uptake levels of ecological compensation area measures have been growing steadily since their introduction. However, individual ECA measures are taken up by farmers to different degrees. The most widely adopted measures are extensive meadows, less intensive meadows and high-stem fruit trees[34]. The most substantial increase in uptake levels can be observed for extensive meadows[35], rising from an estimated 17,000 ha in 2003 to 55,000 ha in 2006 (Figure 8). By contrast, the uptake of less intensive meadows amounted to 31,038 ha in 1993, peaked at 42,344 ha in 1998, and fell constantly after that to 30,693 ha in 2007. The number of high-stem fruit trees remained about constant during the whole period, with a slight increase until 1998 and an even slighter decline until 2006 (FOEN, 2008). More details on current ECA uptake levels, including differences between farm types and farm groups are given in the context of the model analysis in Section 7.2.3.

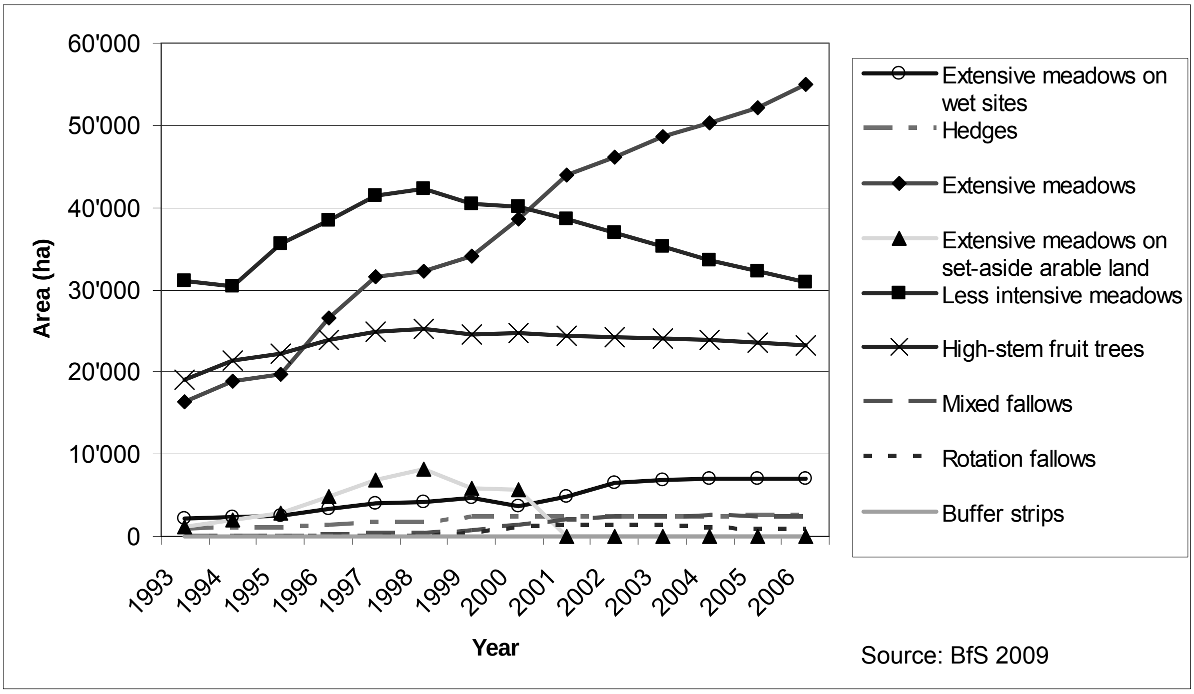


Figure 8 Uptake levels of ECA-B measures between 1993 and 2006

[34] For the calculation a high-stem fruit tree was set equivalent with 1/100 ha of ECA.

[35] Aggregated numbers were available only for the measures 'extensive meadows', 'extensive meadows on wet sites' and 'hedges'.

Payments for extensive meadows on arable set-aside were abolished in 1998 and the remaining areas dropped to zero ha in 2001. Extensive meadows on wet sites increased from 2,000 ha to 7,000 ha, and hedges from 1,000 to 2,500 ha. The uptake of both types of fallows increased while rotational fallows were introduced only in 1999, mixed fallows have been available since 1993. The uptake levels of both measures experienced their steepest rise between 1999 and 2002. Since then the uptake of fallows has levelled off or declined slightly. Figure 8 makes the substitution of less intensive meadows by extensive meadows appear likely.

Public expenditure

Public expenditure on ecological compensation measures amounts to 8 % of total expenditure on direct payments. The most costly measures are payments for extensive, high-stem fruit trees and less intensive meadows. While the expenditure on extensive meadows increased by 75 % between 1999 and 2007, expenditure on high-stem fruit trees decreased in the same period by 7 % (Figure 9). More details on public expenditure on ECA measures are presented in the context of the model analysis in Section 7.2.7 below.

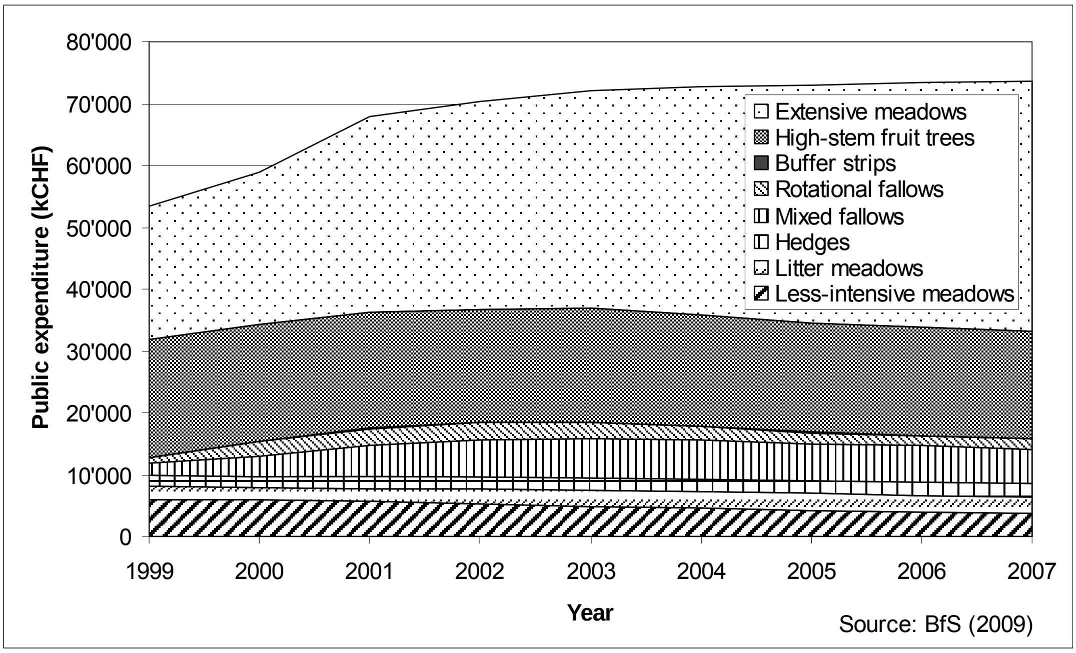


Figure 9 Public expenditure on selected ecological compensation area measures between 2000 and 2007

4.3.4 Extenso payments

'Extenso payments' refer to the extensive cultivation of grains and rape. Extenso payments were introduced for grains in 1991 and carried over to rape in 1996 (Gaillard and Nemecek, 2002).

Eligibility criteria

The eligibility criteria for extenso payments include a ban on growth regulators, fungicides, chemical-synthetic stimulators of natural resistance and insecticides. Due to the cumulative principle of the direct payment system, organic farms are automatically eligible for extenso payments because the eligibility criteria are already covered by the standards for organic farming (EVD, 1997). A full farm branch, *i.e.* a type of grain, needs to be converted to extenso production but not the complete portfolio of grains grown on the farm.

Payment levels

Extenso payment generate an additional annual payment of 400 CHF/ha for the farmers.

Uptake

About 50 % of the grains grown in Switzerland are grown according to extenso standards. While the share is lower for bread cereals, it is higher for fodder cereals. The extenso share for rape is about 25 % (Gaillard and Nemecek, 2002). The uptake of extenso payments is highest in the lowlands due to the larger amount of arable land there. Since 1999 uptake levels have been decreasing slightly in the lowlands (6 %), moderately in the hill-region (20%) and drastically in the mountain areas (35 %) (FOEN, 2009). More details on current extenso uptake levels, including differences between farm types and farm groups are given in the context of the model analysis in Section 7.2.3.

Public expenditure

Corresponding to decreasing uptake levels, public expenditure on extenso payments has been steadily decreasing from 35,136 kCHF in 1999 to 30,629 kCHF in 2007 (FOAG, 2008;

FOEN, 2009) (see Figure 10). More details on public expenditure on extenso payments are presented in the context of the subsequent model analysis in Section 7.2.7 below.

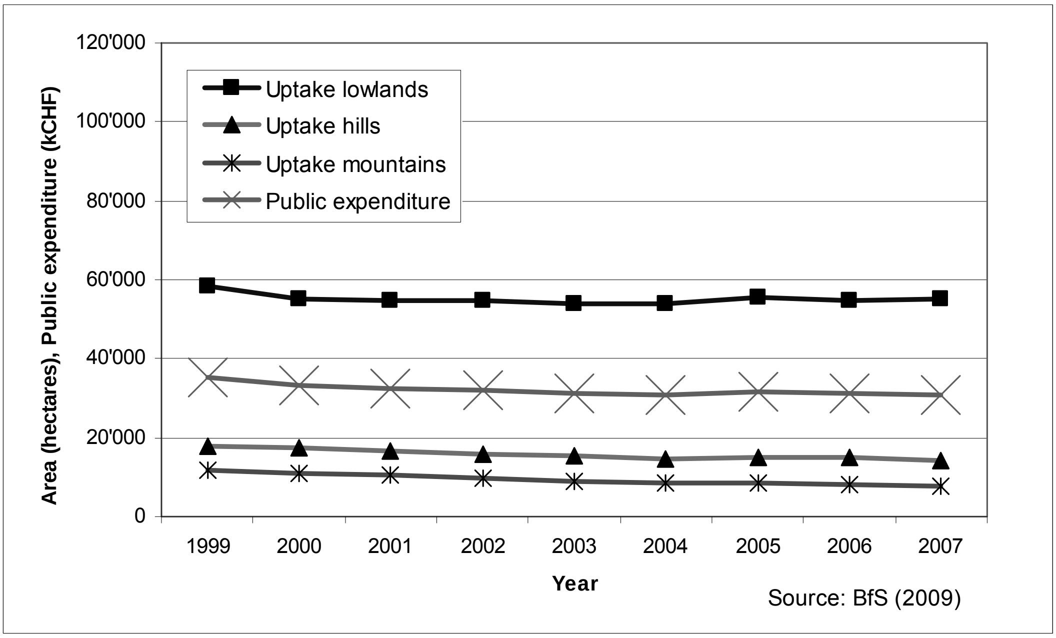


Figure 10 Regional uptake of and public expenditure for extenso payments between 1999 and 2007

4.3.5 Other ecological and ethological policy measures

Apart from the measures included in the 'Ordinance on Direct Payments for Agriculture'[36], which have been described in detail above, the 'Ordinance on Regional Promotion of Quality and Networking of Ecological Compensation Areas in Agriculture'[37], the 'Ordinance on Summering Contributions'[38], the 'DEA Ordinance on Animal Friendly Housing Systems'[39] and the 'DEA Ordinance on Regular Outdoor Access for Livestock'[40] are in place. These

[36] Direktzahlungsverordnung, DZV, SR 910.13

[37] Öko-Qualitätsverordnung, SR 910.14

[38] Sömmerungsbeitragsverordnung, SöBV, SR 910.133

[39] BTS-Verordnung, SR 910.132.4

[40] RAUS-Verordnung, SR 910.132.5 merged with BTS Verordnung to Ethoprogrammverordnung (SR 910.132.4)

policies are not the main focus of this analysis and are therefore not described in detail in this section.

4.4 Results of preceding evaluations of Swiss direct payments

This section summarises the findings of the evaluations of Swiss direct payments in terms of their impacts on energy use, biodiversity and eutrophication. It additionally presents the relevant results of the economic evaluations.

The evaluation of Swiss agricultural policy is based on articles 185, 115 and 116 of the Federal Law on Agriculture. The 'Ordinance on the Evaluation of Sustainability of Agriculture'[41] states that agri-environmental and ethological measures must undergo a periodical evaluation, which the Federal Office of Agriculture (FOAG) bases on a variety of data sources (BLW, 1999). Tissen (2009), comparing the evaluation procedures of agri-environmental policy in Switzerland with those in Germany and Austria, concludes that Switzerland opted for a top-down and long-term mode of evaluation of overall agricultural policy. In Austria and Germany, by contrast, evaluation is shared among several institutions due to strong political influences and the division of competencies for different policies among different institutions. From an *ex-post* perspective, the Swiss evaluations have been ambitious in size and in the number of different projects examined. While it has been possible to identify clearly the impacts of agri-environmental policy change as a whole, the evaluation of individual measures and causal relations has been achieved only to a limited extent (Jung, 2009). Compared to German and Finnish evaluations of agri-environmental policies, the Swiss evaluations concentrated on environmental effectiveness, while aspects of economic efficiency were neglected (Brower, 2004).

The results of those evaluations being relevant for this thesis in terms of instruments and environmental category are analysed in the following sections according to the environmental categories of energy use, biodiversity and eutrophication. Finally, the results of relevant economic evaluations of the direct payment scheme are summarised.

[41] Verordnung über die Beurteilung der Nachhaltigkeit in der Landwirtschaft, 7 December 1998, SR 919.118

4.4.1 Energy use

There are no official evaluations of agricultural policy regarding energy use. Even for the environmental category of climate change effects no evaluations are available due to the lack of policy targets. Nevertheless, monitoring has identified a 10 % reduction in greenhouse gases between 1990 and 2006 which, however, results mainly from structural change, *i.e.* the decline in agricultural holdings and livestock, rather than from gains in energy efficiency. Furthermore, the reduction can be attributed to declines in N_2O and methane emissions by 13.5 and 7.5 %, respectively, while CO_2 emissions declined by only 3 %. The main part of the reduction was caused by major shifts in agricultural policy between 1992 and 2002, from production incentives to direct payments. In the AP2007 period, no significant decline is apparent (Aeschenbacher and Badertscher, 2008).

Life cycle assessments of most of the area-related policy measures have been performed by Agroscope-Reckenholz-Tänikon (ART)[42] with energy use as one indicator among others. The results show that 'extenso payments', ECA measures and organic farming lead to lower energy use per ha. Organic farming reduces energy use per ha, depending on the specific crop, by 42-57 % compared to intensive PEP-compliant conventional production, while extenso payments reduce energy use per ha by 7-32 %. Due to lower yields, however, the energy use efficiency of crops grown under the extenso scheme is generally negatively affected (by up to 18 %). Crops grown organically are generally affected positively (up to 41 %) except for potatoes grown in the lowlands, which have a 10 % higher fossil energy requirement per tonne (Nemecek *et al.*, 2005).

By contrast, payments for animal husbandry in adverse conditions, payments for roughage consuming livestock, and payments for particularly animal friendly stabling and livestock with outdoor exercise represent instruments that have an upward impact on energy use, due to their incentivising effect on the number of animals kept (Swiss Federal Council, 2009). Using the SILAS model, Mack *et al.* (2007) found a higher energy use in the lowlands than in hills and mountain areas. Employing a multi-objective optimisation, they concluded that energy input is more efficiently used for animal production and in the hill and mountain areas than it is in the lowlands and for crop production activities respectively. Mack *et al.* (2007) sug-

[42] Research Station of the Federal Office of Agriculture (FOAG)

gested the introduction of energy taxes, particularly for greenhouses, tariffs for imported feedstuffs, and incentives for an extensification of crop production as policy measures adequately addressing the energy use.

4.4.2 Biodiversity

Official evaluations of the PEP and ecological compensation areas regarding biodiversity proved the positive effects of the PEP rules on biodiversity. The most important component of the PEP is the requirement that each farm needs to declare at least 7 % of its UAA as ECA (3.5 % for speciality crops). Biodiversity monitoring projects showed clearly that general species and demanding species are more abundant on ECA than on control fields (Herzog, 2005). Concerning the ecological quality of the different ECA measures, an average of 29 % (regional variation: 14-63 %) of extensive meadows and 11 % (3-38 %) of the less intensive meadows met the quality requirements. Red list plant species were found on 7 % (5-11 %) of the extensive meadows and on 3 % (0-14 %) of the less intensive meadows. Potentially endangered species were found on 18 % (9-42 %) of the extensive meadows and 17 % (19-26 %) of the less intensive meadows. Due to the high implementation rates both measures are very important for preserving and promoting biodiversity; however, the deficient ecological quality of many plots should be addressed (Knop *et al.*, 2006). With only few exceptions, the other ECA measures performed very well regarding ecological quality, hedgerows, flower strips and 'extensive meadows on wet sites' in particular. The potential of fallows and flower strips for raising the habitat quality on arable land and particularly in the lowlands was stressed. Other PEP requirements, namely an even farm nutrient balance, crop rotation, soil conservation restrictions, and selective pesticide use, have also had positive effects on biodiversity (Herzog and Walter, 2005; Knop *et al.*, 2006). At the same time, PEP requirements were criticised for their only moderate effect on habitat quality (Schläpfer, 2009). The fact that alpine summer grazing contributions provide no incentives to increase habitat quality was perceived as a weakness of the direct payment system (Swiss Federal Council, 2009).

Extenso payments and organic farming contributions were not evaluated in depth with respect to biodiversity. Gaillard and Nemecek (2002) point out the potential positive effects, especially on arthropods. With regard to organic farming, there is much more scientific evidence that payments could have a beneficial effect on many species groups (see Section 3.2.3). In Switzerland, the beneficial effect of organic farming can be expected mainly on arable land (Pfiffner and Luka, 2003). Swiss evaluations highlight research needs in this area; in particular, the synergy between different policy measures (ECA, extenso and organic farming) has not been evaluated sufficiently (Herzog and Walter, 2005).

4.4.3 Eutrophication with nitrogen

Ammonia emissions are the most severe gaseous nitrogen-related emissions from Swiss agriculture (BLW, 2008). Agriculture accounts for 93 % of the total national ammonia emissions. These emissions stem mainly from animal husbandry, arising during manure storage in the livestock housing system and when spreading manure on the field. The national nitrogen surplus in the agricultural sector decreased from 1990 to 2004 by 15 %. The ammonia emissions declined from 51,700 to 41,300 t N per year between 1990 and 2005. Emissions of nitrous oxide (N_2O) as the second most significant gas for nitrogen eutrophication were also reduced from 9,240 to 8,290 t N_2O (Herzog and Richner, 2005).

Regarding nitrate emissions to the ground water, the evaluations show that the reduction target of 5 mg/l NO_3 was narrowly missed. The target regarding the number of wells with nitrate levels below 40 mg/l was achieved, albeit only due to closing down the most critical wells (Table 11) (Herzog and Richner, 2005).

Nitrogen surpluses were reduced by 5-20 % primarily due to the PEP requirement of balanced nutrient budgets, while the 10 % tolerance could be responsible for the significant nitrogen surplus still remaining. A further reduction in nitrate leaching of 10 % resulted from the required soil conservation in the PEP. However, stocking density, as a pressure indicator for nitrogen eutrophication, was not substantially influenced on arable and mixed farms by PEP requirements. Only farms with very high stocking rates have been structurally affected (Zgraggen, 2005).

As a rule, however, policy measures with nitrogen fertiliser restrictions, including many ECA measures, can be understood to be alleviating the nitrogen problem, while incentives for higher stocking rates generally intensify nitrogen eutrophication problems, both in terms of ammonia and nitrates. Furthermore, extenso payments provide incentives to cultivate arable crops instead of grassland crops. This results in greater losses of nitrogen (Zgraggen, 2005).

4.4.4 Eutrophication with phosphorus

Since the reference years 1990-1992, phosphorus loads in surface waters have been reduced by 35 %. This reduction was caused in part by the better compensation of phosphorus loads in sediments. Therefore, the net effect of PEP restrictions in the reduction in phosphorus loads accounted for 10 to 30%. The national annual excess phosphorus level decreased by 65 % and was estimated to be around 6,000 t in 2005. The evaluations show that it is difficult to establish proof for causal relations in the field of phosphorus (Table 11). However, the increase in conservation tillage measures and the increased cultivation of catch crops are major contributors to this reduction (Herzog and Richner, 2005). Furthermore, because of the surplus phosphorus in the system, every policy measure that induces a reduction in P fertilisation, such as the ECA measures, can be expected to have a slight to moderately positive effect on P eutrophication. By contrast, instruments providing incentives for a high stocking rate can be conceived as aggravating the phosphorus eutrophication problem. Finally, Zgraggen (2005) found that extenso payments may have an overall upward impact on eutrophication, as they provide incentives for cultivating arable crops, which are more susceptible to erosion than grassland and therefore may induce an increase in P eutrophication.

4.4.5 Farm economics and efficiency

The direct payment system was evaluated economically using modelling approaches (Mann and Mack, 2004), an empirical analysis of the administrative costs (Buchli and Flury, 2005), and on the basis of theoretical economic considerations addressing questions of targeting and tailoring (Mann, 2005a; OECD, 2007a).

The impact assessment of general direct payments showed the structure-conserving effect of area-linked direct payments. Furthermore, compared to a situation without agricultural support, direct payments generate production incentives for farmers. However, these are weaker than production incentives generated by market-based support policies. Despite this, livestock-based payments in particular contribute to more intensive agricultural production and to adverse effects on environmental parameters. The justification of general direct payments in the lowlands is questionable, since an area-wide covering cultivation of the land is still guaranteed, even if these payments are cut by 100 % (Mann and Mack, 2004). Zgraggen (2005) also analysed ecological direct payments using a regional economic model. The results demonstrate that the change from market support to the direct payment scheme lowered production incentives and led to a more extensive production. PEP was assessed as only marginally influencing the production structure of both crop and animal production activities.

Policy-related transaction costs (PRTC) of the direct payments were analysed in the context of OECD activities (Buchli and Flury, 2005). Both general and ecological direct payments were considered. The study took into account only administrative costs as working hours. However, both infrastructure and the costs of developing the policy rationale and evaluating the policy were disregarded. The main conclusions were that the transfer efficiency of the Swiss Direct Payment system is high, with a share of total PRTC of 1.8-2.8 %. However, per direct payment measure, transfer efficiency varied from 0.63 % for area payments to 16.57 % for payments for extensive cereal and rapeseed cultivation (Buchli and Flury, 2005).

The theoretical economic considerations are based primarily on the Tinbergen Rule, which states that in the absence of transaction costs, each policy goal should be addressed by at least one policy measure in order to design efficient policies (Tinbergen, 1956). As discussed in Section 2.2.3, addressing multiple goals using a single measure leads to inefficiency. For example, general direct payments have the socio-political goal of guaranteeing the cultivation of remote areas and supporting farmers' incomes. However, due to the cross-compliance regulations (PEP), this measure fails to be effective for those farmers who do not comply with the cross-compliance rules. Furthermore, the rationale of the organic farming area support payments is questionable against the background of the Tinbergen Rule, as the payments foster organic cultivation techniques and thus indirectly promote multiple environmental benefits (Mann, 2005a).

There are several studies criticising the direct payment system for its inefficiency and for setting wrong incentives. The lack of clear targets of general direct payments (Swiss Federal Council, 2009) has been emphasised in particular. Furthermore, the 80:20 ratio of the budgets for general and ecological direct payments is perceived as unbalanced (Schläpfer, 2009). Although area payments are fully decoupled, they lack a clear target and hinder structural change (economiesuisse, 2006). The animal-husbandry payments have market-distorting effects and are therefore only partially green box-compatible (Swiss Federal Council, 2009). Moreover, they interfere with ecological measures, particularly in extensively cultivated regions, due to their effect of increasing stocking density (Rentsch, 2006).

4.5 Summary and conclusions

Swiss agri-environmental policy is based on the goals formulated in the federal constitution, supplemented with aspects of resource conservation in 1999. Explicit policy goals for biodiversity, nitrogen and phosphorus eutrophication were formulated. Quantitative targets were set for biodiversity and eutrophication in particular, whereas none were specified for energy use. However, these are included indirectly in the goals regarding climate change.

According to the official evaluations of direct payments, the quantitative targets regarding biodiversity have only partly been met so far. In particular, habitat quality was found to be insufficient for several ECA measures. Not all targets regarding eutrophication with nitrogen and phosphorus have been attained. There have been drastic reductions in nitrogen and phosphorus budgets, whereas nitrogen (particularly due to nitrates) and phosphorus surpluses are still too high. Several policy instruments exist which address the environmental targets regarding biodiversity and eutrophication. However, no particular instruments have been formulated for reductions in energy use. Nonetheless, most of the policy instruments described have indirect effects on all three impact categories.

A qualitative matrix of measures and policy targets is presented in Table 15. Despite the limited precision of such a qualitative approach, the effectiveness of the existing direct payments with regard to the selected environmental policy goals is shown, providing an overview of existing knowledge on the effects of the policy measures. Positive impacts of measures on policy goals are marked with a ‘+’, negative impacts with a ‘-’. The number of symbols indicates the strength of the impact. Causal relations that have not been discussed explicitly in scholarly publications but are nonetheless obvious and those mentioned in grey literature are set in brackets.

Table 15 shows that the PEP rules have a positive effect on all ecological criteria. Energy use is reduced due to the restrictions with respect to input of mineral fertiliser and stocking rates. Biodiversity is affected positively as a result of the requirement of a minimum share of ECA in UAA and due to the limited stocking rates. Eutrophication is influenced primarily by the requirement for even nutrient balances. At the same time, some aspects of general direct payments, which demand compliance with the PEP, benefit production styles that have adverse effects on attaining goals regarding the impact categories of energy use, eutrophication, and biodiversity. In particular, the payments linked to animal numbers provide incentives for intensive production with negative effects on the environmental impact categories.

Table 15 Qualitative[43] matrix of the effects of policy measures on the selected environmental policy goals

General direct payments	Reduction in energy use	Improvement in bio-diversity	Reduction in eutro-phication with N and P
Proof of ecological performance	(+)	++	++
Area payments	/	/	/
Payments for roughage consuming livestock	(--)	(-)	(---)
Payments for animal husbandry in adverse conditions	(--)	(-)	(---)
Steep slope payments	/	/	/
Payments for vineyards on slopes and terraces	/	/	/
Ecological and ethological direct payments			
Payments for ecological compensation[44]			
• Payments for mixed fallows	(++)	+++	++
• Payments for rotational fallows	(++)	+++	++
• Payments for extensive meadows	(+)	+++	+
• Payments for less intensive meadows	(+)	++	+
Payments for extensive production of grains and rape	(+)	(++)	(+)
Organic farming area support payments	(++)	(++)	(++)
Payments for particularly animal friendly stabling and livestock with outdoor exercise	(--)	(-)	(---)

Source: own table, qualitative evaluations, based on literature reviewed in Sections 3 and 4

[43] Qualitative evaluation, based on the literature reviewed in previous chapters, '+++' strong positive effect, '++' moderately positive effect, '+' slightly positive effect; '---' strong negative effect, '--' moderately negative effect, '-' slightly negative effect; '/' no causal relation. Brackets indicate that the evaluation was not based on empirical results but derived from theoretical considerations.

[44] Only those ECA measures which are included explicitly in the model framework are displayed.

ECA payments received good marks in their evaluations regarding biodiversity and eutrophication. ECA payments should have a similarly positive effect regarding the reduction in energy use. Both mixed and rotational fallows were found to positively influence all three policy goals. Energy use per ha is significantly reduced because only very view cultivation processes are carried out. For the same reasons, biodiversity was found to be very positively affected. Particularly because of the lack of nutrient influx into the system, eutrophication with nitrogen and phosphorus is affected to a significantly positive degree. However, when rotational fallows are ploughed up again to create arable land, the risk of nitrate emissions to ground water is considerable. Furthermore, adoption rates for both measures are relatively low so that their effectiveness is apparent on only a limited area.

Payments for extensive grassland have been evaluated as having a strong positive impact on biodiversity, as cutting dates and the ban on fertilisers influence this environmental category in highly positive ways. For the same reasons, there was a reduction in eutrophication and energy use in the areas where the measure is implemented. However, if livestock density on a farm is not reduced in conjunction with implementing more extensive meadows, increasing loads of nutrients are potentially going to be spread on some of the farm's fields. Less intensive meadows have only a medium-range influence on biodiversity because of the less restrictive fertilisation standards. Eutrophication decreases only slightly because there is no complete ban on mineral fertilisers. Energy use is also reduced slightly by the measure 'less intensive meadows'. Extenso payments were evaluated as having slightly positive effects on biodiversity, energy use, and eutrophication. However, as there were no official evaluations and as results are partly contradictory, e.g. with extenso payments leading to incentives for producing arable crops, the net effects of extenso payments are difficult to classify.

The official evaluations lack an in-depth analysis of the organic farming area support payments based on empirical data. Instead, investigation of the cost-effectiveness of these payments was based on the existing body of scientific literature, reviewed in Section 3.3. OFASP are very likely to reduce fossil energy use and eutrophication with N and P and to increase biodiversity. While the effect on biodiversity can be categorised as moderate, there are only slight reductions in energy use and eutrophication.

The ethological payments (RAUS, BTS) provide incentives for higher stocking densities and for energy- and eutrophication-intensive production. Hence, these payments have an adverse effect on the three selected environmental impact categories (Swiss Federal Council, 2009).

5 Working hypotheses

In this chapter, a number of working hypotheses are formulated with regard to the main research aim of this thesis: to compare the cost-effectiveness of organic farming with the cost-effectiveness of individual agri-environmental policies by developing and applying an economic modelling framework at sector level for the Swiss case.

Against the background of the reviewed literature and the conceptual thoughts of the previous sections, namely the conclusions that a) the research question does not require a monetisation of environmental externalities (Section 2.2.2), b) the research problem is too complex to be analysed by qualitative economic methods (Section 2.2.3) and c) the effects of organic farming can vary among farm types and regions (Section 3.2), it was decided to analyse the following hypotheses using a quantitative, economic programming model for the supply side of the agricultural sector, differentiated by region and farm type and based on an analytical framework oriented towards cost-effectiveness.

The working hypotheses are used as a means to structure the analysis (Section 7) and discussion (Section 8) of the research question, and to clarify the resultant contribution to knowledge (Section 9). Since this approach uses not an econometric model but a programming model, the larger aim is not to test the hypotheses by statistical means but to illuminate them by ‘synthetic experiments’ (Berger, 2000) at sector level (see Chapter 6).

The working hypotheses are formulated with respect either to Objective 3: ‘To assess both the environmental impacts and the entailed additional societal cost of organic farming and agri-environmental measures’ or to Objective 4: ‘To compare the cost-effectiveness of organic farming with the cost-effectiveness of agri-environmental measures’.

Working hypotheses related to Objective 3

In Switzerland, conventional agriculture adheres to rather strict environmental cross-compliance standards (proof of ecological performance, Section 4.3), which implies that differences in environmental effects between organic and conventional farming systems could be comparatively minor. However, the national and international literature available on environmental effects, discussed in Section 3.2, clearly suggests significant positive environmental effects of organic management with respect to the reduction in energy use, the promotion of biodiversity and the reduction in eutrophication. Therefore, in consideration of environmental impacts across the entire sector, *i.e.* taking into account structural differences between organic and conventional farms, it is hypothesised that:

H1 Generally, organic farms perform better with respect to the environmental impact categories energy use, habitat quality and eutrophication than conventional farms.

According to the literature, differences in environmental effects between organic and conventional farming have been found primarily in relation to arable land (Section 3). However, the literature on grassland suggests only slight differences between the farming systems. Hence it is expected that the relative difference in environmental performance is smallest for farm types with a high proportion of grassland. Thus it is hypothesised that:

H2 The relative differences in environmental impacts between conventional and organic suckler cow and dairy farms are smaller than on mixed farms, due to the higher proportion of grassland on these farm types and the smaller difference in environmental impacts on grassland between conventional and organic systems.

In mountain areas, conventional farms are already managed extensively. Furthermore, as grassland shares are higher than in the lowlands and hill areas (Roesch and Hausheer-Schnider, 2009), it is anticipated that

H3 Relative differences in environmental performance between the farming systems are smaller in the mountain regions than in other regions.

Working hypotheses related to Objective 4

Section 3.3 discussed the costs of organic agriculture and Section 3.4 contrasted the environmental effects with the costs. When taken together with the theoretical considerations on the efficiency of policy mixes and multi-objective policy instruments outlined in Section 2.2.3, this leads, from an economic standpoint, to the hypothesis that:

H4 Organic farming provides individual environmental services (reduction in energy use, improvement in habitat quality, reduction in eutrophication potential) at a higher cost than specialised agri-environmental measures.

However, organic farming is expected to have a positive impact on all three environmental categories (Section 3.2) in question and is expected to lead to lower public transaction costs (Section 3.3). Thus a comprehensive analysis over all three policy goals, taking transaction costs into account, may reveal a higher cost-effectiveness ratio of organic farming compared to alternative agri-environmental measures. It is therefore hypothesised that:

H5 Considering multiple environmental effects and public transaction costs of policies, the abatement costs of organic farming are comparable with or lower than other existing agri-environmental measures.

The qualitative analysis of agri-environmental instruments (Section 2.2.3), along with the comparison of Swiss agri-environmental measures (Section 2.2.3 and Sections 4.3 to 4.5), showed that there is a greater uptake of agri-environmental measures by organic farms than by conventional farms. Therefore it is hypothesised:

H6 There are synergy effects between the system approach of organic farming and individual agri-environmental policy measures which result in a higher cost-effectiveness of the agri-environmental measures when applied on organic farms than when applied on conventional farms.

6 Research approach

This chapter seeks to explain the research approach which is used to answer the research questions stated in Chapter 1 and to analyse the working hypotheses presented in Chapter 5. First, the main determinants of cost-effectiveness of agri-environmental policies at sector level are derived by developing a conceptual model, both graphically and in algebraic terms (Section 6.1). Policy uptake[45], environmental effects and public expenditure related to the agri-environmental measures are understood as the three main determinants of cost-effectiveness. Since it is argued that the parameters for the algebraic equations need to be determined at sector level[46], this section reviews existing European positive mathematical programming models with a sector-level scope regarding their ability to model the main determinants of cost-effectiveness (Section 6.2). On this basis, Section 6.3 explains the chosen modelling approach, describing a), the way the general FARMIS approach is applied in this thesis (Sections 6.3.1 to 6.3.5) and b) the implementation of additional modules for cost-effectiveness evaluation (Sections 6.3.6 to 6.3.9).

6.1 Conceptual model of cost-effectiveness of agri-environmental policies at sector level

Based on the literature reviewed and the theoretical considerations presented in Chapters 2 to 4, this section describes environmental effects, policy uptake and public expenditure as the major determinants of cost-effectiveness at agricultural sector level. In both the *ex-ante* and the *ex-post* case, substantial data-related constraints make it extremely difficult to determine

[45] The term ‘policy uptake’ is used to refer to farmers implementing an agri-environmental policy measure on their farm. In the scholarly literature, different expressions are used for policy ‘uptake’. Some authors use ‘adoption’ or ‘implementation’ to express the same notion. In the context of organic farming ‘uptake’ is usually referred to as ‘conversion’.

[46] For the benefit of brevity, the term ‘sector level’ is used to refer to the **Swiss agricultural sector**, if it is not specified otherwise. In the methodological context, ‘sector level’ is used to indicate that a given statement refers to a farm sample intended to represent all farms or a subset of all farms in the Swiss agricultural sector, rather than referring to the individual field or farm level. A ‘sector-level average’ refers to an average derived from all farms or a subset of all farms within the Swiss agricultural sector.

cost-effectiveness. It is therefore argued that programming models are useful, yet imperfect, tools for evaluating agri-environmental policies and overcoming the gaps common in observed data, particularly in *ex-ante* evaluations.

As shown in Chapter 2, cost-effectiveness is commonly understood as the ratio of costs and effects (Vedung, 2000). In the context of programme and project evaluation, cost-effectiveness analysis has been formalised as an alternative approach to the welfare-oriented cost-benefit analysis. In contrast to cost-benefit analysis, the effects in a cost-effectiveness analysis do not have to be expressed in monetary terms (Drummond, 2005).

From a policy maker's perspective, cost-effectiveness is an essential parameter for decision making, since resources are scarce and public money needs to be allocated as efficiently as possible (Mankiw, 1998). From this perspective, the cost-effectiveness of a policy measure relates public expenditure to the impacts achieved by the policy. In the context of agri-environmental direct payments, the degree to which a policy achieves goals determines its effectiveness (Marggraf, 2003). Cost is commonly conceived in terms of payments to the beneficiaries (farmers), opportunity and technical costs as well as the associated transaction costs at farm level and for public administration (Mann, 2003b).

Contrary to evaluations at plot or individual farm level, an evaluation at sector level needs to take into account policy uptake by farmers, as the extent of adoption of a policy determines how significant the effects generated by it at plot or farm level will be for the sector as a whole (Mann, 2005b; Osterburg, 2004). For instance, a policy which leads to significant improvements in biodiversity may not have much significance at sector level if only a few farmers decide to adopt the policy on their land.

Therefore, the cost-effectiveness of an agri-environmental policy at sector level can be understood as a function of a) its uptake, b) its cumulative environmental effect (*i.e.* the policy outcome) and c) the cumulative policy-relevant costs, which are in turn a function of the payment rate (Figure 11). Payment levels influence both the uptake of a policy measure and the amount of public expenditure. Environmental effects occur and further costs arise as a direct consequence of policy measures being taken up by farmers. Setting these effects and costs in relation to one another results in the cost-effectiveness ratio of a policy measure.

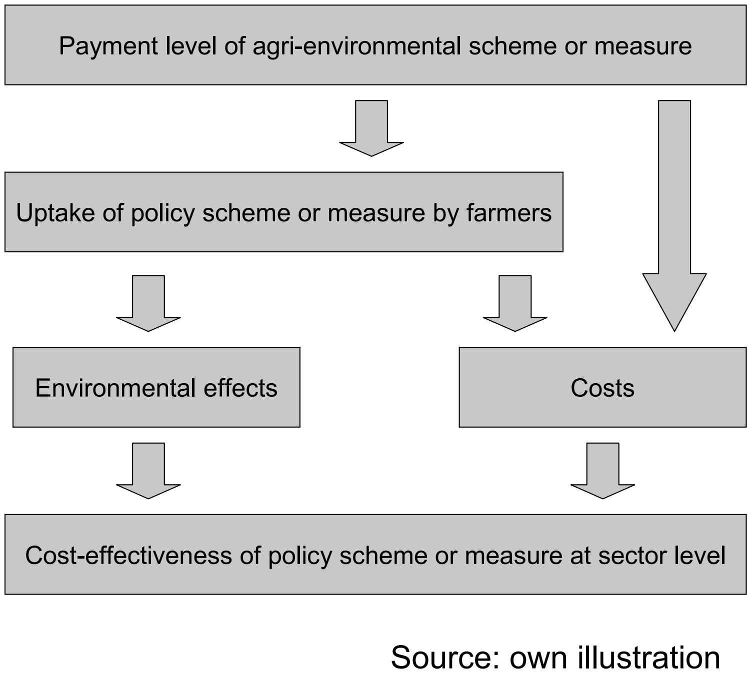


Figure 11 Causal relationship between payment levels and cost-effectiveness

6.1.1 Policy uptake

The uptake of agri-environmental measures and its determining factors have been studied many times in both the EU and Switzerland (e.g. Dupraz *et al.*, 2004; Mann, 2005b). Both economic and non-economic factors influencing policy uptake have been identified.

Non-economic factors include both socio-demographic and intrinsic factors. For instance, both a high age and low level of education make farmers less likely to take up agri-environmental policies (Vanslembrouck *et al.*, 2002). Burton explains the low uptake levels of agri-environmental programmes as being related to the minor gains in social capital experienced by farmers (Burton *et al.*, 2008). Farmers often take up agri-environmental policies in order to bring about a perceivable improvement in the environment, stating convincingly that their uptake decision does not depend on economic considerations at all (Jurt, 2003).

The uptake of measures with a fundamental impact on farm organisation, namely organic farming area support payments, is influenced particularly by complex factors. Conversion to organic farming is driven by a variety of economic and non-economic factors, including contact to neighbouring farms and the farmer's environmental motivation (Bichler *et al.*, 2005). Padel (2001) also examines the relevance of adoption theory as a means of understanding the rate at which organic farming may be adopted, focussing on farmers' goals (financial and non-financial) and on the type of farmers (pioneers, mainstream early and late adopters) who are willing to adopt at any particular stage in organic sector development. Padel (2001) identifies the complexity of the innovation as a key factor affecting the ease and rate of adoption.

At the same time, economic theory asserts that farmers will take up agri-environmental measures as long as it is profitable to do so, *i.e.* as long as the marginal benefit of one hectare of additional agri-environmental measure exceeds its marginal costs (Salhofer and Glebe, 2006). This assumption of rational behaviour on the part of farmers is supported by empirical evidence, as farmers' uptake levels tend to be higher if the opportunity and technical costs of adoption are low. For example, uptake levels of agri-environmental programmes are higher in mountain areas where only an extensive form of production is possible. Furthermore, the lower the technical costs for farmers, the higher the likelihood that they will participate in an agri-environmental programme (Mann, 2005b).

6.1.2 Environmental effects

The most frequently studied topic related to agri-environmental policies is their effectiveness in achieving policy objectives, *i.e.* minimisation of negative environmental impacts or provision of positive externalities of agriculture (e.g. Purvis *et al.*, 2009; Stolze *et al.*, 2000).

In Switzerland, agri-environmental policy has also been studied extensively. In the course of the official evaluations (outlined in Section 4.4), about 40 research projects were set up, most of which have been targeted at environmental effectiveness (Badertscher, 2005; Brower, 2004). With a few exceptions (Zgraggen, 2005), however, environmental effects have so far been studied at field (Gaillard and Nemecek, 2002), rotation (Alföldi *et al.*, 1999) or farm level (Alig and Baumgartner, 2009). Both in Switzerland and internationally, only few studies analyse environmental effects at national level (Pufahl, 2007; Schmidt and Osterburg, 2005).

Nevertheless, 'Swiss Agricultural Life Cycle Assessments' (SALCA) have provided representative values for environmental effects, distinguishing between farming systems (integrated and organic farming) and regions (valley, hill and mountain region). Furthermore, the direct field-level environmental impacts of the most important agri-environmental measures are incorporated and most of the relevant impact categories have been analysed.

A crucial question for upscaling environmental effects from field or farm level to sector level is whether a linear relation can be assumed between uptake levels and effects. The potential reasons for non-linearity, *i.e.* decreasing, increasing or variable marginal effects at sector level, may be various:

- **Deadweight effects and self-selection bias:** Deadweight effects occur for the first hectares under a policy, because there is empirical evidence that those farms take up a policy where no or almost no change in management is necessary (Henning and Michalek, 2008)

- **Regional differences and differences between farm types:** a measure may have a larger impact, if, for example, it is implemented on a specialised cash crop farm rather than on an already extensively managed mixed farm (Pufahl, 2007).

- **Gossen's First Law (law of decreasing marginal utility):** The more of a good is consumed, the lower the gains in utility are. Although this law is developed for commodities, the relationship can be observed for non-commodities as well. For example, the utility of a further decrease in nitrate content in drinking water may be high if the content exceeds a set threshold, but it may be low if the level of nitrate is already low (Schader, 2007).

- **Minimum ecological requirements:** contrary to Gossen's First Law, there may also be cases where marginal utility increases with higher uptake. Sometimes, a minimum of landscape complexity is required before any additional positive effect on species biodiversity can be achieved due to the uptake of agri-environmental measures. Although this effect is locally specific, it can be argued that it leads to a different effect curve at sector level (Roschewitz *et al.*, 2005).

Possible relations between uptake (U) and cumulative environmental effects (E) are shown in Figure 12A. The marginal environmental effect at sector level ($\frac{\partial E}{\partial U}$) may be either constant or variable (decreasing, increasing, variable). The shape of the curve is different for different environmental objectives and indicators. Due to data constraints, the exact course of the uptake-effect curve cannot be observed empirically. However, using econometric models the curves can be estimated, provided that individual farm data on the environmental impacts are available (Frondel and Schmidt, 2005). Lacking these data most studies assume linear relations for environmental effects (Julius *et al.*, 2003).

6.1.3 Public expenditure

As Mann (2003a) pointed out, the costs of policy measures can be interpreted in various ways (see Section 3.3). While some authors understand the costs of policy measures to be the cumulative payments made to farmers (Marggraf, 2003), Mann (2003a) distinguishes between costs at farm level and costs at state level. Farm-level costs comprise technical costs, opportunity costs, and farm-level transaction costs. State-level costs are composed of the payments to the beneficiaries and public transaction costs. Additional tariff revenues due to higher imports have to be deducted from these state-level costs (Mann, 2003a).

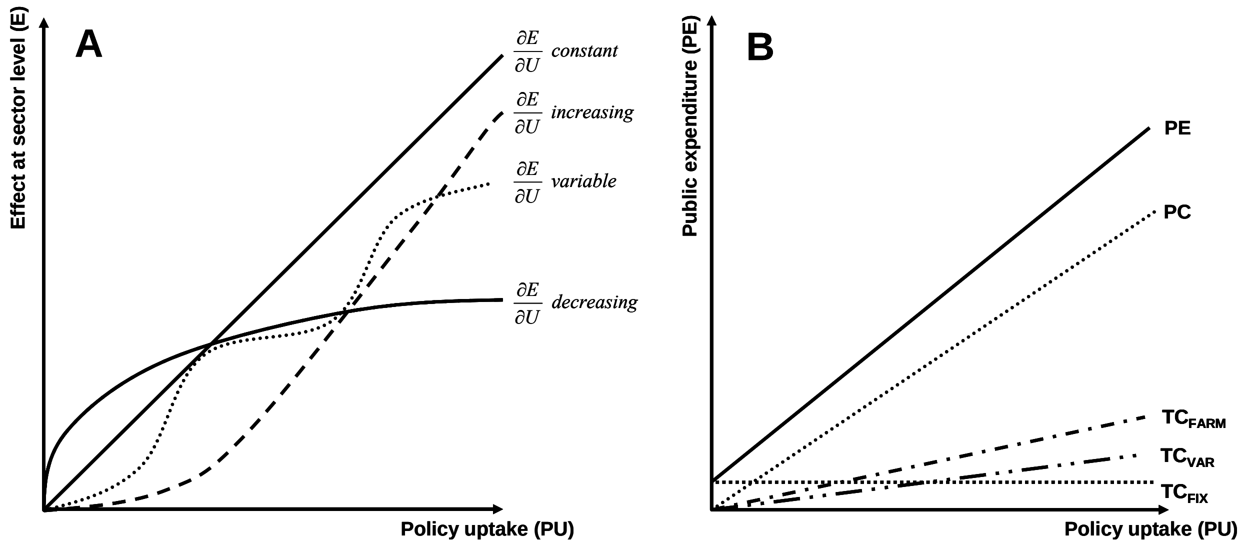
From the budgetary point of view, rather than a farm-level perspective, the costs to public authorities of implementing a policy and achieving environmental effects constitute public expenditure. The principal share of this public expenditure consists of the payments to the beneficiaries, meaning compensation for farm-level costs. Equally, however, there is a share of public transaction costs that is highly variable. Transaction costs occur at different levels and different stages: at national level, during the overall disbursement of payments and in relation to cantonal reporting and supervision. At regional and local level, major transaction costs are caused by managing the payments, gathering monitoring and control data, and verifying eligibility criteria. Farm-level transaction costs – which according to most authors, constitute the bulk of total transaction costs – involve the filling in of forms by the farmer and the additional workload due to farm inspections (Buchli and Flury, 2005; Tiemann *et al.*, 2005). Many studies show that transaction costs at different levels and for different policies can add up to a significant share of total public expenditure (McCann *et al.*, 2005; Vatn, 2002).

The amount of transaction costs depends on the institutional environment, the individual farmer's education and knowledge, farm characteristics such as type and size, and policy-related factors (Buchli and Flury, 2006; Rørstad, 2007). Policy-related factors include: asset specificity, uncertainty, and frequency of transaction (Williamson, 1989); due to these factors the share of policy-related transaction costs occurring at farm level can vary from 0.2-65 % of the payment rates (DG Agri, 2007). Empirical studies show that transaction costs for agri-environmental payments are especially high and should therefore be taken into account, because the differences in transaction costs between policies may influence policy maker's decision (Rørstad, 2007; Vatn *et al.*, 2002).

Although some authors stress the role of transaction costs as 'quality assurance costs' (Buchli and Flury, 2006), it is generally agreed that to achieve an efficient policy process, the share of transaction costs should be kept as small as possible (Jacobsen, 2002; Vatn, 2002).

As farm-level transaction costs – like opportunity and technical cost – are meant to be compensated by direct payments, they should not be added onto public expenditure. Nevertheless, farm-level transaction costs constitute a relevant parameter that needs to be analysed in an evaluation of the cost-effectiveness of policy measures (Tiemann *et al.*, 2005).

As demonstrated in Figure 12B, a linear relation between uptake level (U) and public expenditure (PE) can be assumed because, independent of the area entered into an agri-environmental programme, the same marginal costs for payment rates (PC) and the same transaction costs (TC) occur for public authorities. Apart from the linearly increasing cost components, there is also a fixed transaction cost component (TC_{FIX}), independent of the uptake level (Rørstad, 2007). These fixed transaction costs arise because as soon as a policy is implemented – no matter how high the uptake is – a certain administrative infrastructure for monitoring and control has to be maintained. It should be noted that the transaction costs at farm level (TC_{FARM}) are not a cost component of public expenditure, since these by definition are already remunerated within the payment costs (PC).



Source: own illustration

Figure 12 Possible environmental effects (A) and public expenditure components (B) of an agri-environmental policy in relation to its uptake level

6.1.4 Cost-effectiveness at sector level

Using the determinants of cost-effectiveness identified above, Figure 13 shows how the different payment levels of a hypothetical agri-environmental policy instrument influence cost-effectiveness at sector level.

The **north-eastern quadrant** of Figure 13 presents the relation between payment levels (PL) of a policy measure and policy uptake (PU)[47]. The curve is S-shaped because very low payment levels will not lead to a significant uptake by farmers if farm-level costs (opportunity, technical and transaction costs) are not covered. The more the payment level increases, the higher the uptake, with more farms adopting the policy. When a certain uptake level is reached, it is likely that the farms remaining outside the scheme have not entered due to very high opportunity costs or other factors. Much higher payment levels would be required to encourage them to participate. Therefore, the shape of the uptake curve is likely to flatten in the end. This curve corresponds with Roger's (1993) adoption theory, which assumes a bell curve for adoption rates over time.

[47] An agri-environmental measure with uniform payment rates is assumed, rather than payment rates differentiated by farm or an auction-based policy.

As illustrated above, there is a linear relation between uptake and public expenditure (PE). Thus the uptake-public expenditure curve, shown in the **south-eastern quadrant of Figure 13**, runs according to the course of the uptake-payment level curve. The fixed amount of transaction costs make the curve not start in the centre of the graph.

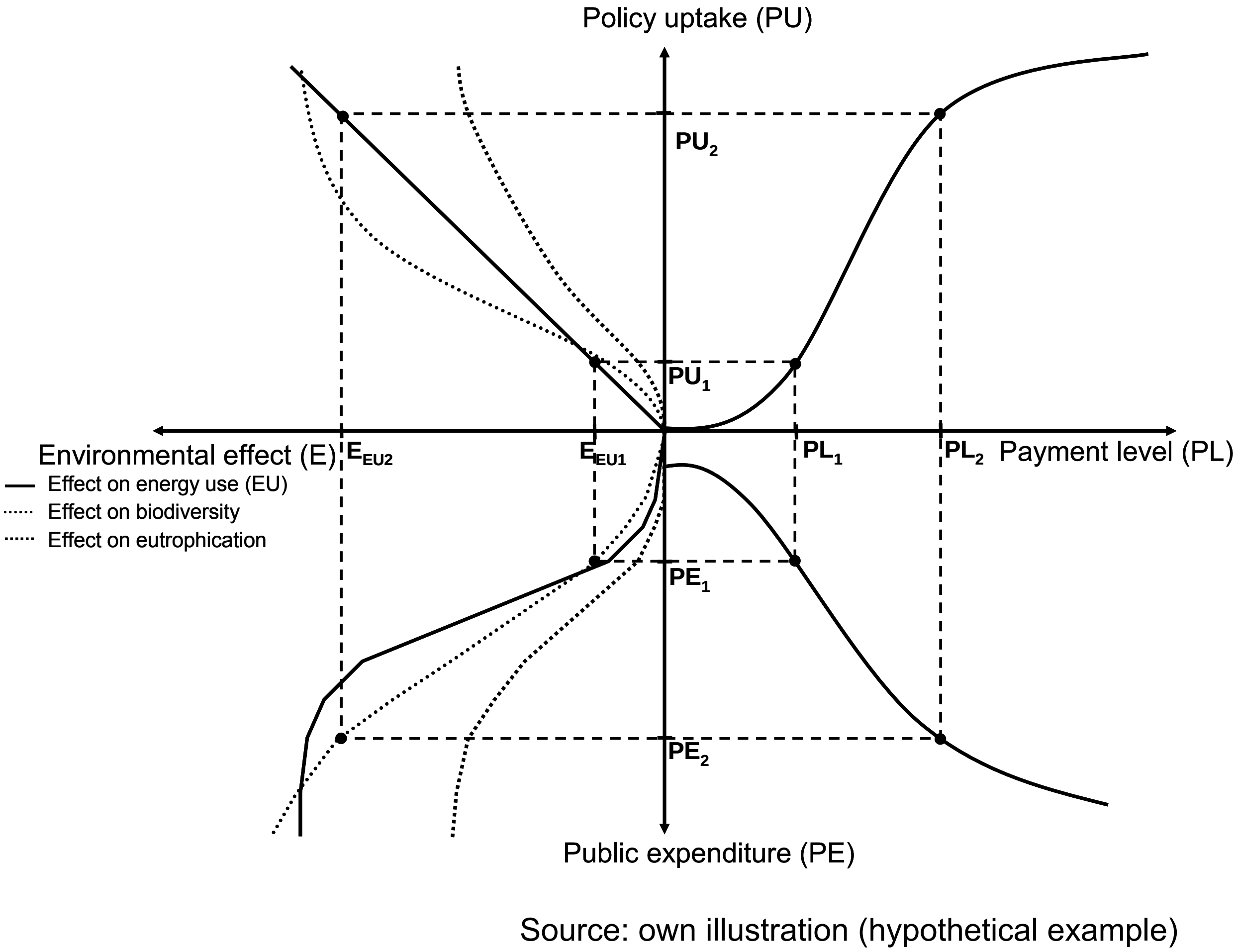


Figure 13 Graphical representation of the cost-effectiveness at sector level of single policy measures depending on the payment level

The **north-western quadrant of Figure 13** shows the relation between policy uptake (PU) and environmental effect (E) with regard to energy use (E_{EU}), biodiversity, and eutrophication, the three environmental impact indicators selected for this analysis. There may be linear relations as well as non-linear relations between the environmental effect and policy uptake, as illustrated for biodiversity and eutrophication in Figure 12A.

Finally, the cost-effectiveness function, *i.e.* the sector-level effects on habitat quality, energy use and eutrophication as a function of public expenditure, is represented in the **south-western quadrant**. The optimal payment level in terms of cost-effectiveness regarding the minimisation of energy use theoretically lies somewhere between PL_1 and PL_2 because according to Figure 13, payment levels which are lower than PL_1 give rise to only minimal effects, while the additional effects of payment levels beyond PL_2 lead to disproportionately high costs. This effect is even stronger for biodiversity and eutrophication due to the supposed non-linear uptake-effect curve.

The graphical representation of cost-effectiveness presented above leads us back to Equation 2 (page 15) for the cost-effectiveness ratio (CER) of a policy in relation to a single environmental effect at sector level (see Section 2.2.2). Equation 2 is the basis for deriving cost-effectiveness algebraically. Suppose several policies (i) and environmental effects (j) are considered, then Equation 3 can be modified to become Equation 10

$$CE_{ij} = \frac{E_{ij}}{C_i} \quad \forall\, i, j \tag{10}$$

where CE_{ij} is the cost-effectiveness of policy i in relation to environmental effect j. CE is defined as the ratio of the environmental effect (E_j) of policy i and the cost (C) of policy i. The total sector-level environmental effect can be calculated, as in Equation 11, by adding up the effects multiplied by the areas where the effects occur.

$$E_{ij} = \sum_{X} E_{ijx}\ AR_{ix}\ \theta_j \quad \forall\, i, j \tag{11}$$

E_{ij} is the total environmental effect of policy i on environmental category j, x is the index for unit of uptake which, in the present context, is always the area on which a policy-induced environmental effect occurs. AR is the size of the unit of uptake (area). Factor θ_j enables the modelling of non-linear relations between uptake level and environmental effect. For instance, if a decreasing marginal effect with higher uptake levels is assumed (as suggested in Figure 12A), θ_j would take on a value below 1. As empirically justified data for θ_j are usually lacking, a linear relation between uptake and cumulative environmental effect can be assumed, which makes θ_j become 1.

Alike the cumulative environmental effects, the total additional policy-related cost at sector level can be calculated using Equation 12

$$C_i = \sum_{X} ((PL_{ix} + TC_{FARM\,ix} + TC_{VAR_{ix}})\ AR_{ix}) + TC_{FIX} \quad \forall\, i \tag{12}$$

where PL_{ix} is the payment level for policy i and area x; TC_{FARM} are the farm-level policy-related transaction costs (PRTC), TC_{VAR} the variable share of the public PRTC, and TC_{FIX} the fixed public PRTC.

Those environmental effects which cannot be expressed in terms of total values (as, for example, in the case of biodiversity), need to be expressed as sector averages. Accordingly, the average environmental effect (AE) can be calculated as shown in Equation 13. The consideration of average effects means bypassing the problem of specifying θ_j. The average cost (AC) is determined using Equation 14. Note that model-specific equations for costs are given in Section 6.3.

$$AE_{ij} = \frac{\sum_X E_{ijx}\ AR_{ix}}{UAA} \quad \forall\, i, j \tag{13}$$

$$AC_i = \frac{\sum_X ((PL_{ix} + TC_{FARM_{ix}} + TC_{VAR_{ix}})\ AR_{ix}) + TC_{FIX}}{UAA} \quad \forall\, i \tag{14}$$

The above equations represent cost-effectiveness calculations for cases where only a single environmental effect is considered. However, if policy instruments induce multiple environmental effects, the cost-effectiveness ratio regarding a single effect may be flawed. There are basically two options for dealing with this problem:

1. Calculate as many cost-effectiveness ratios as there are environmental effects and allocate cost shares to each environmental indicator.

2. Calculate a single cost-effectiveness ratio by expressing the effects in a single unit.

The disadvantage of the first option is that no overall cost-effectiveness ratio can be calculated. Furthermore, allocating differentiated costs to environmental effects, will need to be carefully justified. At the same time, however, the advantage of this option is that no information is omitted, so that the effect on each environmental category remains transparent.

The second option brings us back to the problem of the CBA, namely the difficulty of expressing multiple environmental effects in a single unit. Monetisation can be avoided, however, by expressing the effects of the policy instruments on the environmental indicators in relative values, e.g. ‘percentage reduction’, ‘percentage improvement’ or ‘relative distance-to-target’. While this bypasses the problem of monetisation, the question of weighting, discussed in the context of the MCA (Section 2.2.2), does arise. Since there is hardly any empirical data regarding the relative importance of the different environmental categories and their related targets, there is no robust empirical guide for allocating different weights. Thus it seems to be justified to apply an equal weighting in combination with sensitivity analyses in the context of this thesis.

This leads us to Equation 15, where RE is the relative effect on the environmental category resulting from the policy (expressed as a percentage). Either total effects or average effects can be used to calculate RE. RE_{ij} and C_i are calculated as shown in Section 6.3.9. If both RE_{ij} and C_i are determined, they can be used in Equation 15 to calculate and compare the cost-effectiveness of the policy instruments.

$$CE_i = \frac{\frac{\sum_j RE_{ij}}{I}}{C_i} \quad \forall\, i \qquad \textbf{(15)}$$

I is the number of environmental categories taken into account. Policies with a high CE will be more favourable compared to policies with a low CE for the given set of environmental categories.

The reciprocal of CE is called abatement cost (ABC) in the case of negative externalities and provision cost (PRC) in the case of positive external effects (Equation 16 and 17). Both ABC and PRC serve as a comprehensible indicator for cost-effectiveness in Chapter 7.

$$ABC_i = \frac{C_i}{\frac{\sum_j RE_{ij}}{I}} \quad \forall\, i \qquad \textbf{(16)}$$

$$PRC_i = \frac{C_i}{\frac{\sum_j RE_{ij}}{I}} \quad \forall\, i \qquad \textbf{(17)}$$

Consequently, in order to calculate CE, ABC and PRC using Equations 10 to 17, it is necessary to establish sector-level uptake levels, costs and environmental effects

In this section, it has been shown how the three main determinants of cost-effectiveness can be understood as policy uptake, environmental effects and public expenditure. The procedure for calculating these determinants, employing an economic model based on the sector level, is explained in Section 6.3.

6.2 Review of existing programming models for AEP evaluation

The following sections analyse existing sector-level PMP models (Howitt, 1995) contributing to the derivation of the three determinants of cost-effectiveness (policy uptake, environmental effects and public expenditure).

As Britz and Heckelei (2008) illustrate, both partial equilibrium models and programming models have already been employed to assess the impact of agri-environmental policies. However, due to the partial equilibrium model's inherent characteristics, these models seem less suited to the assessment of agri-environmental policies (Mittenzwei *et al.*, 2007). Therefore, according to Britz and Heckelei (2008), there are only very few examples of partial equilibrium models incorporating environmental indicators.

The use of programming models is more common than partial equilibrium models for assessing the impacts of agri-environmental policies at sector level (Britz and Heckelei, 2008). Several European models exist that address uptake, environmental effects or public expenditure. Some selected approaches are discussed below.

6.2.1 Coverage of environmental effects

Modelling environmental effects at an aggregate level, either for the agricultural sector, for regions, or for different farm types, is a common function of mathematical programming models. In total, 12 European programming models were found which had integrated environmental indicators 7 of which were Positive Mathematical Programming (PMP) models and 5 of which were Linear Programming (LP) models. Furthermore, within the 6th Framework Programme of the EU, several initiatives have been started to link models of different classes together in order to be able to address environmental concerns at an aggregate level. These approaches include the projects SEAMLESS (van Ittersum *et al.*, 2008), SENSOR (Jansson *et al.*, 2007), MEA-Scope (Piorr *et al.*, 2007) and INSEA (Kraxner, 2006).

The most common procedures for integrating environmental concerns into programming models are to link either normative environmental data (Helming, 2003; Julius *et al.*, 2003; Sattler and Zander, 2004; Schmid and Sinabell, 2006a) or complete bio-physical models (Jansson *et al.*, 2007; Kraxner, 2006; Piorr *et al.*, 2007; van Ittersum *et al.*, 2008) to the activities of the economic models. In doing so, these approaches vary in the type of environmental indicator modelled, the quality of the indicator data used, the link between the data and the model, and their general model characteristics (geographical scope, ability to represent separate regions and/or farm types, ability to take into account the element of time, and site specificity), as shown for relevant European PMP approaches in Table 16. LP approaches covering environmental indicators, such as AROPAj (De Cara *et al.*, 2004), S-INTAGRAL (Peter, 2008) and MODAM (Sattler *et al.*, 2006), were excluded for this review for the benefit of brevity.

In terms of geographical scope, all the programming models reviewed, except CAPRI, apply to the national level. The calibration is done according to supply elasticity for the activities in all models, whereas CAPRI follows an econometric calibration of land use activities according to Heckelei (2002). While all models are capable of representing regions, only FARMIS and PROMAPA.G are able to specify according to different farm types. Apart from the Austrian sector model PASMA (Schmid and Sinabell, 2006b), FARMIS is the only model which can separately optimise organic and non-organic farms (Sanders *et al.*, 2008). All models are static, although both CAPRI and SILAS currently implement a dynamisation (Britz, 2005), *i.e.* a yearly calculation of the responses of the farm sector instead of just calculating the base year and the scenario runs. Site-specific characteristics are taken into account endogenously by RAUMIS, while CAPRI considers soil types within the results calculation.

Environmental indicators are covered in different analytical contexts and using different approaches in the reviewed models. For instance, nutrient balances can be modelled either by using completely normative data or according to fertiliser purchase data from FADN, as in RAUMIS (Julius *et al.*, 2003). Nutrient balances and fertiliser-related emissions such as greenhouse gases and ammonia are the most common environmental indicators (see also Britz and Heckelei (2008)). However, only RAUMS and SILAS cover the indicator of pesticide risk or eco-toxicity. The most problematic aspect of eco-toxicity as an indicator within agricultural sector models is the high variability combined with a high degree of uncertainty.

Table 16 Overview of the European PMP models reviewed and of their characteristics

Model	Main publication referred to	Geographic scope	Calibration	Regional representation	Farm type representation	Static/ dynamic	Site specificity	Coverage of environmental indicators
CAPRI	Helming (2003); Kränzlein (2008)	EU-level, NUTS 1, NUTS 2	Econometric for crop activities supply elasticity for animal activities	Yes	indirect representation	Static (dynamisation in progress)	No (but soil types considered in results calculation)	N, P, K balances, Ammonia output, Greenhouse gas emissions Water balances Energy use
DRAM	Helming (2005)	The Netherlands	Supply elasticity	Yes	No	Static	No	Ammonia emissions, Nitrogen surplus
FARMIS	Bertelsmeier (2005); Sanders (2007)	Selected EU member states and Switzerland	Supply elasticity Intensities based on Röhm-Dabbert approach (RDA)	Yes	Yes	Static	No	Currently being developed for CH-FARMIS: Energy use (CH), N and P Eutrophication (CH) Biodiversity (CH)
PASMA	Schmid and Sinabell (2006b)	Austria	RDA, linear approximation	Yes	No	Static	No	Fertiliser balances
PROMAPA.G	Júdez *et al.* (2006)	Spain	Optional econometric calibration	Yes	Yes	Static	No	Nutrient balances
RAUMIS	Julius *et al.* (2003)	Germany, differentiation up to NUTS 3 level	Supply elasticity	Yes	No	Static	Yes (differentiation according to soil type classification)	Nutrient balances, NH3 emissions, Pesticide risk, Crop diversity
SILAS	Mack *et al.*(2007)	Switzerland	Supply elasticity	Yes	No	Static (dynamisation in progress)	No	Energy use, Eutrophication, Greenhouse gas potential, Ecotoxicity

Source: own compilation

Even rarer is the incorporation of biodiversity indicators in agricultural sector models. According to Britz and Heckelei (2008) the coverage of biodiversity requires site specificity in the economic model. However, the possibilities for site-specific modelling are rather limited at sector scale. Only RAUMIS (NUTS 3 level) and CAPRI (NUTS 2 level) consider soil types as site-specific information. RAUMIS models crop diversity, as a habitat diversity indicator, whereas species diversity has not been implemented in an aggregate programming model so far. One exception is the LP model MODAM which covers biodiversity using a fuzzy-set tool (Zadeh, 1997) for different case studies in Europe (Sattler *et al.*, 2007).

Mattison and Norris (2005), however, state that biodiversity impacts in general can be included in economic models, even at a larger scale (see also the above examples of RAUMIS and MODAM). There is certainly a trade-off between ecological relevance and analytic tractability (Eppink and van den Bergh, 2007).

6.2.2 Coverage of public expenditure

Due to their nature as a policy information tool, a necessary common feature of aggregate programming models is coverage of public expenditure on agricultural policies. However, models vary in their ability to allocate public expenditure to administrative units, regions, farm types and policies. These allocations may need to be sophisticated because, unlike payments to beneficiaries, other public expenditure is not straightforwardly allocatable. In particular, transaction costs occurring in public administrations are difficult to allocate specifically (Buchli and Flury, 2005). Presumably due to the non-availability of data in EU Member States and the difficulties of allocating some transaction cost components to specific measures, there is no aggregate programming model available that explicitly takes into account transaction costs for agri-environmental policies.

6.2.3 Coverage of the uptake decision

Modelling the decision of farms to take up agri-environmental programmes might be the biggest challenge (Britz and Heckelei, 2008). Agri-environmental policies are represented in programming models by defining a separate activity for each policy measure. For example, the grassland extensification measure can be represented by defining the activity 'extensive grassland'. In LP approaches, if run without calibration restrictions, the problem of overspecialisation occurs, *i.e.* farms opt either for the extensive or the standard grassland activity, depending on the gross margins of the activities. If run with bounds, e.g. if information on the uptake of an agri-environmental measure is available, the model behaviour is limited (Umstätter, 1999). In positive mathematical programming (PMP) models the problem of overspecialisation is solved by calibration. However, the reactions of the model are not estimated econometrically but are determined on the basis of some few observations and therefore lack satisfactory empirical justification. Furthermore, in PMP models only diagonal elements of the Q-Matrix are defined, while non-diagonal elements are zero (Heckelei, 2002).

The Röhm-Dabbert approach (Röhm and Dabbert, 2003) addresses the uptake decision of PMP models specifically. By defining the agri-environmental policies as sub-activities of their standard activities, different supply elasticities can be attached to each of them. In other words, the slope of the marginal cost function is split into two parts: one that depends on the level of the sub-activity (e.g. extensive or standard grassland) and one that depends on the level of the total activity (e.g. sum of all grassland sub-activities). As a result, the sub-activities can be exchanged more easily than activities that require fundamental changes in farm structure.

However, like the standard PMP approach, the weakness of the Röhm-Dabbert approach remains its arbitrariness in econometric terms (lack of sample size, non-diagonal Q-Matrix elements 0) (Britz and Heckelei, 2008). Thus, the level of exchangeability is defined externally and not necessarily on the basis of econometric estimates (Heckelei, 2002).

Nevertheless, the Röhm-Dabbert approach performs more satisfyingly than standard LP or PMP approaches: an alternative approach is currently not available (Gocht, 2005; Kanellopoulos *et al.*, 2007) and time-series or cross-sectional data on policy specific uptake are often not available. According to the scanned literature, only PASMA (Schmid *et al.*, 2007) and FARMIS (Kuepker, 2004) adapted the Röhm-Dabbert approach for concrete policy analysis. However, several CAPRI calibration procedures have been tested within the EU Integrated Project SEAMLESS, the Röhm-Dabbert approach among them (Kanellopoulos *et al.*, 2007).

Conversion to and from organic farming is not modelled by any of the models examined. A key requirement for modelling conversion to and from organic agriculture, or whole farm agri-environmental policies in general, is a farm-level representation, as the conversion decision is made for the whole farm. Only FARMIS (Sanders, 2007) and PASMA (Schmid *et* *al.*, 2007) have implemented the option of distinguishing between organic and non-organic farms.

The fact that conversion has not been modelled explicitly within a PMP model is mainly due to the complexity of the farmer's decision, which depends on multiple factors (Hollenberg, 2001). While economic factors such as conversion costs and expected changes in farm income influence the conversion behaviour of farmers, non-economic factors (which cannot be included in the objective function of a programming model) play also an important role (Jurt, 2003; Padel, 2001). Some authors suggest that the conversion decision can be addressed by using dynamic models, e.g. based on 'New Investment Theory' (Musshoff and Hirschauer, 2004; Odening *et al.*, 2004) or using the qualitative concept of path dependency (Latacz-Lohmann *et al.*, 2001). Due to these multiple decision-making factors, an econometric estimation of conversion promises to deliver more realistic estimations of conversion rates than programming models.

In essence, several PMP models which are capable of modelling specific determinants of cost-effectiveness were identified. SILAS and RAUMIS, for instance, are capable of modelling a number of different environmental indicators (see Table 1). Furthermore, several approaches can be used to model the uptake of agri-environmental schemes on the basis of the Röhm-Dabbert approach. Public expenditure is generally modelled by each approach. However, none of the models reviewed takes into account policy-related transaction costs explicitly. Furthermore, currently only FARMIS and PASMA model organic farms separately, which is a further precondition for if given research question of this thesis shall be addressed.

Therefore, in view of the challenges faced in modelling the determinants of cost-effectiveness and the specific problem of evaluating organic farming, programming models can generally be regarded as offering a suitable basis for evaluating agri-environmental policies at sector level.

6.3 Modelling approach

In this section, first, the general concept of the modelling approach is described (Section 6.3.1). Second, the data sources of the model are presented (Section 6.3.2). Third, the generation of input-output coefficients is shown (Section 6.3.3) followed by, fourth, the model specification is depicted (Section 6.3.4). Fifth, the model calibration is explained (Section 6.3.5). Sections 6.3.6-6.3.8 demonstrate how the three main determinants of cost-effectiveness of agri-environmental schemes are calculated. Finally, the way organic farming, as a farming system, is compared with targeted agri-environmental schemes is illustrated in Section 6.3.9.

6.3.1 General overview of the modelling approach

As shown in the previous sections, in order to answer the research questions, the modelling approach needs to

1. directly model farm management and relations between farm-internal activities in order to provide an understanding of the responses of the farming sector to changes in agri-environmental policy;
2. display a realistic model behaviour without the problems of overspecialisation (Umstätter, 1999) or extreme solutions;
3. distinguish between organic and non-organic farms with regard to farm-level decision making;
4. include adequate representation of the entire Swiss agricultural sector;
5. model the uptake of agri-environmental policy measures by farms;
6. model environmental effects representative for the Swiss agricultural sector;
7. model public expenditure including policy-related transaction costs.

The approach developed for addressing the research questions is based on the comparative-static farm group model FARMIS, which has been used for policy analysis in Germany since 1998 and which has been adapted for several EU Member States (Offermann *et al.*, 2005). Since 2007, FARMIS has been adapted to the Swiss policy context and extended by representing the agricultural sector based on differentiation according to farming system (Sanders, 2007). Accordingly, the Swiss FARMIS model (henceforth called CH-FARMIS) is able to assess the economic impact of agricultural policies on different farm groups that can be defined in a flexible way.

CH-FARMIS is a comparative-static mathematical programming model for the Swiss agricultural sector based on linear programming (LP). As an optimisation model, FARMIS attempts to model directly the decision-making process of farmers using an objective function which explicitly takes expectations and the technical production environment into account (Bertelsmeier, 2005). Farmers' responses to changes in exogenous conditions (e.g. direct payments or product prices) are thus modelled by conducting 'synthetic experiments' (Berger, 2000). In contrast to econometric approaches, FARMIS models the decision-making process directly. Econometric models use time-series, cross-sectional or panel data as an empirical basis for model calibration and then extrapolate the development into the future, taking into consideration the impact of endogenous variables (Berger, 2000).

The standard FARMIS procedure consist of four steps: First, the farm groups are assembled on the basis of FADN data (Section 6.3.2). Second, input-output data are generated specifically for the assigned farm groups (Section 6.3.3). Third, the detailed model assumptions are specified according to the requirements of the research question (Section 6.3.4). Fourth, the model is calibrated for the base year by running it as a linear programme with calibration constraints in order to reveal the hidden, *i.e.* not explicitly modelled, costs. Fifth, policy scenarios are calculated using the calibrated, quadratic PMP model and scenario-specific assumptions (Section 6.3.5 and 6.3.6).

Thus, CH-FARMIS is already largely capable of fulfilling the above requirements 1 to 4 above. In order to meet requirements 5 to 7 – *i.e.* to take account of uptake, environmental effects and public expenditure of agri-environmental schemes – CH-FARMIS was extended within the scope of this thesis.

6.3.2 Data sources

CH-FARMIS is based primarily on book-keeping records by the *Zentrale Auswertung (ZA)*, which is the Swiss equivalent to the European Farm Accountancy Data Network (FADN)[48].

The Swiss FADN consists of 3,000-3,500 sample farms, claiming to be representative of those 50,937 farms (2007) that fulfil the FADN eligibility criteria. The remaining farms can

[48] Henceforth the term 'Swiss FADN' or FADN refers to the data of the *Zentrale Auswertung (ZA).*

be characterised as small holdings and hobby farms, which do not have a significant influence on the sector as a whole and are therefore excluded from the population which the FADN sample claims to be representative of.

Generally speaking, a differentiated analysis of specific groups of farms is possible, since FADN data contains farm-specific information on structural criteria such as region, farm type, farming system, size class and farmer's age, etc. For instance, organic dairy farms in the mountain regions with a UAA higher than 20 hectares can be analysed separately with FARMIS, provided there are enough farms in the FADN sample. A stratification of the sector according to structural criteria generates major benefits in terms of the scope and quality of the modelling analysis. First, stratification results in relatively homogeneous farm groups, which show more realistic model responses compared to regional models, such as RAUMIS or SILAS (see Table 1). Second, stratification enables comparison of the different responses of the farm groups.

However, due to the limited sample size, not all the different strata can be represented (Table 17). The more stratification criteria are applied, the more strata will be empty, resulting in a lower level of representativeness of the model overall. In particular the possibility of stratification is limited due to the fact that it is necessary for the analysis to differentiate between organic and non-organic farms. Nevertheless, a separate modelling of organic farms is feasible thanks to the comparatively large share of organic agriculture (Sanders, 2007). FADN book-keeping records were adapted for the model according to the following procedure:

1. **Selection of identical farms:** In order to minimise distortions created by flaws in bookkeeping data or from yearly fluctuations, only those farms are taken into account which are present in the datasets of both years. Data from the years 2006 and 2007 was included in the analysis. The datasets contained 3,270 farms in 2006 (of which 417 were organic farms) and 3,326 farms in 2007 (of which 444 were organic farms). In principle, sample data from three or more years could be used, although this would entail a decline in the number of farms in the sample and the representativeness of the sample.

2. **Stratification:** Keeping in mind the above mentioned stratification trade-off between homogeneity and representativeness of the model analysis, stratification according to farm type, farming system and region was opted for. Using these three stratification

criteria many individual strata were not represented. Therefore, size class was applied as a fourth criterion only for those farm groups consisting of more than 50 farms, in order to achieve farm groups that are as homogeneous as possible. Farm groups containing fewer than two FADN farms were excluded from the analysis.

3. **Generation of consistent aggregation factors:** Aggregation factors are generated in a two-step procedure. First, simple aggregation factors are calculated according to the principle of 'free expansion', *i.e.* the number of real farms in the stratum is divided by the number of sample farms of a stratum. In a second step, consistency among the aggregated farms is obtained by adapting the simple aggregation factors using a maximum entropy estimation procedure (Golan *et al.*, 1996), as described by Sanders (2007). The consistency criteria comprised total utilised agricultural area (UAA), arable land, grassland, and total livestock units (LU). In order to avoid extreme shifts in the aggregation factors and to facilitate the solvability of the cross-entropy model, deviations of 1 to 5 % (depending on the criterion) were permitted. The median of average aggregation factors of farm groups was 17.98. The distribution of average aggregation factors of farm groups is displayed in Figure 36, Annex C.

Table 17 shows the number of a) farm groups, b) FADN farms and c) farms represented by each stratum for the strata selected for this study. It also shows the theoretical degree of representation of the selected FARMIS sample as compared to the total FADN sample. In total, 44,838 conventional and 4,888 organic farms were represented with this stratification by FARMIS out of a total of 51,385 (mean from 2006 and 2007) of all FADN-eligible farms (Meier, 2005). This results in a theoretical total representativeness of the joint 2006 and 2007 FADN sample of 96.8 %. For all farm groups except for speciality crop farms (82.3 %), the representativeness is higher than 95 %. The individual strata were represented by 1 to 13 separate farm groups. The minimum number of FADN farms for a separate stratum was four (for the farm group 'organic suckler cow farms in the lowlands'). The figures in Table 17 suggest that it is not advisable to analyse individual strata because of the lack of sufficient FADN farms for some strata. Furthermore, it is not possible to evaluate the totals (across all regions) of arable crops, speciality crops, other grassland farms and pig and poultry farms strata, as there are either not enough organic farms in the FADN sample or none at all. Section 7.1 shows how strata are aggregated for the analysis in order to deal with this problem and obtain valid and reliable results.

Table 17 Farm-group stratification on the basis of FADN data (based on identical farms in 2006 and 2007 FADN sample)

Farm type	Number of farms and farm groups	Lowlands		Hills		Mountains		Total		Farm-type represent-ativeness***
		CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	**ORG**	
Arable crop farms	Farm groups	1		1				2		**99.3%**
	FADN farms	100		6				106		
	Farms represented	3,440		117				3,557		
Speciality crop farms	Farm groups	1	**1**					1	**1**	**82.3%**
	FADN farms	67	**5**					67	**5**	
	Farms represented	2,959	**193**					2,959	**193**	
Dairy farms	Farm groups	3	**1**	6	**2**	6	**3**	15	**6**	**97.8%**
	FADN farms	205	**25**	385	**54**	299	**113**	889	**192**	
	Farms represented	2,601	**240**	5,548	**573**	5,323	**1,275**	13,473	**2,088**	
Suckler cow farms	Farm groups	1	**1**	2	**2**	3	**3**	6	**6**	**100.0%**
	FADN farms	28	**4**	35	**11**	27	**38**	90	**53**	
	Farms represented	490	**91**	735	**247**	609	**732**	1,834	**1,070**	
Other grassland farms*	Farm groups	1		2	**1**	6	**3**	9	**4**	**95.8%**
	FADN farms	12		11	**4**	131	**19**	154	**23**	
	Farms represented	921		1,417	**89**	3,404	**715**	5,742	**804**	
Pig and poultry farms	Farm groups	2		2		1		5		**97.5%**
	FADN farms	35		26		8		69		
	Farms represented	760		586		174		1,520		
Mixed farms**	Farm groups	12	**4**	13	**4**	4	**2**	29	**10**	**98.3%**
	FADN farms	762	**41**	273	**18**	41	**10**	1,076	**69**	
	Farms represented	10,933	**486**	3,975	**164**	845	**82**	15,753	**733**	
Total farms	Farm groups	21	**7**	26	**9**	20	**11**	67	**27**	**96.8%**
	FADN farms	1,209	**75**	736	**87**	506	**180**	2,451	**342**	
	Farms represented	22,104	**1,010**	12,379	**1,073**	10,355	**2,804**	44,838	**4,888**	
	Farms in the population	**23,442**		**14,041**		**13,903**		**51,386**		
	Representativeness***	**98.6%**		**95.8%**		**94.7%**		**96.8%**		

Source: own calculation based on FADN and FSS data

* merged farm group of other cattle farms and horses, sheep and goat farms

** merged farm group from combinded dairy/arable crops, combinded suckler cows, combinded pigs/poultry and combined other farms

*** share of farms represented by the FARMIS sample of the farms represented by the total FADN sample

6.3.3 Generation of input-output coefficients

FARMIS distinguishes between 46 crop production and 27 animal production activities (listed in Table 18). Levels for each activity were taken from the FADN book-keeping records. To specify the production data for the activities, input-output coefficients (IOC) were generated for each activity, farming system and sub-activity.

Table 18 Overview of farm activities represented in CH-FARMIS

Crops activities (ha)		Livestock (yearly livestock housing system place)
Wheat	Rape	Dairy cows
Intensive wheat	Intensive rape	Dairy breeding heifers 04-12 months
Extensive wheat	Extensive rape	Dairy breeding heifers 12-24 months
Rye	Other oilseed crops	Dairy breeding heifers 24-30 months
Intensive rye	Sunflower	Dairy breeding bulls 4-12 months
Extensive rye	Field beans	Dairy breeding bulls 12-24 months
Spelt	Field peas	Dairy breeding bulls 24-30 months
Intensive spelt	Tobacco	Dairy calves for breeding 1-4 month
Extensive spelt	Vegetables	Suckler cows
Other bread cereals	Other arable crops	Suckler breeding heifers 12-24
Barley	Mixed fallow land	Suckler fattening heifers >12 months
Intensive barley	Rotational fallow land	Suckler breeding bulls>12 months
Extensive barley	Buffer strips on arable land	Suckler fattening bulls >12 months
Oats	Short-term ley	Suckler calves 1-12 month
Intensive oats	Meadows	Fattening cattle
Extensive oats	Extensive meadows	Fattening calves
Triticale	Less intensive meadows	Sows for piglet production
Intensive triticale	Intensive used meadows	Pork fattening
Extensive triticale	Pasture	Broiler
Other fodder cereals	Extensive pasture	Laying hens
Intensive other fodder cereals	Alpine meadow	Other poultry
Extensive other fodder cereals	Vineyards	Horses
Grain maize	Fruits	Milk sheep
Fodder maize or silage maize	Berries	Fattening sheep
Potatoes	Other permanent crops	Goats
Sugar beet	Other area	Other roughage consuming livestock
Fodder root crops	Wood	Other animals

Source: Sanders (2007), adapted

In general, input-output coefficients were based on FADN data and supplemented with several further standard and normative data sources.

Output coefficients for **sales revenues** of activities with farm-specific FADN yield data (milk and cereals) were taken directly from FADN accounts. **Yields**, for which no farm-specific data were available, were estimated on the basis of monetary revenues, activity levels of each farm, and standard data on yields[49] and prices[50]. Plausibility checks were conducted for each farm and activity, whereas implausible data[51] was adapted using normative data. A list of all products defined in CH-FARMIS can be found in Table 79, Annex B.

Input-output coefficients for **direct payments** were derived from book-keeping accounts and official premium data. Direct payments, obtained from book-keeping records, were usually lower than the total sum of official payments. The difference was attributed to farms not being eligible for a certain direct payment, e.g. if the payment depends on the slope of land (hillside payments) or specific farm management (e.g. extensive grains and rape) or penalties due to non-conformance with cross-compliance rules.

For obtaining **fertiliser** input-output coefficients, which are both farm and activity-specific and yield-dependent, a cross-entropy model (Leon *et al.*, 1999) was used, as the farm accounts contain only total fertiliser costs. The cross-entropy model calculates the most likely distribution of fertiliser applications and manure nutrient contents of nitrogen, phosphorus and potassium fertilisers to the different crop and livestock activities. The *a-priori* information needed for this model was estimated on the basis of ACW and ART (2009).

Feedstuffs input-output coefficients were also calculated using a cross-entropy model, which estimates the feed use for livestock activities based on farm-specific spending on fodder and activity levels. Standard rations, daily feeding of roughage fodder, number of days with

[49] Agridea Deckungsbeitragskatalog 2004-2008, grassland yields based on Agridea Planungsordner 2008

[50] Agridea Preiskatalog 2004, 2005, 2006, 2007 and 2008

[51] Plausibility ranges were defined on the basis of various expert statements

winter and summer feeding, and weights of farm animals were specified using standard data[52] and adapted on the basis of expert opinions in specific cases.

Other input-output coefficients, e.g. depreciation on machines and buildings, interest rates, repair, fuels, electricity, insurance and labour, were taken from farm accounts and supplemented with standard data[53] where necessary.

Sub-activities for modelling policy uptake are described in Section 6.3.6. For the generation of environmental and PRTC input-output coefficients, see Sections 6.3.7 and 6.3.8.

6.3.4 Model specification

The main model is specified as a programming model which optimises an objective function subject to a set of resource and policy constraints. As the objective function consists of a non-linear (quadratic) term, representing 'hidden costs', *i.e.* costs which are not directly considered in the other model terms, FARMIS can be classified as a positive mathematical programming (PMP) model. The model specification is based on a description by Sanders (2007).

Objective function

Equation 18 shows the standard objective function of CH-FARMIS, without the application of the Röhm-Dabbert approach (see Section 6.3.6), based on Sanders (2007). The formulation of the objective function represents the primal problem as opposed to the dual problem, *i.e.* minimising the cost or input subject to a fixed output level (Chiang and Wainwright, 2005). The income (Z) of each farm group is maximised allowing for revenues from agricultural production, direct payments, fixed and variable cost components. Furthermore, a linear and non-linear PMP cost term is subtracted from the farm income. The derivation of the PMP parameters is explained in detail in Section 6.3.5.

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[52] ALP-Fütterungsempfehlungen, Agridea Deckungsbeitragskatalog 2008, Agridea Planungsordner 2008, Agridea Datenblätter Milchkühe (www.agrigate.ch)

[53] ART Maschinenkatalog 2008, ART-Arbeitsvoranschlag 2008, Wirzkalender 2008, topic-specific experts

The first term of the objective function sums up the revenues for marketed products. The second term adds up all direct cost components per activity. This encompasses various types of expenditure on seeds, crop protection, purchased fodder, veterinary services, animal medicines, primary energy, insurances, and contract work. The third term covers revenues from direct payments, while PX_{ni} specifies the grade of eligibility of the farm for a certain activity (which is relevant, for example, for hill-side payments). The fourth, fifth and sixth term comprise costs for employed labour force (not including contract work), purchased fertilisers and rented land.

$$MaxZ_n = \sum_j p_{nj} Y_{nj} - \sum_i c_{ni} X_{ni} + \sum_i dp_{ni} PX_{ni} - \sum_u r_{nu} U_{nu}$$

$$- \sum_v r_{nv} V_{nv} - \sum_l r_{nl} LAND_{nl} - \sum_i \delta_{ni} X_{ni} - 0.5 \sum_i \omega_{ni} X_{ni}^2 \qquad \forall n \tag{18}$$

where:

$Y_{njk}, X_{ni}, PX_{ni}, U_{nu}, V_{nv}, LAND_{nl} \geq 0$

Indices:
- n = index for farm groups
- i = index for production activities
- j = index for output products
- l = index for land type
- u = index for labour
- v = index for fertilisers

Variables:
- Z = objective (income per farm group)
- Y = sales of agricultural products
- X = level of activities
- PX = level of activities eligible for direct payments
- U = level of labour input/requirements
- V = level of fertiliser input/requirement
- LAND = level of rented UAA

Parameters:
- p = prices for agricultural products
- c = activity-specific costs
- dp = activity-specific direct payments
- r = variable costs
- δ = parameter for linear hidden cost
- ω = parameters for quadratic hidden cost (depending on the alternative intensity levels)

Resource and farm management-related constraints

Apart from the fact that variables need to be greater than zero (non-negativity constraint, see also model calibration, page 124), Equation 18 is subject to a number of constraints, which limit the level of agricultural income (Table 19). **Land** use-related equations include Equation 19, which requires that the sum of all grassland, arable land and permanent crop land activities minus the rented land does not exceed the available land resources of a farm group. Equation 20 states that the sum of all crop activities for all farm resources in each region must not be greater than the available land resources in each region. Equation 21 postulates that the sum of rented land in the target year is equal to the sum of rented land in the base year.

Table 19 Resource and farm management-related restrictions within CH-FARMIS

Restriction related to	Equation		No
Land	$\sum_{l} X_{nl} - LAND_{nl} \leq b_{nl}$	$\forall n$	**(19)**
	$\sum_{n} X_{n_r l} \leq rx_{rl}$	$\forall n$	**(20)**
	$\sum_{n} LAND_{nlBASYR} + \sum LAND_{nlTGYR} > 0$	$\forall n$	**(21)**
Labour	$\sum_{i} (X_{ni} * (\sum_{c} \varphi_{nit} + \varsigma_{ni})) - \sum_{u} U_{nu} \leq 0$	$\forall n$	**(22)**
	$\sum_{i} (X_{ni} * \varphi_{nit}) - \sum_{u} T_{ntu} \leq 0$	$\forall n, u, c$	**(23)**
	$T_{ntu_{nop}} - \eta_t * \sum_{i} (X_{ni} * (\varphi_{nit} + \varsigma_{ni} * \eta_t)) \leq 0$	$\forall n, u, c$	**(24)**
Feeding	$(F_{nif} * \chi_{fm,'drymat'}) \geq (X_{ni} * DM_{ni} * \tau_{\min})$	$\forall n, i, f$	**(25)**
	$(F_{nif} * \chi_{fm,'drymat'}) \leq (X_{ni} * DM_{ni} * \tau_{\max})$	$\forall n, i, f$	**(26)**
	$\sum_{m} (F_{nif} * \chi_{fm}) \geq (X_{ni} * \upsilon_{nim})$	$\forall n, i, f$	**(27)**
Fertilizer	$(\sum_{i_{cro}} X_{ni_{cro}} * \vartheta_{nvi_{cro}}) - (\sum_{i_{liv}} X_{ni_{liv}} * \mu_{nvi_{liv}}) - V_{nv} \leq 0$	$\forall n, v$	**(28)**
Young stock	$\sum_{i_{liv}} (X_{ni_{liv}} * \psi_{ni_{liv}}) = 0$	$\forall n$	**(29)**
Production	$\sum_{i} (X_{ni} * o_{nij}) - W_{nj} - Y_{nj} = 0$	$\forall n, i, j$	**(30)**
Direct payment	$PX_{ni} - X_{ni} \leq 0$	$\forall n, i$	**(31)**
	$(X_{n_r i_{gras}} * \kappa_r) - PX_{n_r i_{liv}} \geq 0$	$\forall n, i$	**(32)**
ECA	$\sum_{i_{eca}} X_{ni_{eca}} * \sigma_{i_{eca}} \geq \sum_{i} X_{ni}$	$\forall n, i$	**(33)**
Organic farming	$V_{n_{org} v,'nitr'} = 0$	$\forall n$	**(34)**
	$(F_{nif} * \chi_{fm,'drymat'}) * 0.5 \geq (X_{ni_{cro}} * o_{ni_{cro}} * \chi_{fm,'drymat'})$	$\forall n, i$	**(35)**

Source: Sanders (2007)

where:

f	=	index for feedstuffs
i_{cro}	=	index for crop activities
i_{eca}	=	index for crop activities defined as ECA
i_{gras}	=	index for grassland activities
i_{liv}	=	index for livestock activities
m	=	index for the nutritional value of feedstuffs
n_{org}	=	index for organic farms
r	=	index for regions
t	=	index for time periods
u	=	index for labour
u_{nop}	=	index for non-permanent labour
v	=	index for different types of fertiliser
DM_{ni}	=	dry matter consumption of livestock (in t)
F_{nif}	=	used feedstuffs (in t)
$PX_{ni_{liv}}$	=	number of livestock eligible for direct payments (in LHU)
T_{ntu}	=	labour input for different time periods (in 1,000 h)
U_{nu}	=	labour requirements (in 1,000h)
W_{nj}	=	home-produced feedstuffs used on the farm (in t)
X_{nl}	=	level of production activities (in ha/LHU)
X_{ns}	=	level of production activities (in ha/LHU)
b_{nl}	=	available land resources (in 1,000 ha)
b_{nu}	=	available labour resources (in 1,000 h)
rx_{rl}	=	total regional land area (1,000 ha)
ς_{ni}	=	labour requirements related to no specific seasonal period (in 1,000 h)
η_t	=	proportion of work that needs to be done by permanent labour (in %)
$\vartheta_{nvi_{cro}}$	=	nutrient requirements of crops (in t)
κ_r	=	maximum stocking rate (LU/ha)
$\mu_{nvi_{liv}}$	=	nutrient content of manure (in t)
o_{nij}	=	marketable output of each production activity (in 1000 CHF / t)
$\sigma_{i_{eca}}$	=	proportion of ECA (in %)
$\tau_{\max}$	=	coefficient defining the upper level of the feed ration (in %)
$\tau_{\min}$	=	coefficient defining the lower level of the feed ration (in %)
υ_{nim}	=	nutritional requirements of farm animals (in t)
φ_{nit}	=	Labour requirements related to specific seasonal periods (in 1,000 h)
χ_{fm}	=	Nutritional value of feedstuff (in t)
$\psi_{ni_{liv}}$	=	Output and input of young stock (in numbers)

Generally, **labour** restraints imply that the labour requirements of the farm activities do not exceed the available resources plus the employed labour. Equation 22 postulates that labour requirements over all seasons are met, while Equation 23 specifies the coverage of labour demand for each season. It is assumed that management-related tasks cannot be fulfilled by non-permanent staff. Therefore, Equation 24 requires these type of tasks to be covered by permanent labour (Sanders, 2007).

Feeding requirements are expressed in Equations 25 and 26, which ensure that the feeding proportions for each feedstuff in the ration differ only within a certain range from the ratios in the base year. Furthermore, Equation 27 postulates that all nutritional requirements (energy, protein, dry matter and crude fibre) of the livestock have to be met by the substances fed (Sanders, 2007).

The **fertiliser** balance (Equation 28) requires that nutrient needs by crops are covered by organic fertilisers, legumes (for nitrogen), and purchased mineral fertilisers. Therefore, if nutrient requirements exceed available nutrients on the farm, the farm group has to purchase additional mineral fertiliser. However, purchasing mineral nitrogen fertiliser is not allowed for organic farms (Equation 34). Organic speciality crop farms, which in reality do purchase nitrogen fertiliser (organic manure, etc.), are not subject to this restriction. Furthermore, it is assumed that organic farms do not fertilise according to crop needs[54], which is common practice according to Dierauer (2009). Therefore, the demand for fertiliser is multiplied by a factor of 2/3 for organic farm groups.

Regarding **young cattle and pig livestock**, the internal rate of transaction between livestock activities must be zero (Equation 29). Replacement costs for all other activities are included in the respective input-output coefficients. Concerning **product sales**, a balance of all physical outputs is given by Equation 30. **Direct payments** for both livestock under adverse conditions and hillside payments depend on natural geographic conditions and thus can not be influenced by farm management. Therefore, according to Equations 31 and 32, the share land and livestock eligible for these payments has to remain constant in the scenarios.

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[54] ‘Organic farmers do not fertilise the crop but fertilise the soil’, refers to the fact that organic farmers think in longer terms over a whole rotation and accept sporadic nutrient deficiencies (Dierauer 2009).

As part of the **proof of ecological performance**, farms have to hold 7 % of their UAA as ecological compensation area (ECA). This is postulated by Equation 33.

In addition to Equation 34, **organic farms** have to fulfil the condition that at least 50 % of the feedstuffs are produced on-farm, *i.e.* in the model within the same farm group (Equation 35).

6.3.5 Model calibration

Positive mathematical programming (PMP) facilitates the exact reproduction of the base-year situation and solves the LP problem of overspecialisation and extreme responses (Howitt, 1995). Nonetheless, drawbacks – most prominently the lacking empirical foundation – have to be accepted. A comprehensive discussion of PMP in contrast with econometric estimation procedures is provided by Heckelei (2002).

The calibration procedure is structured in three parts. First, the primal LP model is extended by calibration constraints. Second, dual values are used to specify a non-linear objective function, and third, the non-linear objective function is defined. This description is based on Heckelei (2002), Bertelsmeier (2005), Kuepker (2004), and Sanders (2007).

Extend the primal LP model by calibration constraints

The primal LP model in a generic, simplified version is displayed as Equation 36:

$$\max Z = \sum_i (p_i y_i + DP_i - c_i)\, x_i \qquad \textbf{(36)}$$

where Z is farm income, x_i is activity level, p_i is output price, y_i level of output, DP_i are direct payments and c_i the cost per unit of activity i (either ha or LU). This objective function is maximised subject to the constraints specified in Equations 37 to 39:

$$\sum_i a_{ik}\ x_i \le b_k \quad \forall k \quad [\pi_k] \qquad \textbf{(37)}$$

$$x_i \le x_i^* + \varepsilon \qquad \forall i \qquad [\lambda_i] \qquad \textbf{(38)}$$

$$x_i \ge 0 \qquad \forall i \qquad \textbf{(39)}$$

where a_{ik} is the required amount of input k for one activity i, b_k is the total available amount of input k, x^*_i observed level of activity i in the base year. ε is a perturbation parameter, needed to avoid 'degenerated solutions' (Cypris, 2000). Equation 37 is the resource constraint, while Equation 38 is called calibration constraint. There is a resource constraint for each resource needed and a calibration constraint for each activity. The dual value for each resource constraint receives the name π_k. The dual value for each calibration constraint is λ_i. Finally, the non-negativity constraint (Equation 39) ensures that the activity levels do not take on negative values. See Bertelsmeier (2005) for a description of the dual optimisation problem. Activities are grouped into:

a) preferred activities x_p with binding calibration constraints (λ_p) and

b) marginal activities x_m, without binding constraints (λ_m).

Supposed that all constraints are binding and all activity levels are greater than 0 (Kuhn-Tucker conditions) (Chiang and Wainwright, 2005), the dual values are calculated as shown in Equation 40, 41 and 42 (Bertelsmeier, 2005):

$$\lambda_p = p_p - c_p - \sum_k a_{pk}\pi_k \quad \forall p \qquad \textbf{(40)}$$

$$\lambda_m = 0 \quad \forall m \qquad \textbf{(41)}$$

$$\pi_k = (p_m - c_m)(a_{mk})^{-1} \quad \forall k \qquad \textbf{(42)}$$

Derive dual values

The dual values of the calibration constraints for preferred activities (λ_p) are used to specify the non-linear objective function in order to adapt the preferred activities' marginal costs to respective marginal returns at the observed activity levels. In principle, the cost function could have any convex form. However, FARMIS uses a quadratic function (Equation 43) because it is easily solved and therefore used in most non-linear programming models. The additional linear and quadratic costs terms are added to the variable costs:

$$C_i = c_i x_i + \delta_i x_i + \frac{1}{2}\omega_i x_i^2 \quad \forall i \qquad \textbf{(43)}$$

where δ_i and ω_i are calibration parameters which have to be estimated and added up to the variable cost of a production activity. The linear and non-linear cost terms can be interpreted as additional costs that cannot be taken into account directly by the model. The reasons for such costs occurring could be adverse soil and weather conditions, investment barriers or risks.

According to the model, the activity level is determined at the point where marginal cost curve and marginal revenue curve intersect. Thus the first derivative of the cost function (Equation 43) and the revenue function have to be taken. The first derivative of Equation 43 is Equation 44 and the first derivative of the revenue function (MR) is Equation 45:

$$\frac{\partial C_i}{\partial x_i} = c_i + \delta_i + \omega_i x_i \quad \forall i \tag{44}$$

$$MR_i = \frac{\partial R_i(x_i^*)}{\partial x_i} \quad \forall i \tag{45}$$

where R_i represents the revenues for activity i.

Equating marginal revenues with marginal costs results in Equation 46.

$$\frac{\partial C_i(x_i^*)}{\partial x_i} = \frac{\partial R_i(x_i^*)}{\partial x_i} = c_i + \delta_i + \omega_i x_i \quad \forall i \tag{46}$$

As described above, there are an infinite number of valid combinations for δ_i and ω_i. Each combination results in a different slope of the marginal cost function, which determines the adoption behaviour of farmers. In order to achieve sound adoption behaviour in the model, the slope is derived from the supply elasticity $\varepsilon_i^{x,MR}$ of the revenues (MR) as displayed in Equation 47.

$$\varepsilon_i^{x,MR} = \frac{\partial x_i^*}{x_i^*}\left(\frac{\partial MR_i}{MR_i}\right)^{-1} \quad \forall i \tag{47}$$

Equation 48 represents the slope term of the supply elasticity of marginal revenues equated with marginal costs.

$$\varepsilon_i^{x,MR} = \frac{1}{\omega_i}\frac{MR_i}{x_i^*} = \frac{p_i y_i + DP_i}{\omega_i^{x,MR} x_i^*} \qquad (48)$$

Equation 48, transformed for ω_i, comes to Equation 49 for activities with revenues. The slope term is calculated according to Equation 50, for activities for which the output is used on the farm itself.

$$\Leftrightarrow \omega_i = \frac{p_i y_i + DP_i}{\varepsilon_i^{x,MR} x_i^*} \quad \forall i \qquad (49)$$

$$\omega_i = \frac{(\lambda_i + c_i)}{x_i^*} \quad \forall i \qquad (50)$$

The non-linear PMP term of the objective function has now been specified. Additionally, a linear term is used for shifting the marginal cost function parallel to cut the marginal revenue function at X^*_i. Equation 51 shows the calculation of δ_i for the linear PMP term. This formulation ensures the exact calibration of the PMP model, as δ_i equals the marginal cost minus the quadratic cost parameter ω_i multiplied by the observed level x_i^*.

$$\delta_i = \lambda_i - \omega_i x_i^* \quad \forall i \qquad (51)$$

Define the non-linear objective function

The parameters derived from above can be used for the linear optimisation problem, which hence becomes non-linear and calibrates exactly according to the observed base year solution without calibration constraints, due to the last term of Equation 52. Only the resource constraints (Equation 34) and the non-negativity condition are needed (Equation 35).

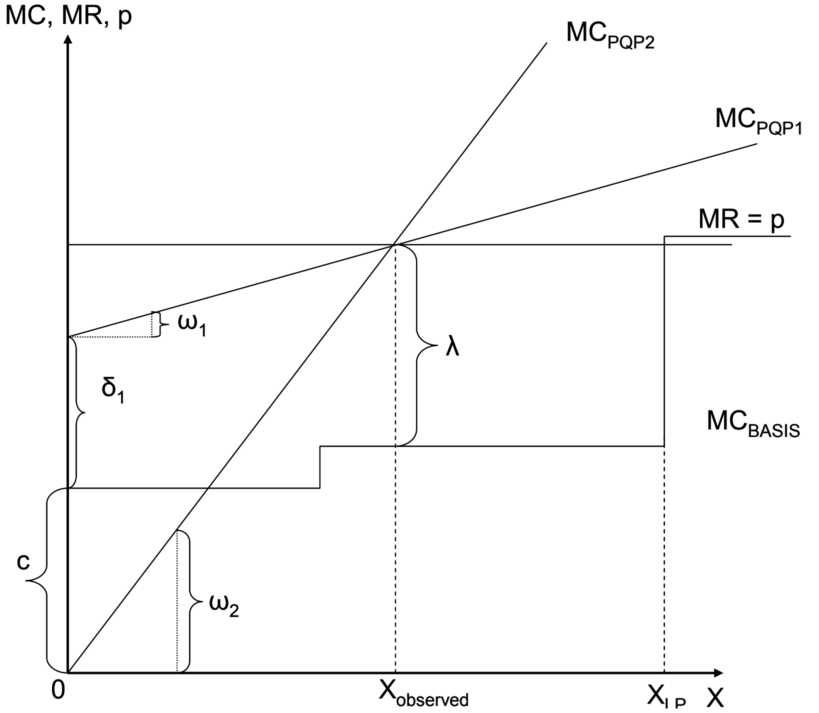
$$\max Z = \sum_i (p_i y_i + DP - c_i) x_i - \delta_i x_i - \frac{1}{2}\omega_i x_i^2 \qquad (52)$$

$$\sum_i a_{ik} x_i \le b_k \quad \forall k \quad [\pi_k] \qquad (53)$$

$$x_i \ge 0 \quad \forall i \qquad (54)$$

The PMP method can be illustrated by means of the graph presented in Figure 14, which shows the marginal cost function (MC) and the marginal revenue function (MR). While MR is assumed to be constant[55], the MC function shows *ad-hoc* shifts when additional constraints have to be met in the linear programme. A simple linear programming solution would lead to activity level X_{LP}, where MC intersects MR. Empirical data, however, show that the activity was implemented only by the level of $X_{observed}$. To adhere to neo-classical economic theory, *i.e.* assuming the rational behaviour of farmers, MC and MR must be equal at $X_{observed}$. If MC exceeded MR at $X_{observed}$, farmers would undertake the activity to a lesser extent. If MR exceeded MC at $X_{observed}$, farmers would further extend the activity until MC intersects MR.

In order to explain the difference between X_{LP} and $X_{observed}$ economically, Howitt (1995) suggests hidden costs, *i.e.* costs which cannot be included in the model directly[56]. Therefore, PMP models implicitly make the assumption that farmers behave rationally and that the empirically observed activity levels in the base year are the result of profit maximising behaviour, which takes into account all natural conditions and personal attitudes of farmers.



Source: Bertelsmeier (2005)

Figure 14 Simplified graphical representation of the PMP approach

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[55] Equalling price*yield + direct payments of an additional infinitesimal small unit of additional activity.

[56] Hidden cost could be anything from locally-specific natural conditions to risk aversion by farmers.

When a calibration constraint is introduced that forces the X_{LP} to equal $X_{observed}$, the hidden cost – *i.e.* costs that real farmers took into account in the base year but are not explicitly modelled – equals the dual value of the calibration constraint (λ). In order to reproduce exactly the $X_{observed}$ in the model, MC is specified to intersect MR at $X_{observed}$ using the parameters δ and ω, which are calculated according to Equation 50 and 51. Thus, MC_{PMP1} shows the marginal cost for activities with marketable products, while MC_{PMP2} shows the marginal cost for activities with non-marketable products ($\delta_2 = -c$).

6.3.6 Uptake of agri-environmental policies

Production economics (Samuelson and Nordhaus, 2007) form the foundation for modelling the uptake of agri-environmental policies in the overall FARMIS approach. Farm-level costs and their compensation via direct payments determine the relative profitability and therefore the policy uptake of the farms modelled. As shown in Section 6.1.1, several empirical studies and statistical data sources reveal that farmers take up agri-environmental policies on fields and plots which are less suitable for production (Salhofer and Glebe, 2006; Schader *et al.*, 2009b). This suggests that economic models based on the assumption of rational behaviour of farmers are a feasible means of estimating aggregate uptake levels for agri-environmental policy.

By contrast, organic farming area support payments (OFASP) cannot be modelled using purely production economic assumptions (Padel, 2001; Schmid and Sinabell, 2006b), as the decision to convert or re-convert depends on multiple factors, including social and cultural environment, neighbour effects and personal attitudes (Bichler *et al.*, 2005; Hollenberg, 2001; Kerselaers *et al.*, 2007).

The uptake of agri-environmental policies needs to be modelled differently from ordinary activities, since farmers' decisions follow a different rationale than when switching between standard activities. The Röhm-Dabbert approach (RDA) (Röhm and Dabbert, 2003) presents a more realistic model of behaviour by defining intensity levels, according to the uptake or non-uptake of an agri-environmental policy. These intensity levels are treated by the model as 'similar activities', *i.e.* activities which entail similar requirements in terms of machinery and labour input. Without the definition of similar activities, all activities are exchanged according to the PMP coefficients derived in Section 6.3.5. However, in reality farmers are able to switch easily between different intensity levels without replacing all their machinery or other farm processes. In contrast, switching from, say, wheat production to grassland requires many changes on the farm, considered in the model as a farm's hidden costs. Since these hidden costs differ depending on whether farms switch from one intensity level to the other or whether they switch between activities, different PMP cost terms need to be included in the objective functions. There are two types of quadratic hidden cost parameters (ω) in the extended objective function (Equation 55). This implies that hidden costs are split into a) a share which depends on the level of the intensity (with ω_{n1} as slope coefficient), and b) a share which depends on the level of the other intensities of a particular activity (with ω_{n2} as slope coefficient), while δ ensures the exact calibration of the intensity levels according to the empirically observed levels in the base year (Kuepker, 2004; Röhm and Dabbert, 2003).

$$\begin{aligned} \max Z_n = & \sum_j \sum_k p_{njk} Y_{njk} - \sum_i \sum_k c_{nik} X_{nik} + \sum_i \sum_k dp_{nik} PX_{nik} - \sum_u r_{nu} U_{nu} - \\ & \sum_v r_{nv} V_{nv} - \sum_l r_{nl} LAND_{nl} - \sum_i \sum_k \delta_{nik} X_{nik} - 0.5 \sum_i \omega_{ni1} X_{nik}^2 - 0.5 \sum_i \sum_w \omega_{ni2} X_{niw}^2 \qquad \forall n \end{aligned} \tag{55}$$

$Y_{njk}, X_{ni}, PX_{ni}, U_{nu}, V_{nv}, LAND_{nl} \geq 0$

where:

Indices:
- n = index for farm groups
- i = index for production activities
- j = index for output products
- k = index for intensity levels
- w = index for intensity levels $\neq$ w
- l = index for land type
- u = index for labour
- v = index for fertilisers

Variables:
- Z = objective (profit per farm group)
- Y = sales of agricultural products
- X = level of activities
- PX = level of activities eligible for direct payments
- U = level of labour input/requirements
- V = level of fertiliser input/requirement
- LAND = level of rented UAA

Parameters:
- p = prices for agricultural products
- c = activity-specific costs
- dp = activity-specific direct payments
- r = variable costs
- δ = parameter for linear hidden cost
- ω = parameters for quadratic hidden cost (depending on the alternative intensity levels)

Equations 56-58 show how ω_{n1} and ω_{n2} are derived. J is the number of optional intensity levels for a given activity and ϕ is the PMP elasticity factor, which determines the sensitivity of the exchange of intensity as compared to the total activity level. ϕ takes on predefined values ranging from 0 to 1. While ϕ =1 would make ω_{n2} become 0 and let the model treat the intensity levels like a standard PMP model, ϕ =0 makes the exchange of intensity levels elastic like in a linear programme, while the total activity level would remain unaffected. A standard value of ϕ =0.5 was applied to all activities due to the lack of empirical data on the exchange elasticity of intensities. In addition to plausibility checks in advance of the model runs, a sensitivity analysis was conducted to check to what extent the choice of the parameter ϕ changes the results of the analysis (see Section 7.5.2). ω_{ni} corresponds to the derivation of the standard PMP coefficient (see Equation 49 and 50), while, unlike the standard formulation, the numerator is divided by the squared activity level because of the two-dimensional character (activities and intensities) of the problem. Finally, ω_{n2} characterises the parameter for the non-linear PMP term of the alternative intensity levels (Kuepker, 2004).

$$\omega_{ni1} = \omega_{ni}((J-1)\phi_i + 1) \tag{56}$$

with

$$\omega_{ni} = \frac{\sum_j x_{ij}(p_i y_i + DP_i)}{\varepsilon_i^{x,MR} x_i^2} \quad \forall n,i \tag{57}$$

$$\omega_{ni2} = \frac{\omega_i J - \omega_{1i}}{J-1} \quad \forall n,i \tag{58}$$

In consequence, specific non-diagonal elements of the Q-matrix (Heckelei, 2002) receive values other than zero. Therefore, the RDA can be understood as a pragmatic alternative improvement to PMP, defining all non-diagonal elements of the Q-matrix by econometric estimates if a) data for a reliable econometric estimation is not available, b) uptake response on specific policies is in the focus of a research study and c) payment levels varied in the scenarios within an unrealistic range, where available empirical data would have to be extrapolated too far.

Thus, the opportunity cost of agri-environmental policies is considered in the same way as for ordinary activities, namely as the consumption of scarce farm resources: land, labour and capital. Technical costs are included directly within the objective function terms, *i.e.* as input-output coefficients for activity-specific costs (c) and variable costs (r). Since no empirical farm-specific transaction cost data at farm level were available for specifying input-output coefficients farm-level policy-related transaction costs are not modelled explicitly but taken into account indirectly as hidden costs for the uptake decision (δ, ω_{n1}, ω_{n2}).

Accordingly, uptake of agri-environmental policies is modelled by defining separate sub-activities reflecting the uptake choices of farmers. Two types of grassland extensification payments, namely ‘payments for less intensive meadows’ and ‘payments for extensive meadows’ (see Section 4.3.3 for descriptions of the policy measures) are modelled using the RDA. Furthermore, as an agri-environmental policy for arable crops, ‘extenso payments’ (see Section 4.3.4 for a description of the policy) are implemented for conventional farms[57].

These three agri-environmental measures have been selected against the following criteria: First, reliable data need to be available on the environmental impacts and the cost of the measures. Second, the measure has to be relevant in terms of a) environmental effects on the selected impact categories, b) uptake levels and c) total public expenditure. Third, the measure can be modelled using a sector model. Finally, the combination of measures should cover both arable and grassland. As shown in Chapter 4, no policy instruments have yet been implemented, which both fulfil the above criteria and specifically address the environmental categories energy use and eutrophication. Since each intensity level of the selected policy measures is specified in FADN and FSS, the input-output coefficient can be defined according to the standard procedure (explicated in Section 6.3.3).

6.3.7 Environmental impacts at sector level

In this section, the process of determining environmental impact data is described. First, conventions related to life cycle assessment (LCA) as a general framework for environmental impacts are explained (‘goal and scope definition’, ‘inventory analysis’, ‘impact assessment’). In particular, the choice of functional unit and the system boundaries are discussed and specified. With regard to impact assessment, indicator-specific data sources and assumptions are illustrated for the impact categories taken into account: fossil energy use[58], biodiversity and eutrophication with nitrogen and phosphorus.

Many different frameworks are available for analysing the environmental effects of economic activities in a standardised form, ranging from environmental impact assessment, greenhouse gas accounting and ecological footprint to life cycle assessment (LCA). The most commonly

[57] Organic farms are eligible for these payments by definition as the restrictions are covered in any case by the organic farming standards. Organic farms therefore have a fixed uptake level of 100 % for extenso payments.

[58] Technically, ‘fossil energy use’ covers two non-renewable energy components: fossil and nuclear energy. For linguistic reasons the term ‘fossil energy use’ is used henceforth.

applied and comprehensive framework is provided by LCA (Curren, 2006; Heijungs *et al.*, 1992). Life cycle assessment is an ISO-standardised[59] approach (Marsmann, 2000), which is used primarily for assessing the relative environmental impact of different categories (ISO, 2006a; ISO, 2006b). LCAs are usually structured in different phases, as presented in Figure 15. As indicated by the arrows, the steps are not finalised in a chronological order instead, LCA has to be conceived as an iterative approach.

According to ISO (2006a), '*LCA assesses, in a systematic way, the environmental aspects and impacts of product systems, from raw material acquisition to final disposal in accordance with the state goal and scope'*. In contrast to other environmental evaluation methodologies, including particularly environmental impact assessments, environmental performance evaluations and risk assessment, LCA results are always expressed relative to the chosen functional unit, *i.e.* the *'quantified performance of a production system* [is used] *as a reference unit'* (ISO, 2006b). In order to understand the LCA approach, it is essential to distinguish strictly between life cycle inventory analysis (LCI), *i.e.* the gathering of inputs and processes, and life cycle impact assessment (LCIA), *i.e.* the analysis of the environmental impacts of these inputs and processes on environmental impact categories.

Applying LCA in agriculture to compare farming systems entails a number of methodological challenges, particularly for the steps of goal and scope definition and impact assessment. In particular, functional units which are related solely to productivity are not appropriate for such a comparison (Haas, 2003; Nemecek *et al.*, 2004). Geier and Köpke (1997) propose the introduction of agriculture-related impact categories such as animal welfare. Nemecek *et al.* (2005) covered biodiversity and soil quality as two agriculture-related impact categories that are not included in prevalent impact assessment methods.

The following paragraphs describe the methodology used to assess environmental impacts, structured according to the steps of a LCA (Figure 15). First, LCA-related goal and scope definition are explained, focussing on the choice of functional unit and the system boundaries. Second, the inventory data is depicted. Third, impact assessment procedures are shown, while

[59] ISO 14040 described the general principles of LCAs, whereas ISO 14044 (formerly 14041, 14042, and 14043) described more detailed standards for each step of an LCA.

a separate section describes each of the three impact categories (fossil energy use, biodiversity, and eutrophication).

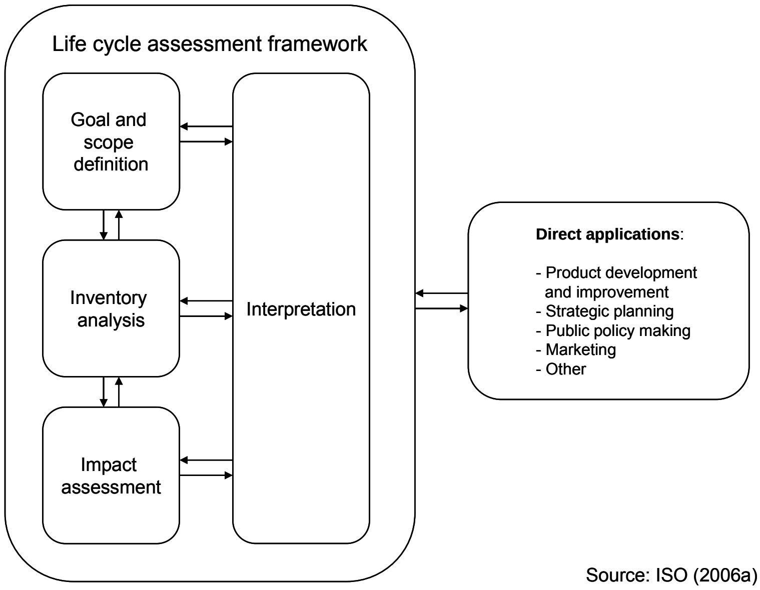


Figure 15 Overview of life cycle assessment approach

Goal and scope definition

For this study, as opposed to production-based LCAs, the ‘utilised agricultural area’ was chosen as a **functional unit**. This choice was made due to the fact that all evaluated direct payments are area-related rather than production-related. For instance, ‘payments for extensive meadows’ are issued for managing a particular hectare of grassland in a restricted way. Furthermore, the functional unit ‘mass of a product’ cannot be applied, since about 40 different products are produced and can not be realistically evaluated individually. An alternative productivity-related functional unit is the energy (calories) for human nutrition produced. This functional unit faces the drawback that different products have different energy concentrations and many products are not produced primarily for their energy content but also for protein or fat. An economic option for a production-related functional unit is to relate environmental impact to the value added by the farms. With this functional unit, the question arises whether the production function of agriculture as a sole indicator is justifiable in a post-productivist or multi-functional policy regime as exists in Switzerland (see particularly Section 4.2.1). Moreover, the argument of consistency between the chosen environmental indicators provides a strong argument for choosing area as a functional unit, since biodiversity, *i.e.* habitat quality, cannot be related to a product, as is common in LCAs.

Finally, as this study employs a regional perspective, eutrophication problems cannot be addressed by an increased productivity approach, as intensified agriculture may improve eutrophication efficiency but will not solve an existing eutrophication problem.

System boundaries were configured specially for this study due to the uncommon scope of the study, namely, sector-representative farm groups. Figure 16 illustrates the definition of system boundaries for this study. Arrows marked with numbers in grey circles were explicitly considered, while arrows with numbers in white circles were disregarded.

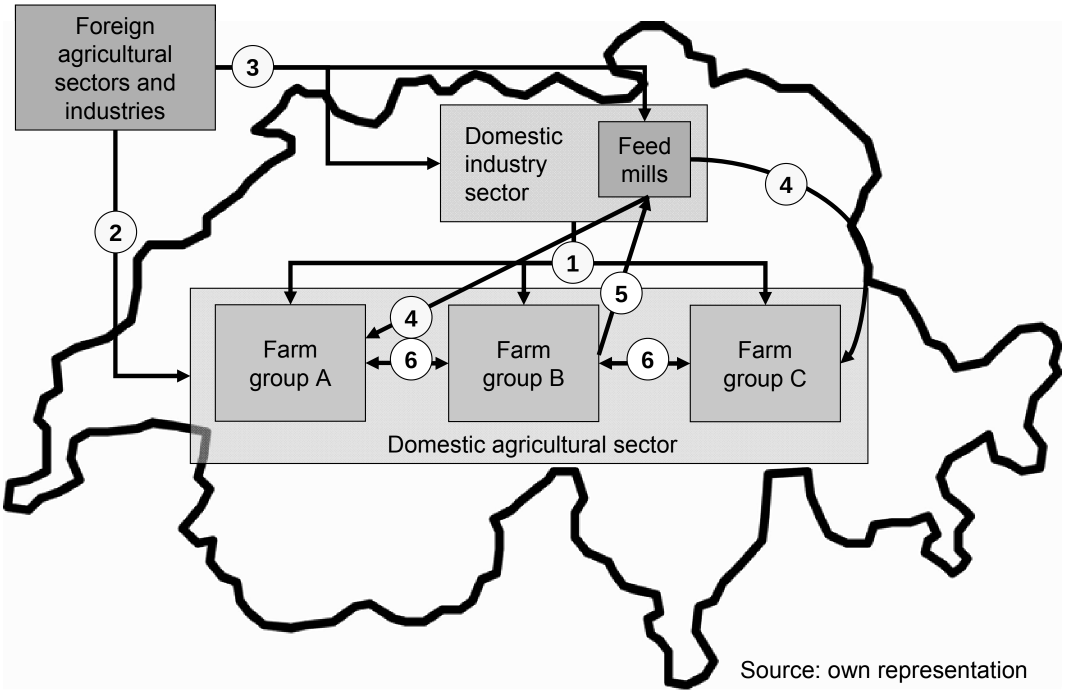
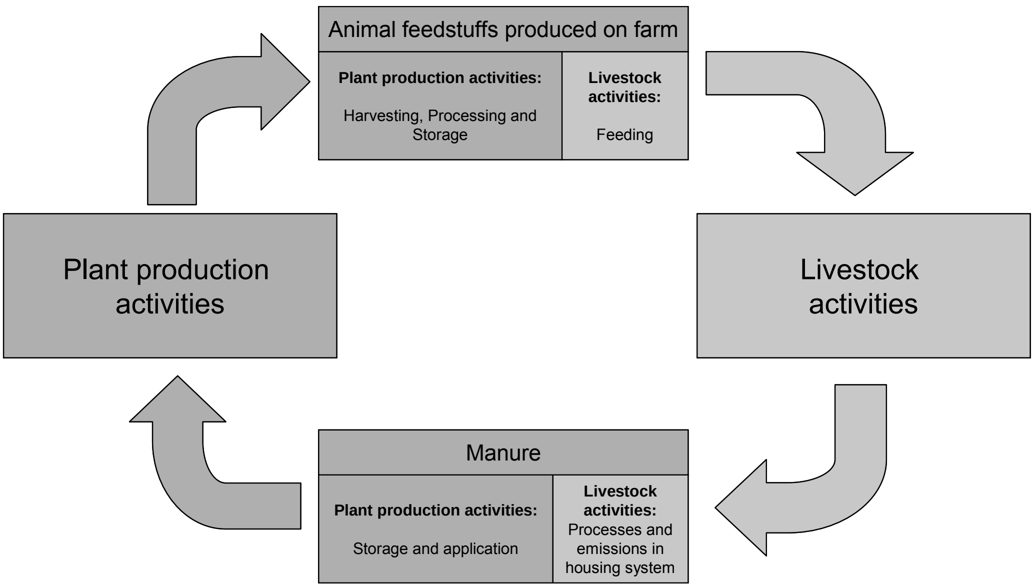


Figure 16 System boundaries of the assessment of sector-representative farm groups

System boundaries include all processes (except irrigation) and inputs that were necessary for the farm groups to cultivate the area in the way they did, *i.e.* according to the principles of the farming system (1). In compliance with LCA methodology, however, foreign inputs for agricultural production were accounted for as a fundamental difference to economic national accounting systems (2). Moreover, the foreign inputs necessary to produce inputs were taken into account (3). For instance, fossil energy use as well as the use of machinery for mining phosphates (e.g. in Morocco), used by the farm groups as a phosphate fertiliser, were included. Concentrates purchased by the farms (4) represent a special case, as they were produced (in part) by the domestic agricultural sector, processed within feed mills, and finally used again as inputs by the farm groups. For the LCAs, only input use is taken into account, while the sale of intermediate products (5) is not subtracted. In particular, this fact does not permit adding up the total fossil energy use for the whole sector, as the fodder cereals produced on Swiss farms would be double counted. Trade between the farm groups (6) was disregarded in the study, as it was assumed that trade of roughage and young animals occurs largely within a farm group.

All inventory data were calculated for a period of 12 months for main crops, whereas green manures were not taken into account in the LCA results. The study employs a ‘cradle-to-gate’ rather than a ‘cradle-to-grave’ perspective, with the gate being the farm group gate. Thus, further steps of the supply chain after agricultural production were disregarded, as they do not depend on farming system characteristics and are therefore not directly related to the research question. System boundaries include capital goods, such as machinery and buildings.

Conventions for system boundaries within the farm groups had to be determined particularly for the linkages between crop and livestock production as shown in Figure 17.



Source: own representation

Figure 17 System boundaries of on-farm processes

The most important linkages are the provision of a farm’s own fodder from crop to livestock activities and the use of livestock manure for crop production activities. Harvesting, processing, and storage of a farm’s own animal feedstuffs (particularly roughage) were linked to crop production activities, while only the feeding process itself was taken into account in the livestock processes. Regarding manure, all processes and emissions within the livestock housing system were linked to livestock activities, while storage and application of the manure was linked to crop production activities.

Life cycle inventory (LCI)

Life cycle inventory data was taken primarily from Swiss Agriculture Life Cycle Assessments (SALCA) data, generated primarily by a Swiss Federal Research Station (*Agroscope Reckenholz-Tänikon (ART)*). Some SALCA data is included in the ecoinvent databases, which includes the necessary external peer reviews (Frischknecht *et al.*, 2007). Assumptions behind these data were reported extensively by Nemecek *et al.* (2004) and Nemecek and Kägi (2007).

Data on the farming activities is differentiated according to farming system (integrated and organic farming), region (valley, hill and mountain region) and therefore compatible with the classical FARMIS farm groupings and the objective of the thesis (Nemecek *et al.*, 2006; Nemecek and Erzinger, 2005). Relevant inventory data for this thesis is included in aggregated form in Table 74 to Table 78 (Annex B). Where feasible, inventory data from FARMIS-endogenous calculations were used (see life cycle impact assessment for details).

Life cycle impact assessment (LCIA)

SALCA data has been calculated for the most relevant impacts of agricultural activities that are typical for Swiss agriculture. Cumulative fossil energy use (CED), biodiversity and eutrophication with nitrogen and phosphorus were chosen as the life cycle impact assessment categories.

Selection criteria for environmental impact categories

As the study had to be limited to a maximum of three impact categories, in order to keep the workload within the scope of a PhD thesis, selection criteria had to be applied with regard to choosing environmental impact categories. As outlined briefly in Chapter 1, impact categories were selected against the following criteria:

1. the importance of the environmental category in the current policy debate,
2. the importance of agriculture for the environmental category,

3. the systematic differences between organic and non-organic farming systems,
4. the feasibility of modelling the environmental indicators at sector level,
5. the availability of comprehensive, quantitative and widely accepted data for Switzerland.

As shown in Chapter 4, Criteria 1 and 2 are given for all selected impact categories, particularly for biodiversity. Chapter 3 revealed systematic differences between the farming systems for all impact categories that were selected (Criterion 3). All the impact categories chosen can be modelled at sector level (Criterion 4), although the considerations expounded in Section 6.1.2 concerning the upscaling of indicators to sector level have to be taken into account. Thus biodiversity results cannot be expressed as cumulative values but only as averages. As shown in this section comprehensive data (SALCA) have been identified (Criterion 5) and used for the study. The impact categories chosen – fossil energy use, biodiversity and eutrophication – are based on the reliable and widely accepted data (Nemecek and Erzinger, 2005). Technically, impact data was linked directly to FARMIS, *i.e.* life cycle impact assessment calculations were not carried out within FARMIS.

In contrast, other environmental impact categories were excluded from the analysis. For Ecotoxicity Criteria 1 to 3 are given, however both Criterion 4 and 5 are not fulfilled and impact assessments would suffer from the lack of precise toxicity impact factors. For climate change all above criteria except Criterion 5 were given, at the time when the impact assessment categories were selected. This is due to uncertainties regarding N_2O emissions. Furthermore, there were no particular policy goals defined for agriculture in relation to climate change. However, against the background of the current high policy relevance of this impact category, an inclusion of climate change into the model seems to be both feasible and relevant.

Cumulative fossil energy use

SALCA fossil energy use data have been taken from Nemecek *et al.* (2005). Data gaps concerning crop activities were closed by making assumptions in agreement with the SALCA team. Apart from the activities covered by Nemecek *et al.* (2005), animal husbandry data (including buildings) were taken directly from the ecoinvent 2.0 database and provided by Mack (2008, personal communication). Data for feedstuffs were provided by ART, taken from the internal SALCA database (Alig, 2007). Assumptions include the geographic origin, means of transportation and processing. The LCIA method underlying these data is Cumulative Energy Demand[60] (CED) (Frischknecht and Jungbluth, 2003). Only the non-renewable energy categories (fossil and nuclear energy) were taken into account. Renewable energy components were disregarded.

Table 20 provides an overview of how the energy-use categories were linked to FARMIS. Generally, LCA data can be linked to either activity/intensity levels, or to endogenously modelled inputs or outputs. Most of the crop production-related categories were linked to farm activities and intensity levels. This includes energy use for seeds, crop protection, fertilisation, mechanisation, organic fertilisation, fences and depots for roughage. For mineral fertiliser use, especially, an alternative linkage to inputs was tested, yet results have not been satisfying, *i.e.* did not correspond with empirical data (FOAG, 2008) or standard data (Meyer, 2008).

Table 20 Interfaces between environmental energy indicators and the FARMIS model

FARMIS Activity type	FARMIS Activity and intensity level	Inputs (FARMIS-endogenous)	Outputs (FARMIS-endogenous)
Crop production activities	• Seeds • Crop protection • Mechanisation and tillage • Organic fertilisation • Purchased mineral fertilisers • Pasture management • Depots		
Animal production activities	• Construction of buildings • Maintenance of buildings • Operation of buildings	• Purchased feedstuffs	• Milking

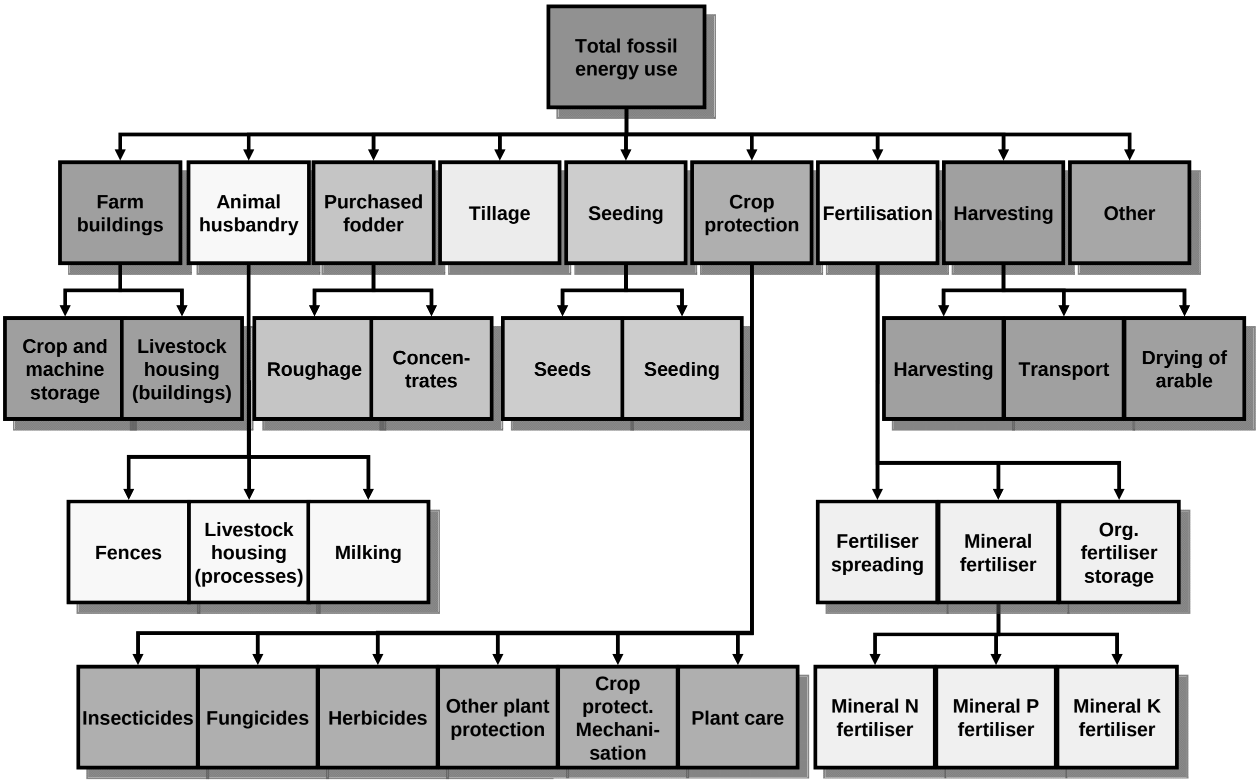
Source: own representation

Figure 18 presents the energy use categories implemented in CH-FARMIS. Total energy use is sub-divided into the categories farm buildings, animal husbandry, purchased fodder, tillage, seeding, crop protection, fertilisation, harvesting and other. Further sub-categories were used for validation and interpretation of the results.

Concerning energy use for livestock activities, construction, maintenance, and operation of buildings were linked to the activity levels. Energy use for purchased feedstuffs was calcu-

[60] Version 1.4 and 1.5

lated on the basis of model-endogenous purchase of feedstuff, while the assumption was made that 50 % of the concentrate feedstuffs is produced within the same farm group. Energy use for transport of concentrates was added to the SALCA inventories. Energy use for milking was linked to the output of milk (milk sales) calculated by FARMIS. Detailed assumptions are presented in Annex B.



Source: own representation, based on Nemecek *et al.* (2005)

Figure 18 Hierarchical order of fossil energy use categories implemented in CH-FARMIS

Biodiversity

The biodiversity impact scores were linked directly to crop activities and then rescaled within FARMIS to allow an assessment of sector-representative farm groups.

The SALCA biodiversity LCIA method was developed by Jeanneret *et al.* (2006). The method enables the integration of biodiversity in terms of organismal diversity for 11 groups of species. Species groups are divided into species groups with high ecological requirements, *i.e.* stenotopic species, or red-list species (amphibians, snails, grasshoppers, butterflies, spiders and beetles), and other indicator species (arable flora, grassland flora, birds, small mammals and wild bees). Jeanneret *et al.* (2006) did not consider soil organisms for this method. Furthermore, cross-relationships between species were not taken into account for single species. For instance, the positive effect of a greater abundance of arable weeds was not considered for the herbivore indicator species groups.

Jeanneret *et al.* (2008) calculated the biodiversity indicator for most Swiss crops according to the following procedure:

First, the impacts of each farm-management options (fertilisation, tillage, etc.) were rated by experts for specific indicator species on a scale of 0 to 5 (Table 21). In total, more than 3,000 management options were included in the activities modelled.

Table 21 Ratings for the impact of management options on indicator species groups

Score	Description
0	The species group is unaffected because it does not occur in the considered agricultural habitat.
1	The option leads to a severe impoverishment of species diversity within the species group considered and renders impossible the occurrence of stenotopic species and red list species.
2	The option leads to a slight impoverishment of species diversity within the species group considered and renders impossible the occurrence of stenotopic species and red list species.
3	The option has no direct effect on the species group considered.
4	The option leads to a slight increase in species diversity within the species group considered
5	The option promotes species diversity within the species group considered and makes possible the occurrence of stenotopic species and red list species.

Source: Jeanneret *et al.* (2008)

In a second step, a) habitats (e.g. extensive meadows) were weighted according to their quality for each indicator species group with a score of 1 to 10 ($C_{habitat}$) and b) management options were weighted according to their relative importance for a given habitat for each indicator species group ($C_{management}$). The coefficients were multiplied to produce a total score (S), as shown in Equation 59. In order to calculate a total biodiversity score (S) across all species groups, Jeanneret *et al.* (2008) took a mean value, weighted according to the importance of the species within the food chain. S is a dimensionless score ranging between 0 and 50. In the case of crops for which no biodiversity score was available from Nemecek *et al.* (2005), values were calculated for this study by the author using the SALCA-BD software tool (ART, 2007).

$$S = R\,\frac{(C_{management} + C_{habitat})}{2} \qquad (59)$$

where:

S	= total score of a management option for a species group
R	= rating for management options for a species group
$C_{management}$	= weight of the management option for a species group
$C_{habitat}$	= weight of the habitat for a species group

However, since both extreme scores cannot be achieved by many indicator species groups, the scores for each indicator species group were rescaled, in order to make them more easily interpretable (Equation 60). The upper benchmark value (S_{MAX}) was the highest score achieved over all habitats, *i.e.* the score which results from always opting for the most beneficial management option in the most beneficial habitat for a certain species group. The lower benchmark (S_{MIN}) was defined as the score which results from opting for the worst management option in the worst habitat (maximum and minimum scores can be found in Table 72, Annex B). The resulting indicator is denoted as S_REL.

$$S\_REL_{iks} = \frac{S_{iks} - S_{MIN_{iks}}}{S_{MAX_{iks}} - S_{MIN_{iks}}} \quad \forall i,k,s \tag{60}$$

The scope of index h can be defined depending on the research question, for example it can be restricted to biodiversity on arable land, grassland, or a more specific type of crop. S_{MAX} and S_{MIN} scores were derived using the SALCA-BD-tool (ART, 2007).

For this thesis, the total habitat quality was assessed, which means that h covers all crops. Finally, the average biodiversity score (B_REL) over all (selected) habitats can be calculated according to Equation 61, where X_h is the area cultivated under a specific crop.

$$B\_REL_{s} = \frac{\sum_{k} S\_REL_{hs} X_{h}}{\sum_{h} X_{h}} \quad \forall s \tag{61}$$

Eutrophication with nitrogen and phosphorus

Like the fossil energy use indicator, eutrophication impact assessment data were linked to FARMIS via the model's activities. Because eutrophication is an environmental impact category with regional consequences, as opposed to the impact indicator energy use, only the eutrophication-related emissions occurring on the farm were accounted for. Therefore, contrary to the general system boundaries and the approach for modelling energy use, eutrophication imported by purchasing feedstuffs was not accounted for. The consequences for the choice of these system boundaries for eutrophication will be discussed when interpreting the results of this study.

Eutrophication data stem from SALCA and are grouped into nitrogen eutrophication (including the sub-groups nitrate (NO_3), ammonia (CH_4) and other substances causing N eutrophication[61]), as published in Nemecek *et al.* (2005).

SALCA nitrate eutrophication was calculated according to a nitrate model developed by Richner *et al.* (2006). Ammonia emissions were taken from Nemecek (2003), who used Menzi *et al.* (1997) and Walther *et al.* (2001). Phosphorus eutrophication was calculated by applying a phosphorus model developed by Prasuhn (2006).

6.3.8 Public expenditure on agri-environmental policies and farming systems

In the model, both total public expenditure on direct payments and total policy-related transaction costs are calculated as two separate parameters.

Total public expenditure (PE_{TOTAL}) on direct payments is calculated by adding up the payments to the beneficiaries (PC) (Equation 62). Furthermore, variable as well as fixed transaction costs at cantonal and national level are added (TC_{VAR} and TC_{FIX}), while farm-level transaction costs are not considered, as they are meant to be compensated already by the direct payments.

$$PE_{TOTAL} = \sum_{n}\sum_{i}\sum_{k} (PC_{nik} + TC_{VAR\,nik}) + TC_{FIX} \qquad \textbf{(62)}$$

where:

n = index for farm group
i = index for production activities
k = index for intensity level
PE_{TOTAL} = total public expenditure on a policy
PC = costs for payments to beneficiaries (farmers)
TC_{VAR} = variable public policy-related transaction costs
TC_{FIX} = fixed public policy-related transaction costs

[61] Covering mainly nitrogen oxides (NOx) and nitrous oxide (N_2O)

The total transaction cost (TC_{TOTAL}) of a policy is estimated in Equation 63. As illustrated in Section 6.1, assessing the total transaction cost as a separate indicator is important for policy analysis because policies entailing lower farm-level transaction costs may not require as high payment rates to compensate farmers for their additional workload. As for the calculation of total public expenditure, all cost components except the fixed transaction costs are modelled specific to the farm group, in order to be able to report per farm group and agri-environmental policy. The additional consideration of total transaction costs (Equation 63) is of particular interest with regard to the differentiation between organic and conventional farms as, according to the literature a decline in public PRTC can be expected for organic farms (Tiemann *et al.*, 2005).

$$TC_{TOTAL} = \sum_{n}\sum_{i}\sum_{k} (TC_{FARM_{nik}} + TC_{VAR\,nik}) + TC_{FIX} \qquad \textbf{(63)}$$

where:

n = index for farm group
i = index for production activities
k = index for intensity level
TC_{TOTAL} = total policy-related transaction costs
TC_{VAR} = variable public policy-related transaction costs
TC_{FIX} = fixed public policy-related transaction costs
TC_{FARM} = transaction costs at farm level

The payments to the beneficiaries were obtained from FADN and public expenditure statistics. Transaction cost data were drawn from recent Swiss and international studies (Buchli and Flury, 2005; Mann, 2003a). While Buchli and Flury (2005) calculated transaction costs for common and ecological direct payments, without differentiating between different ECA measures, Mann (2003a) focussed on agri-environmental payments and calculated separate values for single measures.

Finally, transfer efficiency was determined by taking the ratio between total transaction costs and total public expenditure.

6.3.9 Conceptual levels of the cost-effectiveness evaluation

As shown in the previous sections, the main determinants of the cost-effectiveness of agri-environmental payments were derived by applying three extensions to FARMIS. These extensions were used for calculating the cost-effectiveness of organic farming as a farming system. The farming system as a whole was considered rather than calculating only the cost-effectiveness of the organic farming area support payments (OFASP) because:

a) differences in direct payment receipt between organic and conventional farms relate to all policy measures. OFASP accounts for on average only 7.8 % of the total direct payments received by organic farms, whereas the total difference in direct payments is approximately 27 % on average;

b) conversion to and from organic agriculture cannot be modelled using economic models alone. There are currently not sufficient empirical data for modelling the 'uptake' of OFASP, (see Section 6.1.1);

c) the influence of OFASP payments on conversion and re-conversion decision making is a largely unresolved question (Grey *et al.*, 2003);

Consequently, sole consideration of OFASP is of limited relevance in Switzerland and would require strong assumptions on conversion to and from organic agriculture. For this reason, it was decided to compare the cost-effectiveness of organic farms as a farming system on the one hand with the cost-effectiveness of individual agri-environmental measures on the other.

Figure 19 depicts the two different evaluation designs chosen for the evaluation a) the cost-effectiveness of organic farms and b) the cost-effectiveness of agri-environmental policies. The cost-effectiveness of organic farms is derived by comparing organic farms with their conventional counterparts in the base year situation. For the evaluation of cost-effectiveness of agri-environmental agri-environmental policies, the following procedure is applied:

First, four policy scenarios are defined: Policy Scenario A assumes the abolition of extenso payments, Scenario B assumes the abolition of payments for less intensive meadows, while Scenario C assumes the abolition of payments for extensive meadows. Finally, Scenario D assumes the abolition of all three above-mentioned policies.

Hence for each scenario, the environmental effects and costs (*i.e.* public expenditure) of the abolition of the policy measures is modelled. Thus empirical data on the 'treatment' (base year) and modelled data on the 'counterfactuals' for each policy (policy scenarios) were obtained.

Next, the difference in environmental effects and public expenditure between each policy scenario and the base line is interpreted as the additionality of the respective policy measure. Unlike econometric approaches, which would estimate the 'counterfactual' statistically (as described by e.g. Caliendo and Hujer, 2006; Frondel and Schmidt, 2005; Henning and Michalek, 2008), this approach models the counterfactual as a hypothetical experiment (Berger, 2000; Bertelsmeier, 2005).

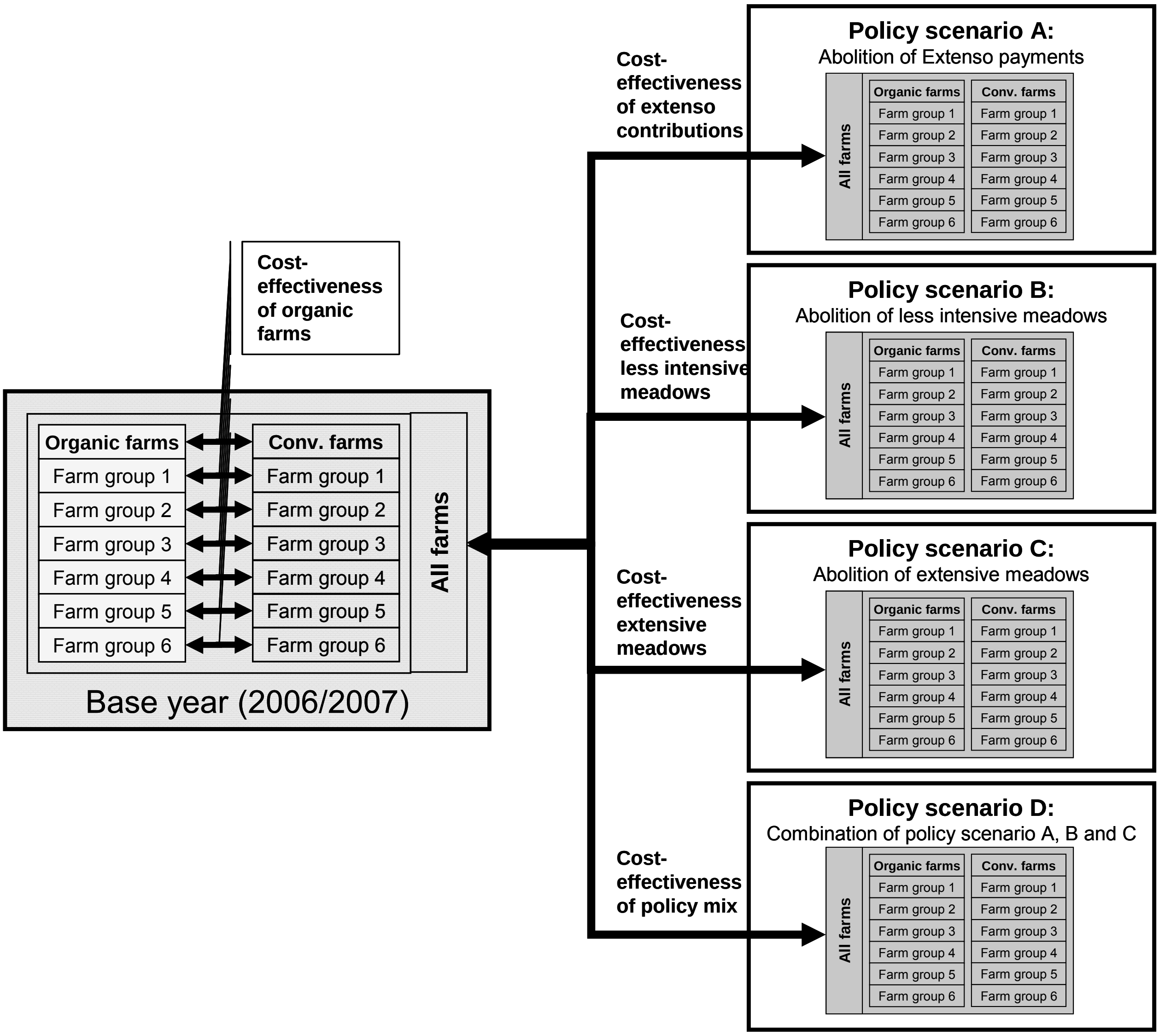


Figure 19 Evaluation design for farming system and policy measure comparison

In the following paragraphs, the procedures for both types of evaluation are illustrated.

Cost-effectiveness of organic farming

Equation 15 (see Section 6.1.4) is the basis for evaluating the farming system 'organic farming'. In order to derive a value for cost-effectiveness (CE_i), the relative environmental effects (RE_{ij}) and the absolute difference in average public expenditure (C_i) have to be determined. Both parameters are obtained by comparing organic with conventional farm groups in the base year. Either all farms of both farming systems are compared or specific farm types or regions, in order to diminish structural differences between organic farm groups and their conventional counterparts[62].

The RE_{ij} are expressed as hectare averages in relative terms (%) in order to avoid upscaling problems and to assure consistency between the environmental indicators (as discussed in Section 6.1.4). It is calculated as in Equation 64, where IND is the average state of the respective environmental impact indicator per ha in the farming system.

$$RE_{ij} = \frac{(IND_{ORG_{ij}} - IND_{CON_{ij}})}{IND_{CON_{ij}}} \quad \forall i, j \qquad \textbf{(64)}$$

C_i is also expressed as a hectare average. The parameter was derived by subtracting the total public expenditure per ha on conventional farms (PE_{CON}) from the total public expenditure per ha on organic farms (PE_{ORG}) (Equation 65). PE_{CON} and PE_{ORG} are obtained equivalently to PE_{TOTAL} (Equation 62) with index 'n' being limited to organic or conventional farms respectively.

$$C_i = \frac{PE_{ORG_i}}{UAA_{ORG_i}} - \frac{PE_{CON_i}}{UAA_{CON_i}} \quad \forall i \qquad \textbf{(65)}$$

Due to this evaluation design, cost-effectiveness refers to organic farms in Switzerland specifically, as a result of the specific geographic and policy environment rather than to 'organic farming' in general. Thus, results cannot be transferred to other policy settings or geographical locations. For linguistic reasons, henceforth 'organic farming' and 'the organic farms' are henceforth used synonymously when referring to the object of the cost-effectiveness evaluation.

[62] The question as to whether the conventional counterparts can be interpreted as the counterfactual of the organic farm groups is discussed in Section 7.1.

Cost-effectiveness of targeted policy measures

Contrary to the derivation of cost-effectiveness of organic farming, the cost-effectiveness of policy measures is derived by comparing the cost-effectiveness of all farms in the policy scenarios with all farms in the base year. In the policy scenarios, the payments for policy measures are set to 0 CHF/ha (Figure 19). Thus, a hypothetical situation is modelled, in which the payments which are under evaluation are abolished. In other words, empirical data on the 'treatment' and modelled data on the 'counterfactuals' (base year) are obtained.

Accordingly, the difference between the reference scenario and each policy scenario without the payment is interpreted as the additionality of the respective policy measure, as the model shows how farmers would respond, if the payment was not disbursed. The additionality consists of both a direct and an indirect component. The direct component relates directly to the policy uptake induced by the payment, while the indirect component refers to other responses from the farm groups, such as changes in stocking density.

Because the model calculations have a hypothetical character, no time dimension is defined for the model analyses. For this reason, no assumptions were made regarding structural change, conversion to and from organic agriculture, or price developments. RE_{ij} and C_i of the various policies are derived according to Equations 66 and 67.

$$RE_{ij} = \frac{(IND_{BASE_{ij}} - IND_{SCEN_{ij}})}{IND_{SCEN_{ij}}} \quad \forall i, j \tag{66}$$

$$C_i = \frac{PE_{BASE_i}}{UAA_{BASE_i}} - \frac{PE_{SCEN_i}}{UAA_{SCEN_i}} \quad \forall i \tag{67}$$

Comparison of cost-effectiveness

Using the values derived for relative effects (RE_{ij}) and costs (C_i), cost-effectiveness (CE_{ij}), and abatement (ABC_{ij}) or provision costs (PRC_{ij}) can be calculated for both farming systems and targeted policy measures using Equations 15, 16, and 17. Finally, the cost-effectiveness indicators of organic farming can be compared with the cost-effectiveness indicators of the agri-environmental policies.

6.3.10 Model validation

The model was validated concerning the main determinants of cost-effectiveness (policy uptake, environmental effects and public expenditure). However, a model validation using empirical data is feasible only to a limited extent.

Policy uptake and general model responses

A general validation of the CH-FARMIS model and its response was carried out before the scenarios were calculated. Furthermore, the model can be regarded generally as validated, since the general FARMIS model has been in use since 1998 and the Swiss version since 2006. Validation procedures for the Swiss FARMIS version have been described by Sanders (2007). As shown by Sanders (2007), the model performed well in several validation procedures. However, modest deviations from sector statistics and FADN data cannot be excluded due to the following reasons:

- FARMIS was calibrated on an average derived from two different years (2006/2007)
- FADN farms had been excluded from the FARMIS sample because only identical farms, *i.e.* farms within the FADN sample of both years were included in the analysis
- Unlike the FADN statistics, FARMIS uses improved farm-specific aggregation factors, which were designed to calibrate exactly to the base year situation.

The same limitations are given for a validation of the Röhm-Dabbert approach (RDA). However, the base year calibration and calculations of the reference scenario showed an almost perfect reproduction of the base year, including policy uptake levels. Further sensitivity analyses were conducted in order to test different specifications of the RDA elasticity coefficient (results are presented in Section 7.5.2).

Environmental effects

A formal model validation of environmental effects could not be conducted, since there were no reliable data available for environmental impact indicators employed. However, results regarding the environmental impacts of the farming systems were checked for plausibility at several times during the course of the thesis. Unusual results were checked with particular care. Additionally, plausibility checks were conducted with the assistance of a group of experts supporting the work on this thesis. This group consisted most prominently of the experts who modelled the LCA data, however, the group also included individual specialists on each impact category[63]. In addition, individual specialists were asked for feedback at several stages of the research. Finally, the results generated were compared with the existing body of literature.

The validation procedure revealed that the results were generally plausible. Deviations in absolute numbers (where available) can be attributed to differing system boundaries and assumptions due to different foci of research. Most controversial was the validation of biodiversity results, as the input data was subject to a scientific dispute stemming from conflicting results from different publications on the biodiversity impacts of organic farming systems. Standard SALCA biodiversity data were used, without adapting them, even if experts had conflicting opinions. However, the methodological limitations of these data need to be emphasised and the differences between organic and conventional activities should be regarded as conservative estimates.

Public expenditure

As described for the general model responses, a validation procedure over several years lacks data which can be precisely compared, because sector statistics differ for the reasons mentioned above. Therefore, only basic checks were made in order to avoid a general over or underestimation of public expenditure. As presented in Table 22, utilised agricultural area was 5.6 % lower in the FARMIS reference scenario, because not all strata could be represented due to a lack of FADN farms in some strata. Total public expenditure was 1 % higher than a hypothetical mean taken from the 2006 and 2007 data. This showed a slight overestimation of 7.0 % of per-ha public expenditure in the reference scenario within FARMIS (Table 22).

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[63] The group consisted of the following experts from Agroscope Reckenholz-Tänikon (ART): Gerard Gaillard, Thomas Nemecek, Daniel Baumgartner and Michael Winzeler regarding the general SALCA approach and energy use in particular; Philippe Jeanneret regarding biodiversity and habitat quality; Ruth Freiermuth and Walter Richner regarding eutrophication with nitrogen and phosphorus.

Thus the deviation of the model in terms of its parameters is acceptable overall, allowing a model analysis of the cost-effectiveness of organic farming compared to targeted agri-environmental payments.

Table 22 Deviations of public expenditure at sector level between FARMIS results and official statistics in 2006 and 2007

Parameter	Unit	2006	2007	Mean 06/07	FARMIS model results	Deviation FARMIS - mean 06/07
Utilised agricultural area	1,000 ha	1,065	1,060	1,063	1,003	-5.60%
Total public expenditure on direct payments	kCHF	2,499,572	2,575,039	2,537,306	2,562,540	0.99%
Average public expenditure per ha UAA	CHF	2,347	2,429	2,388	2,555	6.98%

Source: own calculation based on FADN and BLW (2008)

6.4 Summary and conclusions

A conceptual model has been developed to show the cost-effectiveness of agri-environmental policies at sector level. Three determinants of cost-effectiveness were identified: policy uptake, environmental effects and public expenditure. The graphical model shows the interdependencies between payment rates, public expenditure, environmental effects and cost-effectiveness. The corresponding algebraic model builds the basis for comparing the cost-effectiveness of different policies with each other.

A review of existing sector-wide PMP modelling approaches showed that some of the cost-effectiveness determinants are addressed by each model. However, it also revealed that none of the existing models is suited to evaluate the cost-effectiveness of agri-environmental policy, particularly for organic farming, as the main determinants cannot be modelled fully; organic farming is explicitly modelled in only two of the PMP models reviewed. Conversion to and from organic agriculture was not covered by any of the reviewed models.

On the basis of the preceding conclusions on cost-effectiveness, an approach was outlined which specifically addresses the research questions of this thesis. The approach is based on and extends the PMP model FARMIS using three additional modules. Each module addresses one of the key determinants of cost-effectiveness.

Policy uptake is addressed by the Röhm-Dabbert approach, which allows for an improved modelling of agri-environmental policy uptake. Intensity levels of the same crop are conceptualised as similar activities, which can be exchanged with each other more easily than others. Formally, this is achieved by splitting the PMP term into one part which is intensity-dependent and another part which is dependent on the total level of the activity. Conversion to and from organic agriculture was not modelled explicitly. **Environmental effects** are addressed by linking life cycle assessment data to FARMIS. The impact categories fossil energy use, biodiversity and eutrophication with nitrogen and phosphorus are covered by using representative Swiss agricultural life cycle impact assessment data of farming systems. **Public expenditure** is covered by the modelled payment rates and the public policy-related transaction costs of the direct payments.

This chapter has shown how the cost-effectiveness of organic farming as a farming system can be compared to the cost-effectiveness of targeted agri-environmental policy using the extended PMP model, described above. The cost-effectiveness of organic farming is derived from comparing sector-representative organic farm groups with their conventional counterparts in terms of environmental performance and public expenditure. The cost-effectiveness of targeted agri-environmental payments is determined by comparing policy scenarios in which the different policies are abolished with a reference situation. The difference in environmental performance and public expenditure on the farm groups is understood as the additionality of the policy. The ratios between costs and effects for organic farming and targeted policy measures can then be compared with each other.

By using the procedure presented above for evaluating organic farming by base year comparisons of farm groups, it is possible to avoid modelling conversion to and from organic agriculture. However, using different evaluation procedures for organic farming on the one hand and targeted environmental payments on the other may result in a biased evaluation. For this reason, the results need to be interpreted with care, while potential biases will be discussed extensively.

Finally, the procedure and results of the model validation were outlined. The results of the validation show that the model is generally valid for use, although reference values aimed at determining exactly the performance of the model are lacking, particularly for the environmental indicators. Therefore, sensitivity analyses are proposed with regard to the key determinants of cost-effectiveness.

7 Assessment of the cost-effectiveness of organic farms in providing environmental services

This chapter contains, first, a description of the farm groups included in the analysis (Section 7.1). Second, farm-group comparisons in the base year situation are used for analysing the cost-effectiveness of organic farming (Section 7.2). Third, results concerning the cost-effectiveness of single and combined agri-environmental policy measures are determined (Section 7.3). Section 7.4 compares the cost-effectiveness of organic farming as a farming system with that of the other agri-environmental policy measures. The results of the sensitivity analyses are presented in Section 7.5. Finally, Section 7.6 summarises the main results of the chapter.

7.1 Description of farm groups

This section first explains which organic and conventional farm groups are compared with each other in the model analysis and then goes on to present the distribution of farms in the different groups followed by the choice of strata for a) the farm group comparison and b) the scenario analysis.

Comparison of farm groups

As the foregoing descriptive statistical analysis of both the farm structure survey data and the FADN data has shown[64], there are substantial differences in farm structure parameters between organic and conventional farms. These differences are caused primarily by a varied distribution of farms among the regions and a varied distribution of farm types. There are no substantial structural differences between organic and conventional farm types and regions regarding farm size (see Boxplot graphs in Figure 36, Annex C). Therefore, structural differences are classified as follows:

[64] For the sake of brevity, the results of these analyses are not presented here. Structural differences are discussed for the modelled base year in Section 7.2 in any case. The differences between the modelled base year and the statistical data are minimal.

a. **Regional distribution:** Organic farms are concentrated more in the mountain regions and less in the lowlands compared to conventional farms.

b. **Farm type distribution:** Specific farm types are more or less prevalent among organic farms than among conventional farms. For instance, there are almost no specialised organic arable farms or pig and poultry farms.

c. **Differences in farm structure:** These encompass different levels of labour or physical input intensity and productivity, differences in stocking density, cropping patterns, and policy uptake occurring between organic and conventional farms within a certain farm type or region.

However, it is difficult to prove the causality behind these differences statistically. The reasons for the differences in farm structure may be attributed to either:

1. **Farming-system inherent differences**: As it is known from the literature (Köpke *et al.*, 1997) and evident from organic standards and regulations, the conversion of a farm to organic agriculture entails changes in farm management. These changes include those aspects delineated in Table 13 (page 73).

2. **Self-selection bias** refers to the fact that farms with specific features tend to be more likely to convert to organic agriculture. In particular, farms that already have extensive management practices tend to convert to organic agriculture, even if they are of the same farm type and within the same region. This is because these farms do not have to carry out any major changes on the farm, *i.e.* the costs of adaptation are relatively low. The core of the self-selection bias lies in what is described as the 'fundamental evaluation problem'(Frondel and Schmidt, 2005), meaning that one cannot observe the counterfactual situation, *i.e.* how farms would have developed without taking up the policy (or if the policy had not been available). This implies that 'the effect of the treatment on the treated' is uncertain (Henning and Michalek, 2008). Observational approaches range from before-after comparisons, cross-section, difference-in-difference through to matching estimators (Frondel and Schmidt 2005; Caliendo and Hujer 2006).

Due to the above mentioned 'fundamental evaluation problem' (Frondel and Schmidt, 2005), which makes it difficult to identify the causal relations between conversion and structural differences unambiguously, it is necessary to consider the structural differences within the evaluation. This is done here by making the following assumptions regarding the differences between organic and conventional farms:

a. **Differences in the regional distribution of farms,** *i.e.* farms in specific regions being more likely to convert than others are an indirect cause of variations in the farm-type distribution.

b. **Differences in the distribution of farm-types** are caused by both differences inherent to the respective farming system and the self-selection bias. For instance, the share of arable farms in the total number of organic farms is lower than the share of conventional arable farms in total conventional farms. There are two potential causalities to which this fact could be attributed: First, specialised arable farms are less likely to convert to organic agriculture than mixed farms (self-selection). Second, organic standards could induce a change in farm management and thus turn a formerly arable farm into a mixed farm when converting to organic agriculture (system-inherent).

c. **Differences in farm structure** (other than regional and farm type-based distribution) are farming-system inherent, *i.e.* a direct or indirect cause of the conversion process.

Thus by grouping the farms according to the criteria of farm type and region, structural differences can be minimised, permitting a sound comparison of the farm types. The following comparisons were conducted:

- **Comparison of all farms**, to reveal differences between the farming systems irrespective of the regional and farm type-based differences. This comparison is used for the cost-effectiveness comparison (described in detail in Section 6.3.9), as it comprises the total public expenditure on organic farms in addition to the public expenditure on conventional farms and the additional average relative environmental effects caused by organic farms compared to conventional farms. Thus this comparison acknowledges that different distribution of organic farms according to farm type (e.g. the fact that there are almost no specialised organic pig and poultry farms) is a feature inherent to the farming system rather than being due to a self-selection bias.

- **Comparisons by region**, to minimise the regional self-selection bias, *i.e.* the fact that farms in specific regions are more likely to convert to organic agriculture. Thus similarly to the comparison of all farms, this comparison acknowledges the different distribution of organic farms according to farm type as characteristic inherent to the farming system.

- **Comparisons by farm types**, to minimise the farm group-related self-selection bias, *i.e.* the fact that specific farm types are more likely to convert to organic agriculture.

Composition of farm groups

Table 23 shows the total number of farms represented in the model, the number of FADN farms taken as a sample for these farms, and the number of farm groups which were optimised separately.

For the comparison of all farms, 67 different conventional farm groups and 27 organic farm groups were optimised. These farm groups were generated using 2,451 (conventional) and 342 (organic) FADN farms, which were representative of 44,838 (conventional) and 4,888 (organic) actual farms.

Concerning the regional farm groups (totals across all farm types), the majority of all farms can be found in the lowlands (23,442), while approximately 14,000 farms can be found in both hill and mountain areas. Since all regional farm groups, including organic farms, were formed using at least 50 FADN farms, the validity of the model can be regarded as good. The minimum number of FADN farms was 75 for the stratum 'organic farms in the lowlands'.

With respect to the farm types (totals across all regions), the representativeness is lower because several farm types do not contain enough FADN farms to be used for modelling. Both for 'arable farms' and 'pig and poultry farms' no organic FADN farms were found at all[65]. Furthermore, both organic 'speciality crop farms' and 'other grassland farms' consisted of less than 50 FADN farms. Due to potential biases regarding the input-output coefficients, these two farm types were also excluded from the scenario analysis (Section 7.3). Base year results for the above mentioned farm groups are, however, presented in Section 7.2.

[65] Due to using only identical farms from both 2006 and 2007 FADN samples

Table 23 Farm groups and strata for the base year comparison and the scenario analysis (based on identical farms in 2006 and 2007 FADN sample)

Farm type	Number of farms and farm groups	Lowlands		Hills		Mountains		Total		Farm-type represent-ativeness***
		CON	ORG	CON	ORG	CON	ORG	CON	ORG	
Arable crop farms	Farm groups	1		1				2		99.3%
	FADN farms	100		6				106		
	Farms represented	3,440		117				3,557		
Speciality crop farms	Farm groups	1	**1**					1	**1**	82.3%
	FADN farms	67	**5**					67	**5**	
	Farms represented	2,959	**193**					2,959	**193**	
Dairy farms	Farm groups	3	**1**	6	**2**	6	**3**	15	**6**	97.8%
	FADN farms	205	**25**	385	**54**	299	**113**	889	**192**	
	Farms represented	2,601	**240**	5,548	**573**	5,323	**1,275**	13,473	**2,088**	
Suckler cow farms	Farm groups	1	**1**	2	**2**	3	**3**	6	**6**	100.0%
	FADN farms	28	**4**	35	**11**	27	**38**	90	**53**	
	Farms represented	490	**91**	735	**247**	609	**732**	1,834	**1,070**	
Other grassland farms*	Farm groups	1		2	**1**	6	**3**	9	**4**	95.8%
	FADN farms	12		11	**4**	131	**19**	154	**23**	
	Farms represented	921		1,417	**89**	3,404	**715**	5,742	**804**	
Pig and poultry farms	Farm groups	2		2		1		5		97.5%
	FADN farms	35		26		8		69		
	Farms represented	760		586		174		1,520		
Mixed farms**	Farm groups	12	**4**	13	**4**	4	**2**	29	**10**	98.3%
	FADN farms	762	**41**	273	**18**	41	**10**	1,076	**69**	
	Farms represented	10,933	**486**	3,975	**164**	845	**82**	15,753	**733**	
Total farms	Farm groups	21	**7**	26	**9**	20	**11**	67	**27**	96.8%
	FADN farms	1,209	**75**	736	**87**	506	**180**	2,451	**342**	
	Farms represented	22,104	**1,010**	12,379	**1,073**	10,355	**2,804**	44,838	**4,888**	
	Farms in the population	**23,442**		**14,041**		**13,903**		**51,386**		
	Representativeness***	**98.6%**		**95.8%**		**94.7%**		**96.8%**		

Source: own calculation based on FADN and FSS data

* merged farm group of other cattle farms and horses, sheep and goat farms

** merged farm group from combinded dairy/arable crops, combinded suckler cows, combinded pigs/poultry and combined other farms

*** share of farms represented by the FARMIS sample of the farms represented by the total FADN sample

strata taken into account separately for the farming-system comparison but not for the cost-effectiness analysis (Section 7.2)

strata taken into account separately for both the farming-system comparison and for the cost-effectiveness analysis (Section 7.2)

Only for the farm types 'dairy farms', 'suckler cow farms', and 'mixed farms', were sufficient FADN farms available for the organic strata. 'Dairy farms' are representative of 13,473 conventional farms and 2,088 organic farms, 'suckler cow farms' are representative of 1,834 conventional and 1,070 organic farms, while ‘mixed farms’ are representative of 15,753 conventional and 733 organic farms.

7.2 Cost-effectiveness of organic farms in the base year

As described in Section 6.3.9, the abatement and provision costs of organic farming were obtained by comparing organic farm groups with their conventional counterparts in the baseline scenario. The comparison of farm groups is structured as follows:

First, structural differences regarding crops, livestock and labour (Section 7.2.1) as well as regarding the financial performance (Section 7.2.2) are described. Second, the results regarding the determinants of cost-effectiveness policy uptake (Section 7.2.3), environmental indicators (energy use (Section 7.2.4), habitat quality (Section 7.2.5), and eutrophication (Section 7.2.6)) and public expenditure (Section 7.2.7) are illustrated. Finally, the results of these sections are combined to produce an estimation of the cost-effectiveness of organic farming as an agri-environmental policy option in Switzerland (Section 7.2.8).

Each of the sections, first, conducts a regional comparison. Subsequently, the farming systems are compared by farm type. Each of both comparisons refers first to general differences between the respective groups (regions or farm types) in conventional farm groups, before describing differences between the farming systems (conventional/organic).

7.2.1 Farm structure

This section examines, first, the farm structure in terms of crop production. Second, animal husbandry and labour requirements are shown.

Crop production

Comparison by region

Table 24 presents the average utilised agricultural area (UAA) and the share of crops by region. The average UAA of organic and conventional farm groups shows minor variations between the regions, ranging between 19 and 21 hectares per farm. As a function of altitude, the share of open arable land in total UAA decreases markedly from 47 % in the lowlands to

1 % in the mountain areas, whereas the share of grassland rises from 48 % in the lowlands to 97 % in the mountain areas. The share of permanent crops in total UAA levels between approximately 1 % (hills) and 5 % (lowlands). Most significant arable crops in terms of cultivated area are bread cereals, fodder cereals, maize and oilseeds. Organic farm groups, which generally have a lower share of arable crops, grow relatively more bread cereals than conventional farms and fewer other arable crops, reflecting both market demand and technical considerations. Instead, organic farms have leys in their rotation for soil fertility and nutrient management reasons. Thus, leys cover about 22 % of the UAA on organic farms as opposed to 15 % on conventional farms in the lowlands.

Table 24 Crop shares on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountains		Total	
		CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	**ORG**
UAA	**ha/farm**	21.07	**18.97**	19.06	**18.67**	19.70	**20.74**	20.20	**19.92**
Share open arable land	**%**	48.0	**32.1**	17.1	**6.6**	1.2	**0.2**	29.4	**7.8**
Share bread cereals	**%**	14.7	**15.1**	4.4	**2.9**	0.1	**0.0**	8.7	**3.6**
Share fodder cereals	**%**	8.4	**2.6**	6.1	**1.3**	0.7	**0.0**	6.0	**0.8**
Share maize	**%**	9.7	**5.6**	4.2	**1.2**	0.4	**0.1**	6.2	**1.4**
Share root crops	**%**	7.5	**2.2**	0.9	**0.8**	0.1	**0.0**	4.1	**0.6**
Share pulses	**%**	1.0	**0.4**	0.2	**0.1**	-	**-**	0.6	**0.1**
Share oilseeds	**%**	4.7	**0.2**	0.9	**0.1**	-	**-**	2.7	**0.1**
Share fallow	**%**	0.6	**0.4**	0.1	**-**	-	**-**	0.3	**0.1**
Share other arable land	**%**	1.3	**5.5**	0.1	**0.3**	-	**0.0**	0.7	**1.2**
Share total grasland	**%**	47.3	**65.4**	81.7	**92.6**	96.8	**98.6**	67.4	**90.8**
Share ley	**%**	15.3	**21.8**	17.5	**13.3**	4.0	**1.2**	13.3	**7.7**
Share permanent meadows	**%**	28.5	**37.3**	54.8	**68.0**	76.0	**84.4**	46.0	**71.8**
Share permanent pastures	**%**	3.5	**6.3**	9.5	**11.2**	16.9	**13.0**	8.1	**11.3**
Share permanent crops	**%**	4.7	**2.5**	1.3	**0.8**	2.0	**1.2**	3.2	**1.4**

CON = Conventional farms; ORG = Organic farms Source: own calculations based on Swiss FADN

Grassland is dominated by meadows, which generally account for more than 50 % of total grassland in all farm groups. Due to lower shares of arable land, *i.e.* land included in crop rotation, the ley areas are less prevalent in the mountain areas. Conventional farms demonstrate a wider range in the share of pastures in total UAA, ranging between 3.5 % (lowlands) and 17 % (mountains), whereas the share on organic farms ranges from 6.3 % (lowlands) to 13 % (mountains). The shares of pastures in total UAA vary between organic and conventional farms depending on specific areas: Organic farms have lower shares in the mountain areas, while conventional farms have lower shares in the lowlands and hill areas.

Comparison by farm type

The comparison by farm type reveals significant differences in the average UAA (Table 25).

Table 25 Crop shares on conventional and organic farms by farm type (2006/07)

Indicator	Unit	Arable farms	Speciality crop farms		Dairy farms		Suckler cow farms		Other grassland farms		Pig / Poultry	Mixed farms	
		CON	CON	ORG	CON	ORG	CON	ORG	CON	ORG	CON	CON	ORG
UAA	ha/farm	24.90	13.18	**14.96**	19.27	**19.77**	20.04	**19.78**	16.14	**21.11**	17.68	22.99	**20.53**
Share open arable land	%	81.8	49.5	**58.4**	5.3	**2.5**	6.1	**2.1**	4.0	**0.2**	11.1	41.8	**29.3**
Share bread cereals	%	28.6	16.8	**21.7**	0.7	**1.0**	1.5	**1.2**	1.2	**-**	1.4	11.9	**14.8**
Share fodder cereals	%	10.6	3.5	**0.5**	2.0	**0.4**	1.5	**0.4**	1.3	**-**	3.8	10.0	**3.5**
Share maize	%	10.6	6.6	**6.1**	2.3	**0.8**	2.4	**0.3**	1.3	**-**	5.0	9.5	**5.4**
Share root crops	%	14.0	7.1	**4.7**	0.2	**0.2**	0.2	**0.1**	-	**0.1**	0.8	5.9	**2.5**
Share pulses	%	3.1	1.0	**1.3**	-	**0.1**	0.1	**0.1**	-	**-**	-	0.6	**0.3**
Share oilseeds	%	11.9	6.6	**-**	0.1	**-**	0.2	**-**	0.1	**-**	-	3.0	**0.2**
Share fallow	%	1.0	2.4	**1.7**	-	**-**	-	**-**	-	**-**	0.1	0.3	**0.1**
Share other arable land	%	2.0	5.5	**22.1**	0.1	**0.1**	0.0	**0.2**	-	**0.1**	0.1	0.7	**2.7**
Share total grasland	%	16.5	15.6	**33.0**	93.0	**95.9**	91.6	**97.0**	93.9	**99.6**	88.3	56.4	**69.4**
Share ley	%	5.4	3.5	**20.3**	9.9	**4.8**	13.6	**7.9**	5.1	**1.2**	18.3	20.5	**20.6**
Share permanent meadows	%	9.5	11.5	**12.7**	71.7	**80.6**	61.7	**76.3**	73.8	**83.9**	63.0	30.4	**38.9**
Share permanent pastures	%	1.6	0.6	**-**	11.5	**10.6**	16.3	**12.8**	15.0	**14.4**	7.1	5.4	**9.9**
Share permanent crops	%	1.6	34.9	**8.6**	1.6	**1.6**	2.3	**0.9**	2.2	**0.2**	0.5	1.9	**1.2**

CON = Conventional farms; ORG = Organic farms Source: own calculations based on Swiss FADN

Arable farms, for example, cultivate about 25 hectares, while speciality crop farms are smaller than 15 hectares on average. Substantial differences in farm size between organic farms and their conventional counterparts occur only on other grassland farms, where conventional farms have 16 ha and organic farms have 21 ha. Table 25 shows that structural differences, e.g. different shares of arable and grassland between organic and conventional farms, are less distinct in the farm-type comparison, using the Swiss farm typology grid (Table 71, Annex A) than in the regional comparison. However, conventional dairy, suckler cow, other grassland and mixed farms have 30 to 50 % higher shares of arable land than organic farms of the same type. Accordingly, grassland shares are higher on organic farms. For speciality crop farms, marked differences in crop shares are evident. Conventional speciality crop farms have about 35 % permanent crops, whereas organic ones have only 8.6 %. As an exception among farm types, conventional speciality crop farms cultivate less arable land than their organic counterparts.

Livestock units and labour

Comparison by region

The regional average number of livestock units (LU) on conventional farms is about 25, ranging from 22 in the mountain areas, through 25 in the lowlands, to approximately 28 LU in the hill regions (Table 26). The share of ruminants rises from 74 % in the lowlands to 94 % in the mountain areas. About 50 % of the livestock units on conventional farms are dairy cows, whereas approximately 10 % of cows are kept for beef production. As a function of altitude, the share of pig LU in total LU decreases markedly from about 19 % in the lowlands to 15 % in the hill area and 4 % in the mountain area. The same trend, although not as marked and at a lower absolute level, can be observed for poultry, starting from 6 % in the lowlands and falling to 4.5 % in the mountain areas.

The maximum total livestock density can be found in the hill regions averaging about 1.4 LU/ha. In the lowlands, which are relatively strongly dominated by arable farming, the livestock density is 1.2 LU/ha, while the most extensive farms are in the mountain areas with 1.1 LU/ha. The indicator 'average main fodder area per roughage-consuming livestock unit' (MFA/RLU) is highest in mountain areas (0.2 ha/LU) and lowest in the lowlands (0.1 ha/LU) due to the strong focus on arable farming.

Total annual working units (AWU) vary only slightly between regions, ranging from 1.6 AWU/farm in the mountain and hill areas to 1.7 AWU/farm in the lowlands. By contrast, the number of family working units (FWU) per farm rises gradually as a function of altitude from 1.17 FWU/farm in the lowlands to 1.25 FWU/farm in the hills and 1.33 FWU/farm in the mountain areas.

Organic farms in all regions have lower stocking densities than conventional farms. Among all livestock categories, except for beef, the livestock share is higher on conventional farms compared to organic farms. The stocking differences are most noticeable for pig and poultry stocking rates and the total number of LU relative to UAA. Two (mountain area) to four (lowlands and hills) times more grassland was available per roughage-consuming livestock unit (MFA/RLU). The generally lower intensity on organic farms is also indicated by the lower stocking rate.

Table 26 Livestock units and labour requirements on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountains		Total	
		CON	ORG	CON	ORG	CON	ORG	CON	ORG
Total livestock units (LU)	**LU/farm**	25.70	**19.73**	27.85	**22.95**	22.04	**19.41**	25.45	**20.25**
Ruminant LU	**%**	73.26	**90.44**	81.52	**94.76**	93.47	**98.22**	79.80	**95.79**
Dairy cow LU	**%**	48.99	**55.06**	52.63	**49.47**	53.99	**37.88**	51.09	**44.23**
Beef LU	**%**	9.55	**17.62**	7.45	**21.36**	6.44	**22.61**	8.29	**21.29**
Pig LU	**%**	19.73	**5.53**	15.01	**2.58**	4.22	**1.08**	15.20	**2.35**
Poultry LU	**%**	6.51	**3.97**	3.37	**2.09**	2.25	**0.64**	4.71	**1.67**
Other animal LU	**%**	0.43	**-**	0.03	**-**	0.05	**0.00**	0.23	**0.00**
Main forage area per ruminant LU	**ha/LU**	0.07	**0.40**	0.17	**0.71**	0.19	**0.42**	0.12	**0.48**
Stocking density	**LU/ha**	1.22	**1.04**	1.46	**1.23**	1.12	**0.94**	1.26	**1.02**
Average working units (AWU)	**AWU/farm**	1.69	**2.05**	1.55	**1.53**	1.57	**1.63**	1.62	**1.69**
Family working units (FWU)	**FWU/farm**	1.17	**1.18**	1.25	**1.24**	1.33	**1.36**	1.23	**1.30**

CON = Conventional farms; ORG = Organic farms   Source: own calculations based on Swiss FADN

There are no noteworthy differences in average family working units between organic and conventional farm groups. However, total working units are usually higher on organic farms. This indicates a higher dependency on hired labour.

Comparison by farm type

Comparing the farming systems by farm type (Table 27), the structural differences between the farming systems are levelled out to a larger extent than in the regional comparison. However, differences between the farm types are greater. The average number of livestock units on conventional farms ranges from 1.8 LU/farm on speciality crop farms to 65.2 LU/farm on pig and poultry farms. The share of ruminant livestock on conventional farms is mostly above 90 % except for pig and poultry farms (35 %), arable farms (73 %) and mixed farms (73 %). Specialised dairy farms have a share of 71 % of dairy livestock; suckler cow farms have a 73 to 75 % share of beef-related livestock.

The main forage area per roughage-consuming livestock unit remains in a similar range for all conventional farm types. By contrast, the number of livestock units per ha differs between farm type. Specialised arable and speciality crop farms keep 0.1 to 0.3 LU/ha, while pig and poultry farms keep about 3.7 LU/ha. Conventional dairy and mixed farms are in the same range, with about 1.4 LU, while suckler cow and other grassland farms are slightly less intensive, with 1.2 and 0.9 LU per ha.

Working units differ from farm type with arable, suckler cow and other grassland farms being most work-extensive. The number of family working units remains in a narrow range for each farm type.

The general trend of a lower stocking density on organic farms, compared to their conventional equivalent, is apparent for all farm types, except other grassland farms. While organic speciality crop farms do not keep animals at all, except for 0.03 LU of poultry, the share of ruminant livestock on other farm types is comparable to the respective conventional farm group. However, organic farms tend to have a higher share of beef production-related ruminants, while dairy cows are relatively less abundant. The differences in pig livestock shares mentioned above are prevalent in all farm groups. Regarding poultry, only the organic mixed farms possess slightly higher shares. However, due to the higher total livestock density, the absolute number remains higher on conventional farms.

Table 27 Livestock units and labour requirements on conventional and organic farms by farm type (2006/07)

Indicator	Unit	Arable farms	Speciality crop farms		Dairy farms		Suckler cow farms		Other grassland farms		Pig / Poultry	Mixed farms	
		CON	CON	ORG	CON	ORG	CON	ORG	CON	ORG	CON	CON	ORG
Total livestock units (LU)	**LU/farm**	7.82	1.83	**0.03**	26.67	**22.43**	23.01	**18.09**	14.31	**19.33**	65.22	33.32	**23.56**
Ruminant LU	**%**	72.84	83.59	**-**	96.36	**97.92**	95.03	**98.33**	97.39	**98.73**	35.04	73.27	**84.55**
Dairy cow LU	**%**	41.94	30.46	**-**	71.74	**71.08**	1.40	**2.01**	38.77	**18.18**	25.44	48.43	**42.15**
Beef LU	**%**	17.02	31.57	**-**	0.42	**0.81**	73.35	**74.99**	8.36	**16.22**	2.53	8.83	**21.25**
Pig LU	**%**	11.14	10.57	**-**	3.09	**1.50**	3.43	**1.49**	2.49	**1.02**	47.94	20.50	**6.82**
Poultry LU	**%**	9.50	5.82	**100.00**	0.49	**0.45**	0.17	**0.18**	0.12	**0.24**	17.02	6.09	**7.89**
Other animal LU	**%**	6.51	-	**-**	0.06	**0.01**	1.36	**-**	-	**-**	-	0.01	**-**
MFA/RLU	**ha/LU**	0.05	0.05	**-**	0.08	**0.25**	0.31	**0.57**	0.19	**0.56**	0.28	0.13	**1.06**
Stocking density	**LU/ha**	0.31	0.14	**0.00**	1.38	**1.13**	1.15	**0.91**	0.89	**0.92**	3.69	1.45	**1.15**
Average working units	**AWU/farm**	1.34	1.95	**3.44**	1.63	**1.62**	1.35	**1.39**	1.39	**1.80**	1.67	1.73	**1.75**
Family working units	**FWU/farm**	1.00	1.15	**1.09**	1.33	**1.30**	1.13	**1.16**	1.21	**1.57**	1.16	1.23	**1.25**

CON = Conventional farms; ORG = Organic farms
Source: own calculations based on Swiss FADN
MFA/RLU = Main forage area per roughage-consuming livestock
AWU = Average working units
FWU = Family working units

The stocking rate and MFA/RLU vary markedly between farm types. Arable farms and speciality crop farms have stocking rates below 0.5 LU/ha, while dairy, suckler cow, mixed and other grassland farms have stocking rates between 0.89 and 1.45 LU/ha. Maximum stocking rates are found on pig and poultry farms with 3.69 LU/ha. The MFA/RLU is more evenly distributed among the farm types. Arable and speciality crop farms have slightly less MFA/RLU than conventional dairy farms. Conventional mixed farms also have a low MFA related to the ruminant livestock units. Peak values among conventional farms can be found on suckler cow farms (0.31 ha MFA/RLU).

Organic farms of all farm types have substantially lower stocking rates than their conventional counterparts. For instance, organic mixed farms keep 1.15 LU/ha compared to 1.45 LU/ha on conventional farms. The differences in main fodder area per roughage-consuming livestock are even higher between organic farms and conventional farms. Particularly pronounced differences exist between organic and conventional mixed farms (conventional 0.13 LU/ha, organic 1.06 LU/ha).

The higher number of working units on organic farms is also reflected in each farm group except for dairy cow farms. Particularly striking is the difference in working units for speciality crop farms, which indicates the much higher labour requirements on organic farms, particularly with respect to hired labour.

The analysis of structural differences showed that both organic speciality crop farms and other grassland farms are structurally very different from their conventional counterparts. As described beforehand, these differences may be attributed partly to a low representation of these farm types in the FADN. Because the input-output data may be flawed due to the low sample size, these farm types are not analysed specifically in the modelling scenarios (Section 7.3). Moreover, pig and poultry farms as well as arable farms are excluded due to their low abundance among organic farms.

7.2.2 Financial performance

Conventional farms have an income of 115 kCHF on average over all regions (Table 28). Farm income per annual working unit (AWU) is on average 71 kCHF/AWU. Family farm income is 88 kCHF and 63 kCHF per family working unit (FWU). Organic farms have a slightly lower farm income (111 kCHF and 66 kCHF/AWU). Family farm income per farm is slightly higher (91 kCHF). Related to FWU, farm income is marginally lower (63 kCHF/FWU).

Comparison by region

In regional terms, all the above mentioned income parameters are smallest in the lowlands and greatest in mountain areas. As shown in Table 28, this difference is due to the gaps in production value, with 192 kCHF/farm in the lowlands, 136 kCHF/farm in the hill areas and 83 kCHF/farm in the mountain areas. The production value of both crops and livestock is lower in the mountain areas, whereas the regional differences in crop production-related value are greater. The lower production values are compensated for by general direct payments, while ecological direct payments decrease slightly with higher altitude. Further compensation for decreasing production value is achieved due to the lower total costs incurred by farms with higher elevation.

Differences in farm income between organic and conventional farms are marked. Farm incomes are generally greater for organic farms in all regions. However, related to labour input, only in the lowlands farm income is slightly lower on organic farms. Farm income on organic farms in the lowlands is 74 kCHF/AWU, while organic farms yield 81 kCHF/AWU. The family farm income related to labour input differs only marginally, with 69 kCHF/FWU on organic farms compared to 71 kCHF/FWU on conventional farms.

Both total costs and total revenues are almost equal on conventional farms and organic farms in the regions, with slightly lower values for both indicators on organic farms in most regions. However, the composition of total revenues differs between the farming systems. Organic farms tend to derive a larger share of their income from direct payments, while on conventional farms the production values are higher. While general direct payments are similar on conventional and organic farms, ecological direct payments are higher on organic farms in all regions, even when taking into account the lower values for organic farms in the lowlands. Production values of crop products are higher on organic farms, with 85 kCHF/farm in the lowlands compared with 65 kCHF/farm on conventional lowland farms. However, in hill and mountain regions the production values are only marginally higher. The higher values of crop production are overcompensated for by the 40 kCHF/farm (lowlands), 30 kCHF/farm (hills), and 20 kCHF/farm (mountains) lower values for livestock production.

Table 28 Income structure on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountains		Total	
		CON	ORG	CON	ORG	CON	ORG	CON	ORG
Farm income	kCHF/farm	136.86	151.89	97.87	110.51	87.29	97.15	114.65	111.39
Farm income per AWU	kCHF/AWU	81.02	74.27	63.05	72.02	55.58	59.77	70.60	65.83
Family farm income	kCHF/farm	99.73	106.55	77.59	92.00	73.99	84.59	87.67	90.75
Family farm income per FWU	kCHF/FWU	71.31	68.95	57.29	67.52	52.56	57.91	63.42	62.58
Total costs	kCHF/farm	189.90	183.46	139.39	118.23	96.09	84.83	154.29	112.54
Total revenues	kCHF/farm	289.63	290.01	216.98	210.24	170.08	169.41	241.97	203.30
Total direct payments	kCHF/farm	45.74	52.00	49.05	57.98	60.72	71.60	50.11	64.56
General direct payments	kCHF/farm	33.55	31.42	38.35	41.65	50.97	55.95	38.90	47.74
Ecological direct payments	kCHF/farm	8.68	17.99	8.03	12.85	5.44	10.26	7.75	12.43
Total prodcution value	kCHF/farm	192.25	176.67	136.18	108.24	82.94	63.28	151.53	96.58
Production value of crops	kCHF/farm	65.00	85.03	13.05	15.60	3.74	3.95	36.51	23.26
Production value of livestock	kCHF/farm	127.25	91.65	123.13	92.64	79.20	59.33	115.02	73.32

CON = Conventional farms; ORG = Organic farms
AWU = Average working units
FWU = Family working units

Source: own calculations based on Swiss FADN

Comparison by farm type

Substantial differences in farm income parameters can be observed for the farm types (Table 29). Among conventional farms, above-average farm income was calculated for arable farms (148 kCHF/farm), speciality crop farms (142 kCHF/farm), pig and poultry farms (140 kCHF/farm) and mixed farms (133 kCHF/farm). Other grassland farms have the lowest farm income with 69 kCHF, while dairy farms (98 kCHF/farm) and suckler cow farms (94 kCHF/farm) have farm incomes below average. A similar ranking of farm types was found for the other farm income parameters.

Analogous to the comparison by region, all organic farm types had higher income levels than their conventional equivalents, except suckler cow farms which had slightly lower farm income (91 kCHF/farm compared to 94 kCHF for conventional farms). The strongest differences were calculated for organic speciality crop farms, with 229 kCHF/farm compared to 142 kCHF/farm for conventional counterparts. Due to higher labour input the pronounced differences for speciality crop farms are overcompensated, resulting in a lower farm income per AWU (66 kCHF/AWU compared to 73 kCHF/FWU for conventional speciality crop farms) and family farm income per FWU (63 kCHF/AWU as opposed to 63 kCHF/FWU for conventional farms).

Table 29 Income structure on conventional and organic farms by farm type (2006/07)

Indicator	Unit	Arable farms	Speciality crop farms		Dairy farms		Suckler cow farms		Other grassland farms		Pig / Poultry	Mixed farms	
		CON	CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	CON	**ORG**
Farm income	**kCHF/farm**	147.64	141.83	**228.52**	97.82	**105.78**	94.01	**90.75**	69.38	**102.01**	139.54	132.99	**136.96**
Farm income per AWU	**kCHF/AWU**	109.95	72.60	**66.42**	59.88	**65.28**	69.85	**65.14**	49.96	**56.52**	83.38	76.84	**78.31**
Family farm income	**kCHF/farm**	111.87	87.03	**125.73**	79.90	**88.65**	78.48	**78.67**	57.76	**90.58**	106.52	99.14	**105.38**
Family farm income per FWU	**kCHF/FWU**	92.82	63.31	**62.81**	55.68	**62.09**	64.38	**62.48**	46.78	**55.08**	75.47	68.21	**72.40**
Total costs	**kCHF/farm**	125.49	170.00	**263.31**	117.40	**104.17**	98.66	**83.14**	83.86	**86.05**	348.67	202.79	**168.70**
Total revenues	**kCHF/farm**	237.36	257.03	**389.03**	197.30	**192.82**	177.14	**161.81**	141.62	**176.63**	455.20	301.93	**274.07**
Total direct payments	**kCHF/farm**	51.38	26.94	**40.04**	49.73	**60.26**	65.22	**70.94**	52.30	**74.27**	48.07	52.15	**63.32**
General direct payments	**kCHF/farm**	37.59	21.23	**23.06**	39.66	**44.94**	53.26	**55.17**	44.47	**57.99**	31.45	38.87	**40.13**
Ecological direct payments	**kCHF/farm**	6.98	3.52	**16.57**	6.60	**11.24**	8.31	**11.20**	4.69	**10.34**	15.13	10.05	**18.80**
Total prodcution value	**kCHF/farm**	136.86	137.53	**264.12**	117.47	**93.81**	78.03	**52.38**	61.57	**71.64**	350.58	208.74	**152.28**
Production value of crops	**kCHF/farm**	97.57	128.62	**263.59**	8.35	**7.43**	8.05	**7.64**	4.68	**6.43**	10.27	46.96	**46.38**
Production value of livestock	**kCHF/farm**	39.29	8.91	**0.52**	109.12	**86.39**	69.98	**44.74**	56.89	**65.21**	340.31	161.79	**105.90**

CON = Conventional farms; ORG = Organic farms
AWU = Average working units
FWU = Family working units

Source: own calculations based on Swiss FADN

Total costs are lower on organic farms, with the exception of other grassland farms and speciality crop farms. Total revenues are lower on organic farms, with the exception of speciality crop farms and other grassland farms. The same pattern emerges for the production value, which is lower on most organic farm groups, except for other grassland farms and speciality crop farms. While crop production values are almost equal between the farm types (with the exception of speciality crop farms), livestock-related production values are generally lower. Only organic other grassland farms produce higher values than their conventional counterparts (65 kCHF/farm compared with 57 kCHF on conventional farms).

Direct payments are higher on organic farms throughout the farm types. Thus the direct payments compensate the lower production values, resulting in the above mentioned higher incomes of organic farms.

7.2.3 Policy uptake

As the cost-effectiveness of organic farming is compared with agri-environmental policy measures in Section 7.4, this section focuses on the policy uptake of these policy measures (see Section 4.3 for a description of the policy measure). However, other ECA measures, such as extensive pastures, mixed and rotational fallows, which are also explicitly included in the model, are discussed as well.

This section is structured as follows: To begin, the uptake level of the extenso payments for regional farm groups and farm types is examined. Following this, uptake levels of ECA measures are analysed, starting with total ECA area and then moving on to the specific measures: less intensive and extensive meadows, extensive pastures, mixed and rotational fallows. As in the previous section, first regional farm groups are compared, followed by a comparison by farm type.

Extenso payments

Comparison by region

Organic farms automatically fulfil the eligibility criteria for extenso payments. Therefore, they show an uptake level of 100 %, if the farm group cultivates the relevant crop (either cereals or oilseed rape) (Table 30).

On conventional farms, the share of extenso cereals in total grain area increases from 44 % in the lowlands up to 92 % in the mountains. Conventional farms in lowlands and hill areas take up extenso payments for rape on about 28 % of the total rape area, while in mountain areas no rape is grown. However, as the total shares of grains and rape fall with the rising altitude of the farms, the extenso uptake relative to total UAA decreases continuously from lowlands to mountain areas. The total area under crops cultivated according to extenso standards is about 16.8 % on conventional farms and 4.4 % on organic farms due to the lower share of cereals on total UAA.

Table 30 Uptake of extenso payments on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountains		Total	
		CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	**ORG**
Shares relative to the farms' crop area									
Intensive grains	**%**	55.6	**-**	29.4	**-**	7.8	**-**	50.2	**-**
Extenso grains	**%**	44.4	**100.0**	70.6	**100.0**	92.2	**100.0**	49.8	**100.0**
Intensive rape	**%**	72.4	**-**	72.4	**-**	-	**-**	72.4	**-**
Extenso rape	**%**	27.6	**100.0**	27.6	**100.0**	-	**-**	27.6	**100.0**
Shares relative to farm UAA									
Intensive grains	**%**	12.8	**-**	3.1	**-**	0.1	**-**	7.4	**-**
Extenso grains	**%**	10.2	**17.7**	7.4	**4.2**	0.7	**0.1**	7.3	**4.4**
Intensive rape	**%**	2.6	**-**	0.6	**-**	-	**-**	1.5	**-**
Extenso rape	**%**	1.0	**0.2**	0.2	**0.0**	-	**-**	0.6	**0.0**
Total extenso	**%**	26.6	**17.9**	11.4	**4.2**	0.7	**0.1**	16.8	**4.4**

CON = Conventional farms; ORG = Organic farms Source: own calculations based on Swiss FADN

Comparison by farm type

Relative to the total grain area, the highest extenso grain uptake occurs on other grassland farms, dairy farms, and suckler cow farms (about 80 %), while specialised arable farms and pig and poultry and mixed farms take up extenso grain payments on 44 to 46 % of their total grain area (Table 31). The same pattern applies to extenso uptake for rape, although the uptake levels of arable and mixed farms are lower, at 23 to 25 %.

Looking at the uptake levels relative to the total UAA, it becomes evident that, similar to the regional analysis, those farm types with high uptake levels grow the respective crop to a lower absolute level. Rape is grown almost exclusively on arable farms. Regarding cereals, only mixed and speciality crop farms cultivate a noteworthy share alongside arable farms.

Table 31 Uptake of extenso payments on conventional and organic farms by farm type (2006/07)

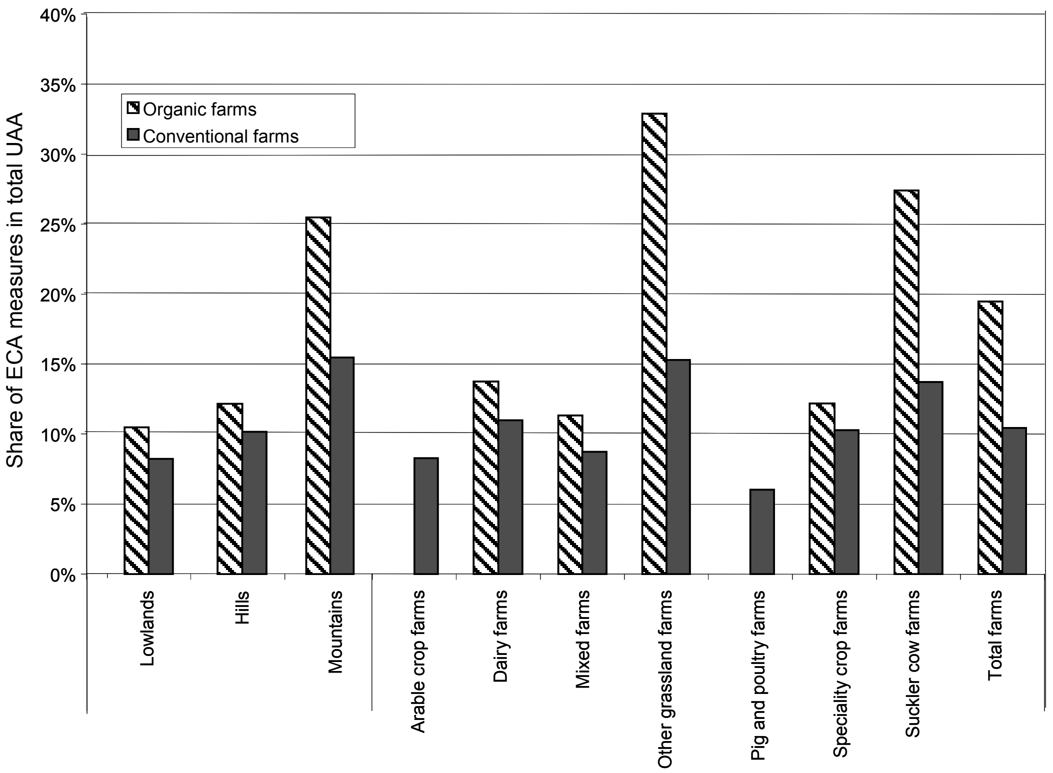
Indicator	Unit	Arable farms	Speciality crop farms		Dairy farms		Suckler cow farms		Other grassland farms		Pig / Poultry	Mixed farms	
		CON	CON	ORG	CON	ORG	CON	ORG	CON	ORG	CON	CON	ORG
Shares relative to the farms' crop area													
Intensive grains	%	55.2	32.1	-	22.1	-	19.4	-	19.0	-	47.0	53.6	-
Extenso grains	%	44.8	67.9	**100.0**	77.9	**100.0**	80.6	**100.0**	81.0	-	53.0	46.4	**100.0**
Intensive rape	%	74.7	41.2	-	59.3	-	100.0	-	-	-	-	77.0	-
Extenso rape	%	25.3	58.8	-	40.7	-	-	-	100.0	-	-	23.0	**100.0**
Shares relative to farm UAA													
Intensive grains	%	21.7	6.5	-	0.6	-	0.6	-	0.5	-	2.4	11.7	-
Extenso grains	%	17.6	13.8	**22.3**	2.1	**1.4**	2.5	**1.5**	2.0	-	2.7	10.2	**18.2**
Intensive rape	%	6.3	1.8	-	0.0	-	0.2	-	-	-	-	2.0	-
Extenso rape	%	2.1	2.6	-	0.0	-	-	-	0.1	-	-	0.6	**0.2**
Total extenso	%	47.7	24.6	**22.3**	2.7	**1.4**	3.3	**1.5**	2.6	-	5.2	24.4	**18.5**

CON = Conventional farms; ORG = Organic farms

Source: own calculations based on Swiss FADN

Ecological compensation areas

The share of ecological compensation areas (ECA) ranges from 8 % in the lowlands to 25 % in the mountain areas. There are also substantial differences between the farm types. Conventional arable farms, mixed farms and pig/poultry farms have 6 to 8 % of their UAA cultivated as ECA. Higher shares of 10 to 15 % are taken up by other grassland farms, suckler cow farms, special crop farms and dairy farms (Figure 20). Organic farms demonstrate higher uptake levels of ECA measures in all regions and for all farm types. The differences are most striking in the mountain areas and for other grassland farms and suckler cow farms (see also Table 32).



Source: own calculations based on Swiss FADN and SALCA data

Figure 20 Share of ECA measures in UAA on conventional and organic farms by region and farm type (2006/07)

Comparison by region

Table 32 shows the uptake levels of selected ECA measures both relative to total crop and UAA. In total, about 80 % of the permanent meadows cultivated in Switzerland are intensively managed, 7.4 % are less intensively managed and 12.8 % are extensively managed. Regional differences in uptake levels are relatively small. The maximum share of extensive meadows in total meadows can be found in the lowlands, while the highest share of less intensive meadows was found in the mountain areas. Relative to total UAA, an increasing portion of both extensive and less intensive meadows from the lowlands to the mountain area was found.

Organic farms have generally higher shares of less intensive and extensive meadows relative to total meadows and total UAA (Table 32). However, looking at individual regions, the share of extensive meadows is smaller in hill areas. The share of less intensive meadows is smaller in the lowlands and in hill areas. With regard to total UAA shares, there are no major differences in the share of less intensive and extensive meadows between organic and conventional farms in the lowlands and hill areas. In the mountain areas, however, the share of less intensive and extensive meadows is, at 10.7 % and 13.6 % respectively, higher on organic farms than on conventional farms (6.3 % and 7.1 %, respectively).

Table 32 Uptake of ECA measures on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountains		Total	
		CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	**ORG**
Shares relative to the farms' crop area									
Intensive meadows	**%**	75.7	**76.8**	83.5	**87.4**	82.4	**71.2**	80.6	**75.0**
Less intensive meadows	**%**	6.0	**5.2**	6.3	**4.0**	8.3	**12.6**	6.9	**10.2**
Extensive meadows	**%**	18.3	**18.0**	10.2	**8.6**	9.4	**16.1**	12.5	**14.8**
Intensive pastures	**%**	86.6	**87.7**	86.4	**65.2**	83.0	**87.6**	84.9	**83.1**
Extensive pastures	**%**	13.4	**12.3**	13.6	**34.8**	17.0	**12.4**	15.1	**16.9**
Shares relative to farm UAA									
Intensive meadows	**%**	21.5	**28.6**	45.7	**59.3**	62.3	**60.1**	37.0	**53.8**
Less intensive meadows	**%**	1.7	**1.9**	3.4	**2.7**	6.3	**10.7**	3.2	**7.3**
Extensive meadows	**%**	5.2	**6.7**	5.6	**5.8**	7.1	**13.6**	5.7	**10.6**
Intensive pastures	**%**	3.0	**5.5**	8.2	**7.3**	14.0	**11.4**	6.8	**9.4**
Extensive pastures	**%**	0.5	**0.8**	1.3	**3.9**	2.9	**1.6**	1.2	**1.9**
Mixed fallows	**%**	0.4	**0.2**	0.1	**-**	-	**-**	0.2	**0.0**
Rotational fallows	**%**	0.2	**0.1**	0.0	**-**	-	**-**	0.1	**0.0**
Other ECA	**%**	0.5	**0.9**	0.8	**0.6**	1.9	**1.2**	0.9	**1.0**
Total ECA	**%**	8.5	**10.7**	11.2	**13.0**	18.1	**27.0**	11.4	**20.9**

CON = Conventional farms; ORG = Organic farms Source: own calculations based on Swiss FADN

The share of extensive pastures in total pastures increases from 13.3 % in the lowlands to 15.5 % in the hill areas and reaches 16.1 % in the mountain areas. The share of extensive pastures in total UAA rises from 3.1 % in the lowlands to 8.1 % in the hill areas and 13.4 % in the mountain areas. Organic farms tend to have a higher share of extensive pastures both relative to total UAA and to total pastures. Relative to UAA, this trend is confirmed for all regions. However, relative to the share of total pastures, extensive pastures are less abundant on organic farms in the lowlands and mountain areas.

Mixed and rotational fallow have a total share of 0.2 and 0.1 % out of total UAA, respectively, almost all of which is located in the lowlands.

Comparison by farm type

Table 33 shows the uptake levels of less intensive and extensive meadows, extensive pastures as well as for fallows and other ECA measures for the different farm types. Relative to the total share of the respective crop, the share of less intensive and extensive meadows is highest on arable crop farms (70 %) and speciality crop farms (65 %). Specialised livestock farms, such as dairy or suckler cow farms, have shares of less intensive and extensive meadows of less than 20 %. Pig and poultry farms have less than 10 % of extensive meadows. About one quarter of meadows on mixed farms is less intensive or extensive. Relative to total UAA, specialised crop farms have the lowest share of extensive and less intensive grassland, while dairy, suckler cow and other grassland farms have shares of more than 10 % of less intensive or extensive meadows. Organic farms take up more ECA grassland compared to their conventional counterparts. Uptake levels are about 60 to 100 % higher, while there are only slight differences for mixed farms, if the total grassland area is taken as the point of reference.

Similar to meadows, extensive pastures are most abundant on arable farms. Other grassland farms also have significant shares of extensive pastures, relative to total pastures. Similar shares of extensive pastures are found on organic and conventional farms. Relative to total UAA, arable and speciality crop farms hold the lowest share of extensive pastures, while livestock-dominated farm types show slight variations between the farming systems.

Rotational and mixed fallows make up around 1 % of total UAA on arable and speciality crop farms and are only minimally represented among the other farm types. Organic speciality crop farms have higher shares of mixed fallows and lower shares of rotational fallows than the corresponding conventional farm type.

Table 33 Uptake of ECA measures on conventional and organic farms by farm type (2006/07)

Indicator	Unit	Arable farms	Speciality crop farms		Dairy farms		Suckler cow farms		Other grassland farms		Pig / Poultry	Mixed farms	
		CON	CON	ORG	CON	ORG	CON	ORG	CON	ORG	CON	CON	ORG
Shares relative to the farms' crop area													
Intensive meadows	%	29.7	34.7	**20.3**	86.2	**84.2**	81.3	**67.2**	81.5	**64.1**	90.9	74.6	**73.7**
Less intensive meadows	%	12.1	14.4	**-**	6.1	**4.8**	7.6	**13.3**	8.4	**20.9**	4.1	7.0	**7.0**
Extensive meadows	%	58.2	50.9	**79.7**	7.7	**11.0**	11.2	**19.5**	10.1	**15.0**	5.1	18.3	**19.3**
Intensive pastures	%	77.3	88.1	**-**	86.3	**86.5**	88.3	**79.1**	77.3	**78.1**	98.0	86.2	**88.4**
Extensive pastures	%	22.7	11.9	**-**	13.7	**13.5**	11.7	**20.9**	22.7	**21.9**	2.0	13.8	**11.6**
Shares relative to farm UAA													
Intensive meadows	%	2.8	4.0	**2.6**	61.6	**67.8**	50.0	**51.1**	60.1	**53.9**	57.2	22.5	**28.7**
Less intensive meadows	%	1.1	1.6	**-**	4.4	**3.8**	4.6	**10.1**	6.2	**17.5**	2.6	2.1	**2.7**
Extensive meadows	%	5.5	5.8	**10.2**	5.5	**8.9**	6.9	**14.8**	7.4	**12.6**	3.2	5.5	**7.5**
Intensive pastures	%	1.3	0.5	**-**	9.9	**9.1**	14.3	**10.2**	11.6	**11.3**	6.9	4.7	**8.7**
Extensive pastures	%	0.4	0.1	**-**	1.6	**1.4**	1.9	**2.7**	3.4	**3.2**	0.1	0.8	**1.1**
Mixed fallows	%	0.6	0.8	**1.0**	0.0	**0.0**	0.0	**0.0**	0.0	**-**	0.1	0.3	**0.1**
Rotational fallows	%	0.4	1.6	**0.7**	0.0	**-**	-	**-**	-	**-**	-	0.1	**-**
Other ECA	%	0.4	0.3	**0.5**	1.3	**1.5**	1.8	**0.9**	1.5	**0.1**	0.3	0.6	**0.8**
Total ECA	%	8.5	10.3	**12.3**	12.8	**15.7**	15.2	**28.5**	18.5	**33.4**	6.3	9.3	**12.3**

CON = Conventional farms; ORG = Organic farms Source: own calculations based on Swiss FADN

7.2.4 Fossil energy use

Results on fossil energy use for the farming systems are compared first by region followed by a comparison by farm type. Each comparison is discussed in relation to the conventional farm groups in order to reveal general differences between regions and farm types. This is followed by a description of the differences between organic and conventional farms.

Comparison by region

Figure 21 depicts fossil energy use per ha from a farm-level life cycle perspective, *i.e.* taking into account the energy input for all processes and inputs needed, in lowlands, hill and mountain areas (see Section 6.3.7). Conventional farms have an average fossil fuel demand of 44.3 GJ/ha, while average fossil energy use is 53.5 GJ/ha in the lowlands and 44 GJ/ha in hill areas; energy use in mountain areas is only 24.9 GJ/ha (Figure 21).

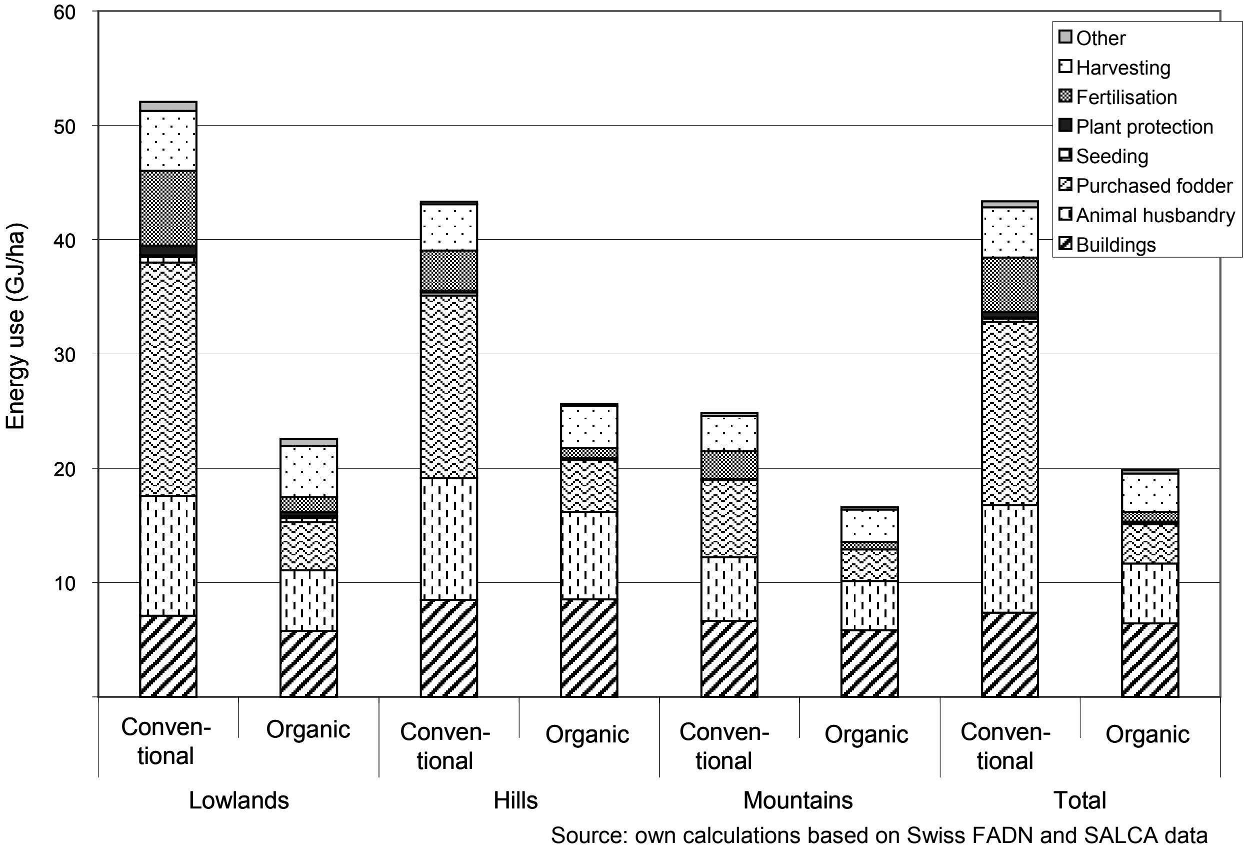


Figure 21 Fossil energy use per ha on conventional and organic farms by region (2006/07)

Animal production-related activities have the largest share of total energy use; most prominently, purchased fodder accounts for 16 GJ/ha over all regions. There are, however, remarkable differences between the regions, with 20.3 GJ/ha in the lowlands, 16 GJ/ha in the hill areas and a relatively low share of 6.8 GJ/ha in the mountain regions. Furthermore, significant amounts of fossil energy use can be attributed to general animal husbandry (9.4 GJ/ha). While energy use for animal husbandry is equally high in the lowlands and in hill areas (approximately 10.5 GJ/ha), it is significantly lower in the mountain regions (5.6 GJ/ha). Whilst milking accounts for only 1.6 GJ/ha (average over regions), other processes in livestock housing systems, *i.e.* feeding, ventilation, heating (in case of pig housing systems), the use of water, lubricating oil and clearing agents, determine most of the energy use (average 7.8 GJ/ha). Further significant amounts of fossil energy use can be allocated to buildings (7.4 GJ/ha), predominantly livestock housing systems (4.6 GJ/ha) (Table 34).

Table 34 Fossil energy use per ha on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountains		Total	
		CON	ORG	CON	ORG	CON	ORG	CON	ORG
Total energy use	**GJ/ha**	**53.54**	**23.96**	**43.97**	**26.01**	**24.88**	**16.62**	**44.28**	**20.20**
Buildings	**GJ/ha**	**7.09**	**5.78**	**8.48**	**8.52**	**6.64**	**5.83**	**7.37**	**6.41**
Crop and machine storage	GJ/ha	2.08	2.33	3.29	3.30	3.53	2.95	2.75	2.90
Livestock housing (buildings)	GJ/ha	5.01	3.45	5.19	5.22	3.12	2.89	4.62	3.51
Animal husbandry	**GJ/ha**	**10.51**	**5.31**	**10.69**	**7.68**	**5.56**	**4.30**	**9.42**	**5.25**
Fences	GJ/ha	0.02	0.03	0.05	0.05	0.06	0.06	0.04	0.05
Livestock housing (processes)	GJ/ha	8.85	3.96	8.71	6.16	4.43	3.56	7.79	4.21
Milking	GJ/ha	1.64	1.32	1.93	1.47	1.07	0.68	1.59	0.99
Purchased fodder	**GJ/ha**	**20.34**	**4.21**	**15.94**	**4.49**	**6.77**	**2.76**	**15.99**	**3.44**
Tillage	**GJ/ha**	**1.56**	**1.39**	**0.65**	**0.38**	**0.06**	**0.02**	**0.97**	**0.38**
Seeding	**GJ/ha**	**0.63**	**0.50**	**0.25**	**0.18**	**0.02**	**0.01**	**0.38**	**0.15**
Seeds	GJ/ha	0.44	0.35	0.17	0.13	0.02	0.01	0.27	0.11
Mechansiation	GJ/ha	0.19	0.15	0.08	0.04	0.01	0.00	0.11	0.04
Plant protection	**GJ/ha**	**0.85**	**0.40**	**0.19**	**0.02**	**0.08**	**0.00**	**0.49**	**0.09**
Insecticides	GJ/ha	0.13	-	0.01	-	0.00	-	0.06	-
Fungicides	GJ/ha	0.07	0.00	0.01	0.00	0.00	0.00	0.03	0.00
Herbicides	GJ/ha	0.24	-	0.10	-	0.07	-	0.17	-
Other plant protection	GJ/ha	0.02	0.01	0.00	-	0.00	-	0.01	0.00
Mechansiation	GJ/ha	0.26	0.02	0.05	0.00	0.00	0.00	0.14	0.00
Plant care	GJ/ha	0.14	0.38	0.02	0.02	0.00	0.00	0.07	0.08
Fertilisation	**GJ/ha**	**6.54**	**1.29**	**3.50**	**0.89**	**2.40**	**0.64**	**4.74**	**0.83**
Mechansiation	GJ/ha	1.53	0.98	1.23	0.75	0.96	0.54	1.32	0.68
Organic fertilisation	GJ/ha	0.11	0.29	0.13	0.13	0.11	0.09	0.12	0.14
Mineral nitrogen fertiliser	GJ/ha	3.28	-	1.61	-	0.99	-	2.29	-
Mineral phosphorus fertiliser	GJ/ha	0.93	0.01	0.46	0.01	0.34	0.01	0.67	0.01
Mineral potassium fertiliser	GJ/ha	0.68	-	0.08	-	0.00	-	0.36	-
Harvesting	**GJ/ha**	**5.25**	**4.49**	**4.03**	**3.65**	**3.09**	**2.85**	**4.41**	**3.36**
Mechansiation	GJ/ha	4.26	4.07	3.85	3.58	3.07	2.83	3.87	3.25
Drying	GJ/ha	0.87	0.35	0.13	0.04	0.01	0.00	0.47	0.08
Transports (field-farm)	GJ/ha	0.12	0.07	0.04	0.02	0.01	0.01	0.08	0.03
Other processes	**GJ/ha**	**0.78**	**0.60**	**0.23**	**0.22**	**0.25**	**0.21**	**0.51**	**0.29**

CON = Conventional farms; ORG = Organic farms Source: own calculations based on Swiss FADN and SALCA data

Energy use related to crop production is comparatively low, with about 4.7 GJ/ha for fertilisation and 4.4 GJ/ha for harvesting being the major categories. Energy use significantly varies among the regions, with 5.3 GJ/ha in the lowlands, 4 GJ/ha in the hill areas and 3.1 GJ/ha in the mountain regions. Harvest-related energy use also decreases with higher farm altitude, though not as markedly as for fertiliser use. The other categories of energy use related to crop production, namely crop protection, seeding, and tillage, contribute only 1 GJ/ha or less to the total fossil energy use.

Total energy use on organic farms is markedly lower than on conventional farms (54 % less energy use per ha). This applies to all regions, while the smallest differences can be found in the mountain region (33 % less energy use per ha) and the largest differences in the lowlands (55 %). It should be noted that, contrary to conventional farms, energy use per ha on organic farms is highest in the hill regions (26 GJ/ha) rather than in the lowlands.

These disparities between conventional and organic farms are caused by differences in individual energy-use categories, primarily energy for purchased fodder (total organic farms 3.4 GJ/ha compared to 16 GJ/ha on conventional farms) (Table 34). The gap in purchased fodder can be attributed to the following causes: First, structural differences between the regions, *i.e.* different farm-type distributions (see subsequent pages); second, lower stocking rates on organic farms, particularly in the cereals-intensive production of pigs and poultry (Section 7.2.1), third, restrictions applying to the share of purchased fodder on organic farms, fourth, a higher relative concentration of organic farms in the mountain regions, where the conventional farms also show lower fossil energy use. While the fourth reason is offset in the regional comparison, the first, second, and third reasons remain, resulting in 4.5 GJ/ha (hills) to 2.8 GJ/ha (mountains) energy use for purchased fodder. The same reasons influence the difference in energy use between the farming systems for animal husbandry and buildings. However, differences between the farming systems are much smaller than for fodder purchase.

Regarding energy use related to crop production, the absolute differences between the farming systems are not as large as for the above mentioned categories, particularly for tillage, seeding and harvesting. It needs to be stressed that due to the overall lower energy use on organic farms, the relative share of these energy categories acquires greater significance on organic farms.

Concerning crop protection, organic farms show high relative differences to their conventional regional equivalents (more than 50 % in all regions). However, these are of little importance, due to the low absolute level of this category. Fertilisation-related energy use is much lower on organic farms due to the non-usage of mineral nitrogen fertiliser. Mechanisation and organic fertilisation are, however, at a similar level.

The most significant differences among the categories can be observed for purchased fodder, which alone accounts for 20 GJ/ha in the lowlands and 15 GJ/ha in the hill areas. In contrast, only 6 GJ/ha are used for purchased fodder in the mountain regions. Energy use for both animal husbandry and buildings is highest in the hill regions due to the high stocking density. Slightly lower fossil energy use can be found in the lowlands, whereas the lowest fossil energy use per ha was modelled for the mountain regions. Energy demand for tillage, fertilisation, and seeding declines from lowlands to mountains. This can be attributed to the lower share of arable land with rising altitude. Energy use for harvesting also decreases from lowlands to mountain areas for the same reason as well as due to the fact that meadows can be cut fewer times in mountain regions thanks to a shorter vegetation period.

Comparison by farm type

The differences between farm types are bigger than the differences between regions. Figure 22 shows the differences between all farm types, except pig and poultry farms. Pig and poultry farms have been excluded from the graph in order to maintain its readability for the other farm groups; due to the high stocking rates and the high share of cereal-fed livestock, these farms have a calculative energy demand of 194.9 GJ/ha. Beside pig and poultry farms, conventional mixed farms have the highest total energy use (60.1 GJ/ha). The average energy use (as a sum of all energy use components) of dairy, suckler cow, other grassland, arable and finally speciality crop farms ranges from 20 to 30 GJ/ha (Table 35).

Differences in energy-use components between the farm types are remarkable. Conventional mixed farms have the highest energy share due to high stocking rates, particularly because in this farm type, pig and poultry stocking rates are significant[66]. This causes energy use for purchased fodder, animal husbandry and buildings to increase to 47 GJ/ha.

Arable and speciality crop farms show very low energy use for livestock-related activities with less than 8 GJ/ha (arable farms) compared to 3.5 GJ/ha (conventional speciality crop farms).

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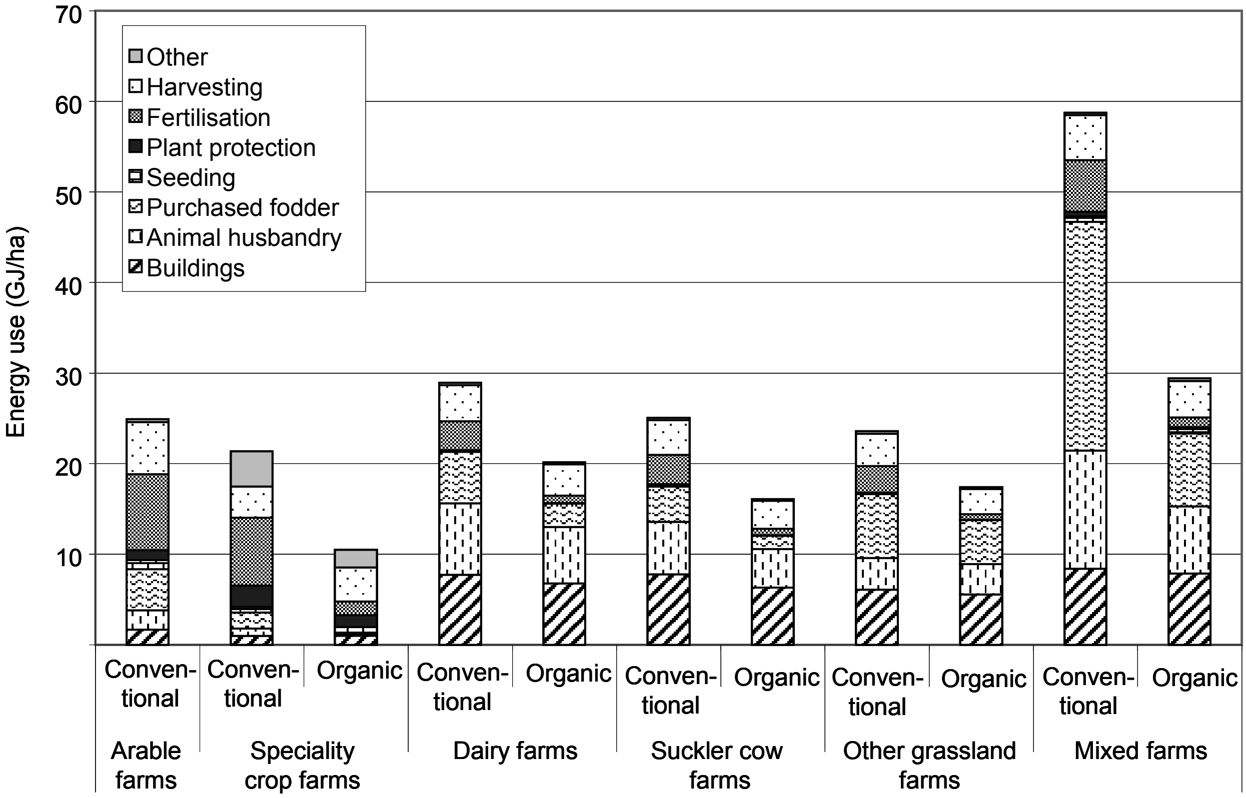
[66] One in four mixed farm types (which have been merged to one farm group due to sufficient FADN farms). One of these farm types is 'Mixed pig and poultry farms' (see page Table 17, page 116).



Source: own calculations based on Swiss FADN and SALCA data

Figure 22 Fossil energy use per ha on conventional and organic farms by farm type (2006/07)

On all farm types, except for arable and speciality crop farms, energy use for crop production on conventional farms is responsible for only a minor share of total energy consumption. Only the amount of energy used for harvesting and fertilisation is significant.

Harvesting-related energy use varies little among the farm types, while a strong fluctuation was identified for fertilisation. Particularly those farm types with high shares of arable and permanent land, namely arable farms, speciality crop farms and mixed farms, use 5.7 to 8.4 GJ/ha on fertilisation, while the largest share of this energy use can be allocated to mineral fertilisers, especially nitrogen fertiliser.

Table 35 Fossil energy use per ha on conventional and organic farms by farm type (2006/07)

Indicator	Unit	Arable farms	Speciality crop farms		Dairy farms		Suckler cow farms		Other grassland farms		Pig / Poultry	Mixed farms	
		CON	CON	ORG	CON	ORG	CON	ORG	CON	ORG	CON	CON	ORG
Total energy use	**GJ/ha**	**27.55**	**22.92**	**12.79**	**29.16**	**20.29**	**25.38**	**16.25**	**23.76**	**17.42**	**194.91**	**60.14**	**30.72**
Buildings	**GJ/ha**	**1.67**	**0.97**	**0.99**	**7.71**	**6.78**	**7.76**	**6.32**	**6.10**	**5.57**	**23.92**	**8.39**	**7.85**
Crop and machine storage	GJ/ha	0.47	0.48	0.98	3.80	3.32	3.36	2.90	3.57	2.85	4.02	2.29	2.25
Livestock housing (buildings)	GJ/ha	1.20	0.49	0.02	3.91	3.46	4.40	3.42	2.52	2.71	19.90	6.09	5.61
Animal husbandry	**GJ/ha**	**2.14**	**0.82**	**0.02**	**7.93**	**6.23**	**5.81**	**4.23**	**3.50**	**3.33**	**45.47**	**13.04**	**7.43**
Fences	GJ/ha	0.01	0.00	0.00	0.05	0.05	0.06	0.05	0.06	0.06	0.04	0.03	0.04
Livestock housing (processes)	GJ/ha	1.75	0.69	0.02	5.43	4.34	5.73	4.15	3.30	3.16	42.98	10.99	6.23
Milking	GJ/ha	0.39	0.12	-	2.45	1.84	0.03	0.03	0.14	0.12	2.44	2.02	1.16
Purchased fodder	**GJ/ha**	**4.54**	**1.77**	**0.10**	**5.63**	**2.59**	**3.90**	**1.49**	**7.05**	**4.88**	**115.42**	**25.18**	**8.03**
Tillage	**GJ/ha**	**2.64**	**1.56**	**2.30**	**0.26**	**0.13**	**0.34**	**0.16**	**0.17**	**0.02**	**0.49**	**1.49**	**1.30**
Seeding	**GJ/ha**	**0.99**	**0.64**	**0.87**	**0.10**	**0.05**	**0.12**	**0.06**	**0.06**	**0.02**	**0.20**	**0.61**	**0.51**
Seeds	GJ/ha	0.69	0.45	0.63	0.07	0.03	0.08	0.04	0.04	0.01	0.14	0.44	0.37
Mechansiation	GJ/ha	0.31	0.18	0.25	0.03	0.02	0.04	0.02	0.02	0.00	0.06	0.18	0.14
Plant protection	**GJ/ha**	**1.09**	**2.34**	**1.29**	**0.13**	**0.01**	**0.15**	**0.01**	**0.12**	**0.01**	**0.17**	**0.53**	**0.20**
Insecticides	GJ/ha	0.06	0.68	-	0.01	-	0.01	-	0.01	-	0.00	0.03	-
Fungicides	GJ/ha	0.07	0.21	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.04	0.00
Herbicides	GJ/ha	0.34	0.44	-	0.09	-	0.09	-	0.08	-	0.10	0.19	-
Other plant protection	GJ/ha	0.05	0.02	0.03	0.00	0.00	0.00	-	0.00	-	0.00	0.01	0.00
Mechansiation	GJ/ha	0.38	0.63	0.03	0.02	0.00	0.02	0.00	0.02	0.00	0.04	0.17	0.01
Plant care	GJ/ha	0.20	0.35	1.22	0.01	0.01	0.01	0.00	0.00	0.01	0.02	0.08	0.18
Fertilisation	**GJ/ha**	**8.40**	**7.47**	**1.51**	**3.16**	**0.81**	**3.24**	**0.70**	**2.91**	**0.63**	**4.03**	**5.68**	**1.09**
Mechansiation	GJ/ha	1.54	1.33	0.86	1.18	0.69	1.17	0.60	1.09	0.53	1.42	1.47	0.89
Organic fertilisation	GJ/ha	0.07	0.05	0.64	0.13	0.11	0.13	0.10	0.12	0.09	0.16	0.12	0.19
Mineral nitrogen fertiliser	GJ/ha	4.53	3.82	-	1.34	-	1.45	-	1.26	-	1.75	2.83	-
Mineral phosphorus fertiliser	GJ/ha	1.16	1.02	0.00	0.46	0.01	0.44	0.01	0.42	0.01	0.61	0.79	0.01
Mineral potassium fertiliser	GJ/ha	1.10	1.25	-	0.04	-	0.06	-	0.03	-	0.10	0.47	-
Harvesting	**GJ/ha**	**5.77**	**3.50**	**3.76**	**3.99**	**3.46**	**3.81**	**3.09**	**3.56**	**2.76**	**4.97**	**4.97**	**4.02**
Mechansiation	GJ/ha	3.55	2.15	2.54	3.92	3.43	3.73	3.06	3.50	2.75	4.87	4.28	3.76
Drying	GJ/ha	2.02	1.16	1.08	0.05	0.01	0.05	0.01	0.04	0.00	0.07	0.59	0.20
Transports (field-farm)	GJ/ha	0.20	0.19	0.13	0.02	0.02	0.03	0.02	0.02	0.01	0.04	0.10	0.06
Other processes	**GJ/ha**	**0.30**	**3.87**	**1.94**	**0.26**	**0.23**	**0.25**	**0.18**	**0.30**	**0.21**	**0.23**	**0.26**	**0.29**

CON = Conventional farms; ORG = Organic farms

Source: own calculations based on Swiss FADN and SALCA data

Differences between conventional and organic farms by farm type are less pronounced than by region. However, for mixed farms and speciality crop farms in particular, organic farms use only 50 % of the energy per ha of their conventional counterparts. For the ruminant-focussed farm types, however, differences are lower, at just 30 %. Figure 22 reveals that the differences in energy use between the farming systems can be attributed to the following components: First, energy use for purchased fodder is substantially lower due to the above mentioned lower stocking rate, particularly for cereal-fed livestock. Second, fertiliser-related energy use is markedly lower (for instance, 1.1 GJ/ha on organic mixed farms, compared to

5.7 GJ/ha on conventional ones), particularly because mineral nitrogen fertilisers are not used at all and other mineral fertilisers are used to a lesser extent on organic farms.

Table 36 summarises the relative difference in average cumulative energy use per ha[67]. It shows that differences between organic and conventional farms are large in the lowlands in particular, where per-ha energy use is less than half compared to conventional farms (reduction in 55 %). A substantially lower energy demand can also be applied to hill regions (reduction in 41 %), while the mountain areas show the smallest absolute and relative differences (reduction in 33 %) between the farming systems. A similar pattern can be observed for the farm types, as mixed farms show the greatest relative differences with 49 %, whereas organic dairy and suckler cow farms have only a 30 to 33 % lower energy demand per ha.

Table 36 Relative difference in fossil energy use per ha between conventional and organic farms by region and farm type (2006/07)

Indicator	Unit	Low-lands	Hills	Moun-tains	Dairy farms	Suckler cow farms	Mixed farms	Total farms
Energy use on conventional farms	**GJ/ha**	53.54	43.97	24.88	29.16	25.38	60.14	44.28
Energy use on organic farms	**GJ/ha**	23.96	26.01	16.62	20.29	16.25	30.72	20.20
Relative difference in energy use per ha	**%**	55.25	40.85	33.21	30.44	35.99	48.92	54.39

Source: own calculations based on Swiss FADN and SALCA data

* energy use on conventional farms = 100 %

In summary, it can be said that the dissimilarities in energy use between the farming systems can be attributed to a lower purchase of concentrate fodder, a lower stocking density, and the ban on mineral nitrogen fertiliser input.

7.2.5 Habitat quality

Biodiversity is assessed in terms of species diversity, expressed as average habitat quality for different species. Figure 23 shows the average habitat quality (AHQ) in conventional and organic farm groups for all species groups that were included in the analysis. The AHQ over all farms in Switzerland is about 16 % of a hypothetical maximum habitat quality, which

[67] The farm types for which no organic sub-group could be formed, namely arable and pig and poultry farms, and those for which the sub-groups could only be formed using a weak sample of FADN farms, are omitted as a separate farm group.

would be achieved, if the most beneficial management practices were applied and all harmful management practices were avoided on each and every hectare (see 6.3.7 for further descriptions to the methods). On the average organic farm the AHQ reaches approximately 25 %.

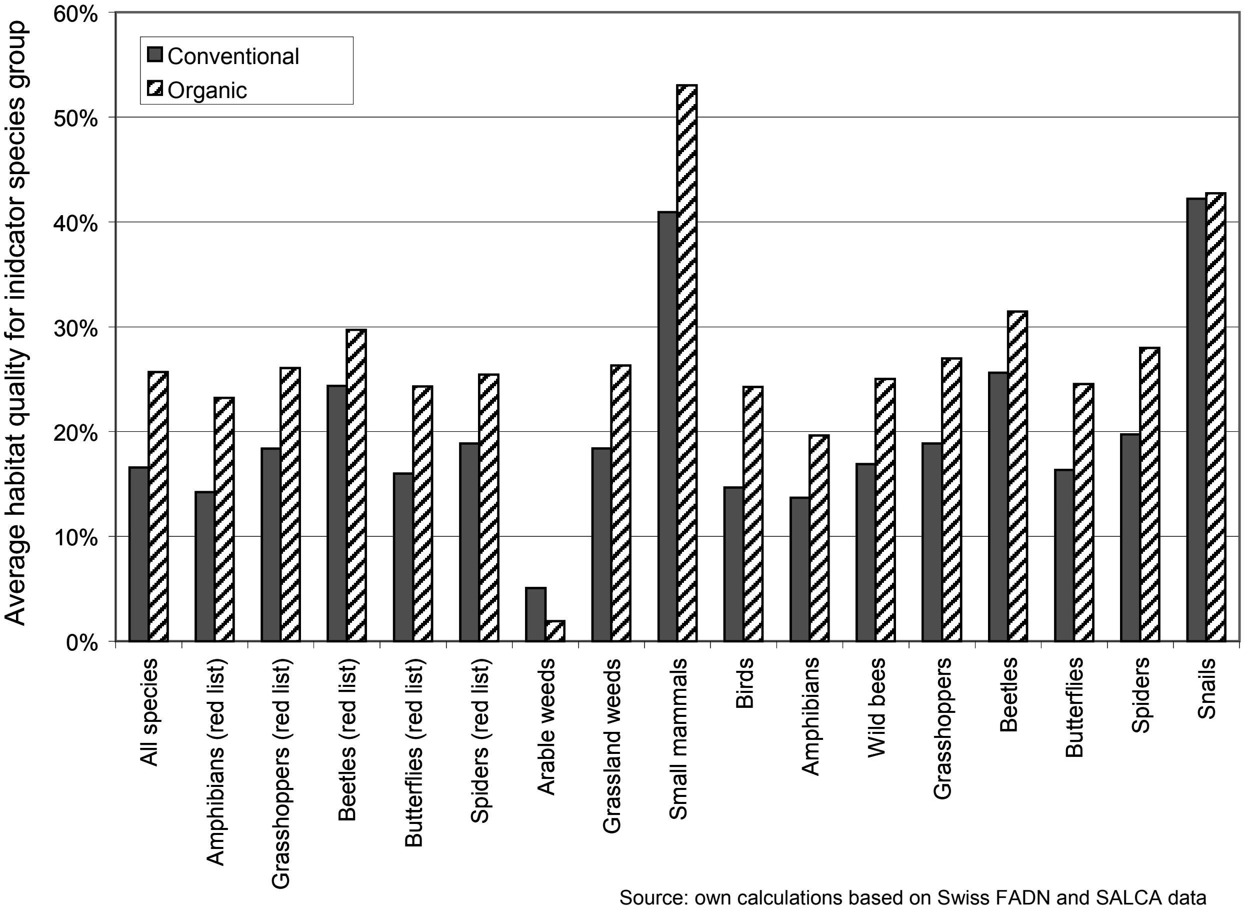


Figure 23 Average habitat quality by species group (average over all regions and farm types, 2006/07)

The highest benchmarking values on conventional farms were achieved for snails (42 %) and small mammals (41 %), whereas the lowest habitat quality relative to the maximum achievable score was calculated for arable weeds (5 %)[68]. The rest of the indicator groups score 15 to 25 % on conventional farms.

Differences in habitat quality between conventional and organic farms are relatively homogeneous over all indicator groups. The average habitat quality on organic farms tends to be 5 to 10 % higher related to the total benchmark. Related to the average habitat quality on conven-

[68] This is due to the fact that most of the agricultural utilised area is grassland, which is no habitat for arable weeds and receives a theoretical habitat score of 0.

tional farms, the relative difference ranges from 22 (beetles) to 66 % (birds). Exceptions were both snails, for which the average habitat quality on both faming systems is about equal, and arable weeds, for which the overall habitat quality on organic farms is lower due to the lower share of arable land. Due to these relatively homogeneous responses of the indicator species and due to the high relevance of red-list species, the following paragraphs focus on the total habitat quality indicator and the species groups with 'high ecological requirements' (amphibians, grasshoppers, beetles, butterflies and spiders). Finally, relevant differences for the other species are described.

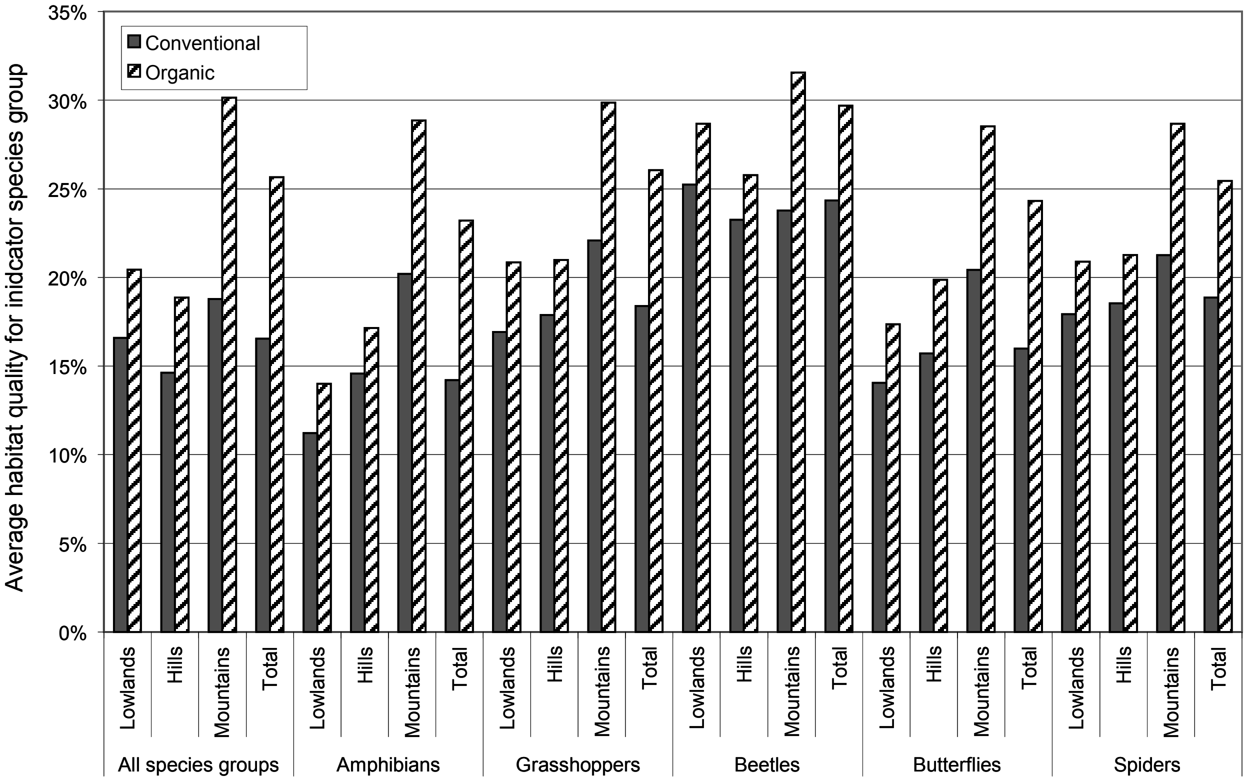
General habitat quality and habitat quality for red-list species

Comparison by region

Compared by region, the average habitat quality on conventional farms varies only slightly with the lowest habitat quality in the hill areas (14.6 %) and the highest in the mountain areas (18.8 %) (Figure 24). Looking at different indicator species groups, except for beetles, habitat quality is always lowest in the lowlands and highest in the mountain areas. Regional differences are smallest for spiders and beetles, whereas they are highest for amphibians and butterflies.

AHQ on organic farms is generally higher and more variable than on conventional farms. However, regional differences generally reveal the same patterns. The lowest AHQ is achieved in hill regions (18.9 %). A higher AHQ was modelled for the lowlands (20.4 %), whereas an uppermost AHQ can be found on organic farms in mountain areas (30.1 %).

Average habitat quality is higher on organic farms compared to their regional conventional counterparts, by about 4 % in the lowlands and hills to 11.4 % in the mountain regions, compared to the 100 % benchmark. There is a similar pattern for all species groups, showing the largest differences between the farming systems in the mountain regions. Relative to conventional habitat quality the improvements on organic farms are 10 to 30 % and up to 60 % in the mountain areas. The particularly high differences in habitat quality between the farming systems were associated primarily with the pronounced differences in uptake levels for ecological compensation areas (see Figure 20, page 171).



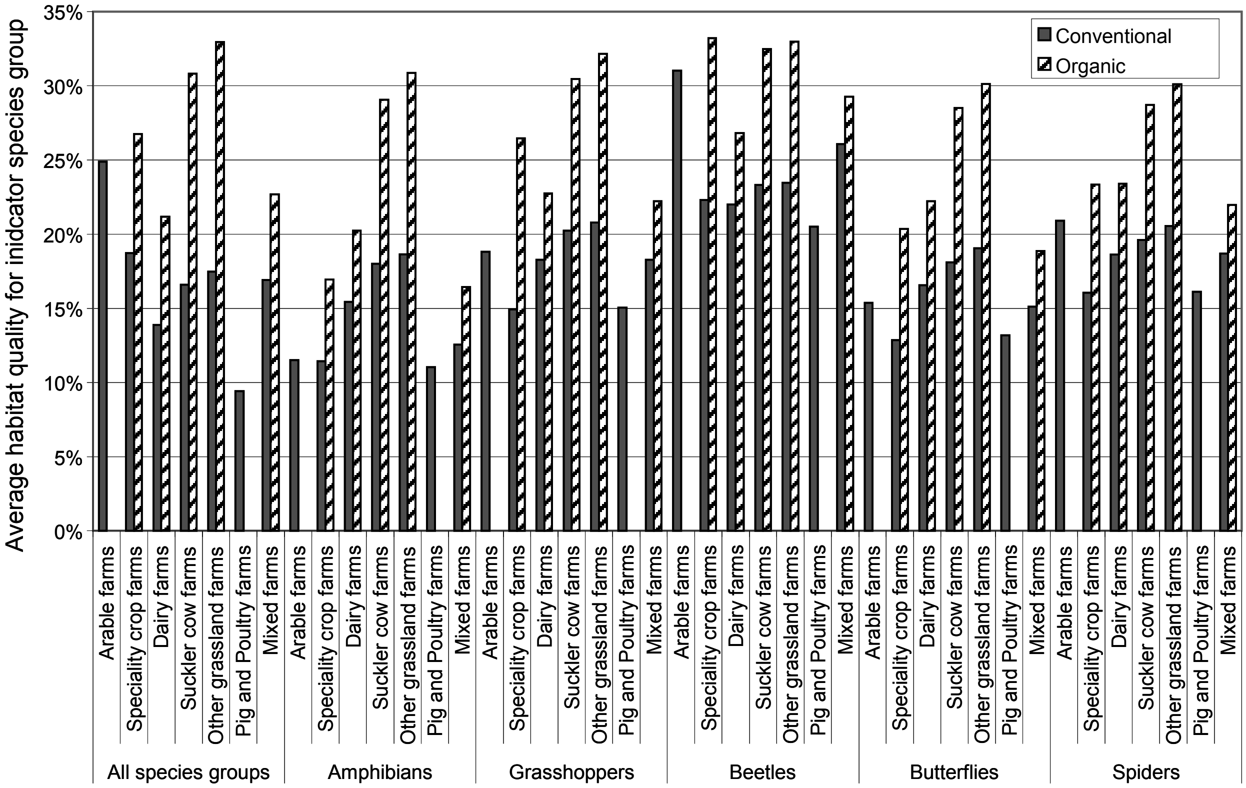
Source: own calculations based on Swiss FADN and SALCA data

Figure 24 Average habitat quality per ha on conventional and organic farms by region (2006/07)

Comparison by farm type

As for the conventional farm types, AHQ varies between 9 % on pig and poultry farms, 14 % on dairy farms and over 17 % on suckler cow, other grassland, and mixed farms. The AHQ is highest on speciality crop farms (19 %) and arable crop farms (25 %).

On organic farm types, AHQ is significantly higher overall, with the highest scores achieved for suckler cow farms (31 %) and other grassland farms (33 %). An AHQ of less than 30 % is reached on speciality crop farms (27 %) mixed farms (23 %) and dairy farms (21 %).



Source: own calculations based on Swiss FADN and SALCA data

Figure 25 Average habitat quality per ha on conventional and organic farms by farm type (2006/07)

Table 37 shows the relative differences in habitat quality between organic and conventional farm groups. The total relative difference amounts to 55 %, while the largest relative difference among the regions could be observed for the mountain areas (61 %), leaving the lowlands at 23 % and the hill regions at 29 % lagging quite far behind. As for the farm types, the largest relative difference to the respective conventional farm group was modelled for suckler cow farms (86 %), while dairy farms differ by 53 % and mixed farms by 34 %.

Table 37 Average habitat quality of conventional and organic farms by region and farm type (2006/07)

Indicator	Unit	Low-lands	Hill	Moun-tain	Dairy farms	Suckler cow farms	Mixed farms	Total farms
Average habitat quality on conventional farms	% of max.	8.03	11.59	19.08	13.86	16.59	16.91	16.55
Average habitat quality on organic farms	% of max.	12.52	14.94	25.59	21.17	30.81	22.68	25.66
Relative difference between farming systems*	%	55.97	28.94	34.12	52.70	85.69	34.11	55.03

max. = maximally achievable score

Source: own calculations based on Swiss FADN and SALCA data

* with conventional farms = 100 %

Table 38 shows the AHQ for individual groups of species. The model revealed a relatively stable difference among most species groups for each farm group. An exception to this should be noted for weeds on arable land and weeds on grassland, since the share of arable land and grassland differs significantly between organic and conventional farms in each group. As the only habitat conductive to arable weeds is arable land, the overall habitat quality on organic farms is lower. However, looking at average habitat quality on arable land, habitat quality for arable weeds would be higher on organic than on conventional farms.

Other species groups

Comparison by region

For the other indicators, organic lowland farms differ from their conventional equivalent by 0 (snails) to 36 % (birds). The different species-specific indicators in the hill regions differ by 7 % for small mammals up to more than 40 % for species with high ecological requirements such as amphibians (43 %) and butterflies (40 %). In mountain areas, the differences for land cultivated under different farming systems are even greater, ranging from 1 % for snails to over 50 % for amphibians (63 %), butterflies (52 %) and birds (66 %).

Comparison by farm type

Differences in habitat quality on organic dairy farms compared to conventional dairy farms range from 8 % for small mammals and snails to over 30 % for amphibians, butterflies, birds and spiders.

Differences for suckler cow farms are most prominent among the farm types analysed, ranging from 15 % for snails to 61 % for amphibians, 51 % for grasshoppers, 57 % for butterflies, 51 % for birds and 52 % for spiders.

Habitat quality for different species groups differs on organic mixed farms by 3 % for snails to 47 % for birds between conventional and organic farming systems. For the total of all farms, habitat quality for amphibians (25 %), grasshoppers (23 %), butterflies (23 %), birds (51 %), bees (23 %), and spiders (23 %) differs most in relative terms. Medium-range relative differences were modelled for beetles (14 %), small mammals (12 %) and snails (7 %). Weeds were predominantly influenced by the abundance of the respective general habitat (grassland or arable land), while the relative differences between the different grassland types were of secondary importance for the average score.

Table 38 Relative difference of average habitat quality for 11 indicator species between conventional and organic farms by region and farm type (2006/07)

Indicator species group	Unit	Lowlands	Hill	Mountain	Dairy farms	Suckler cow farms	Mixed farms	Total farms
All species	%*	23.18	29.06	60.52	52.70	85.69	34.11	55.03
Amphibians (red list)	%*	24.82	17.63	42.83	31.04	61.39	30.86	63.27
Grasshoppers (red list)	%*	23.20	17.36	35.22	24.54	50.59	21.71	41.72
Beetles (red list)	%*	13.60	10.79	32.76	21.86	39.25	12.30	21.92
Butterflies (red list)	%*	23.49	26.44	39.63	34.17	57.41	24.77	52.12
Spiders (red list)	%*	16.62	14.72	34.88	25.54	46.38	17.45	34.87
Arable weeds	%*	-11.95	-46.20	-79.13	-39.98	-52.20	-3.85	-61.98
Grassland weeds	%*	45.66	9.95	35.27	15.29	35.27	21.86	42.97
Small mammals	%*	12.12	9.26	6.83	7.58	16.34	15.39	29.61
Birds	%*	51.14	35.66	34.95	36.37	51.46	46.89	65.64
Amphibians	%*	15.89	14.26	28.99	21.11	44.56	21.02	43.38
Wild bees	%*	23.35	16.32	27.37	19.41	43.45	25.12	48.20
Grasshoppers	%*	22.84	17.60	34.24	24.37	50.19	22.18	43.04
Beetles	%*	12.94	11.73	32.79	22.97	40.33	12.60	22.79
Butterflies	%*	21.98	23.74	37.40	31.04	54.70	23.08	50.02
Spiders	%*	20.81	20.71	40.23	32.23	52.00	22.04	41.69
Snails	%*	7.33	0.47	16.80	7.97	14.64	2.73	1.29

* relative difference in AHQ on organic farms with conventional farms = 100 %

Source: own calculations based on Swiss FADN and SALCA data

7.2.6 Eutrophication

The indicators for eutrophication cover all relevant eutrophication substances for both nitrogen and phosphorus. They were divided into nitrogen compounds (nitrate (NO_3^-, NH_3^+, and other) and phosphorus in the form of phosphoric acid (P_2O_5). First, the values modelled for nitrogen eutrophication are presented. Second, the phosphorus-related substances are described. Within each section (nitrogen and phosphorus), total systems are compared first across all farms, *i.e.* without regional or farm-type stratification. Then, regional differences followed by farm-type related differences are presented. General differences between the groups are illustrated for both regional and farm-type comparisons, before the farming systems are compared with each other.

Nitrogen eutrophication

The model shows that in total 80 kg N-eq per ha are emitted from the average conventional farm into sensitive ecosystems, while about 50 kg N-eq per ha are emitted from organic farming systems (Figure 26).

Comparison by region

The highest eutrophication rates per ha occur on conventional farms in the lowlands (104 kg N-eq per ha) and 95 kg N-eq per ha for organic farming systems. In the hill regions, the highest differences between the farming systems occur, with 26 kg N-eq compared to 10 kg respectively in lowlands and hills. On average 86 kg N-eq per ha are emitted from conventional farms, while organic farms emit 60 kg. In the mountain areas 42 kg N-eq per ha are emitted by conventional farms and 32 kg N-eq per ha on organic farms.

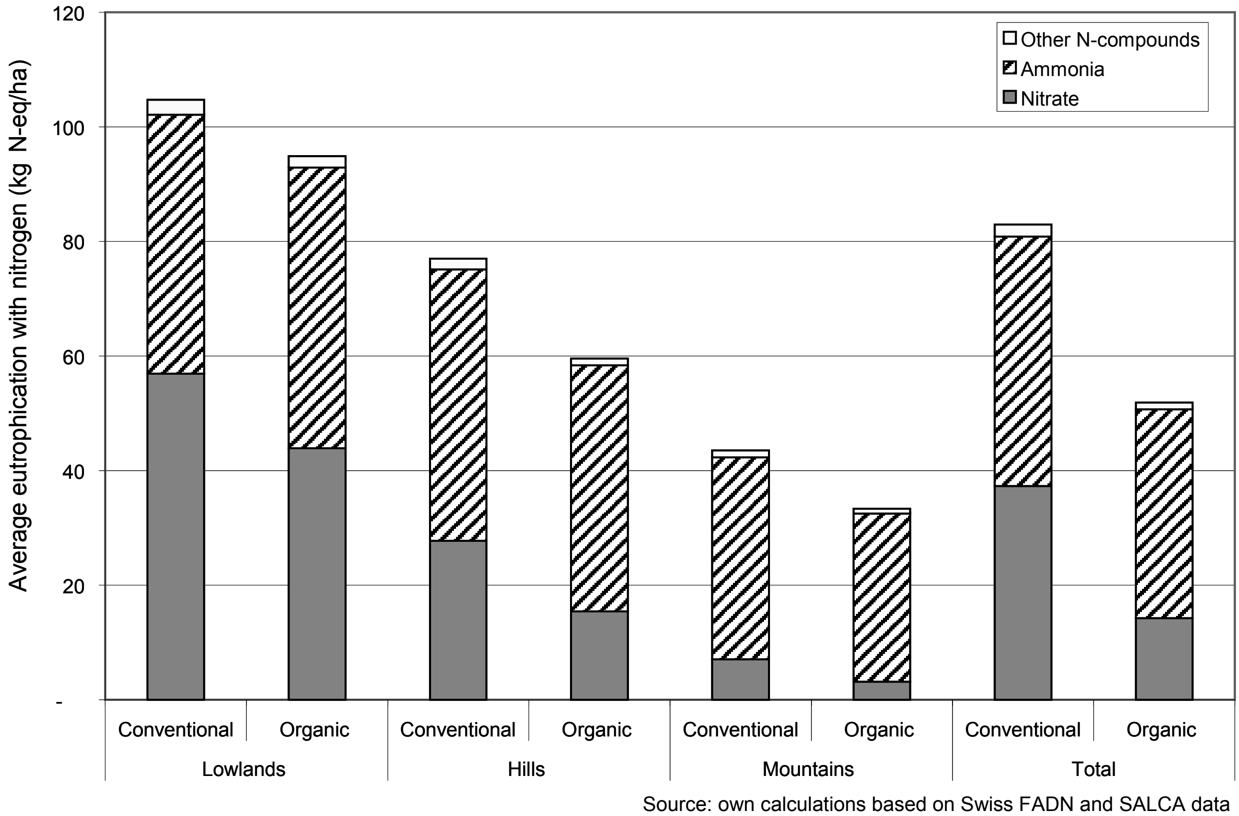


Figure 26 Nitrogen eutrophication per ha on conventional and organic farms by region (2006/07)

Table 39 shows the composition of nitrogen eutrophication rates. The most significant N-compounds are NO_3 and NH_3. NH_3 is found in substantial amounts across all regions and displays low differences between the farming systems. Of note here is a higher modelled NH_3

eutrophication on organic farms in the lowlands (49 kg N-eq as against 45 kg N-eq on conventional farms). NH_3 emissions in hill areas are in a similar range on organic farms (43 kg N-eq instead of 47 kg N-eq on conventional farms). In mountain areas the NH_3 emissions are lower, with 35 kg N-eq for conventional farms and 29 kg N-eq for organic farms. NO_3 emissions are highly regionally dependent, with 57 kg N-eq in the lowlands (on organic farms 44 kg N-eq), 28 kg N-eq in the hill areas (15 kg N-eq on organic farms) and 7 kg N-eq in the mountain areas (3 kg N-eq on organic farms). Other N-components play a minor role of 1 to 3 kg N-eq per ha.

Table 39 Eutrophication per ha on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountains		Total	
		CON	ORG	CON	ORG	CON	ORG	CON	ORG
Total eutrophication	**kg N-eq / ha**	111.2	**99.3**	85.9	**66.9**	52.4	**40.9**	90.6	**58.7**
N-eutrophication	**kg N-eq / ha**	104.7	**94.9**	77.0	**59.6**	43.6	**33.4**	82.9	**51.8**
NO_3-eutrophication	**kg N-eq / ha**	56.9	**43.9**	27.7	**15.4**	7.1	**3.2**	37.4	**14.3**
NH_3-eutrophication	**kg N-eq / ha**	45.2	**49.0**	47.4	**43.0**	35.2	**29.4**	43.5	**36.4**
Other N-eutrophication	**kg N-eq / ha**	2.6	**2.0**	1.8	**1.2**	1.3	**0.8**	2.1	**1.2**
P-eutrophication	**kg P-eq / ha**	6.5	**4.4**	8.9	**7.3**	8.9	**7.5**	7.7	**6.8**

CON = Conventional farms; ORG = Organic farms; Source: own calculations based on Swiss FADN and SALCA data

Comparison by farm type

A comparison of farm types reveals large differences in total eutrophication. The highest eutrophication rates, at 80 to 110 kg N-eq per ha, have been modelled for arable, pig and poultry, specialty crops, and mixed farms. Significantly lower eutrophication rates are estimated for dairy, suckler cow and other grassland farms (Figure 27).

The farm types differ significantly regarding the role of the substances that cause N eutrophication. While 80 % of the eutrophication on arable farms is caused by NO_3, the NH_3 eutrophication rates on pig and poultry farms make up about 80 % of total eutrophication. Nitrogen eutrophication from speciality crop farms is also dominated by nitrate eutrophication, while ammonia eutrophication accounts for 22 kg N-eq (conventional) and 30kg N-eq per ha (organic). Farms dominated by roughage-consuming livestock, *i.e.* dairy, suckler cow and other grassland farms, have only 3 to 21 kg nitrate-N eutrophication per ha, while ammonia eutrophication for these farm types amounts to 30 to 45 kg N-eq. On all farm types, the amounts of substances other than nitrate and ammonia are minimal at only 1 to 3 kg N-eq per ha (Table 40).

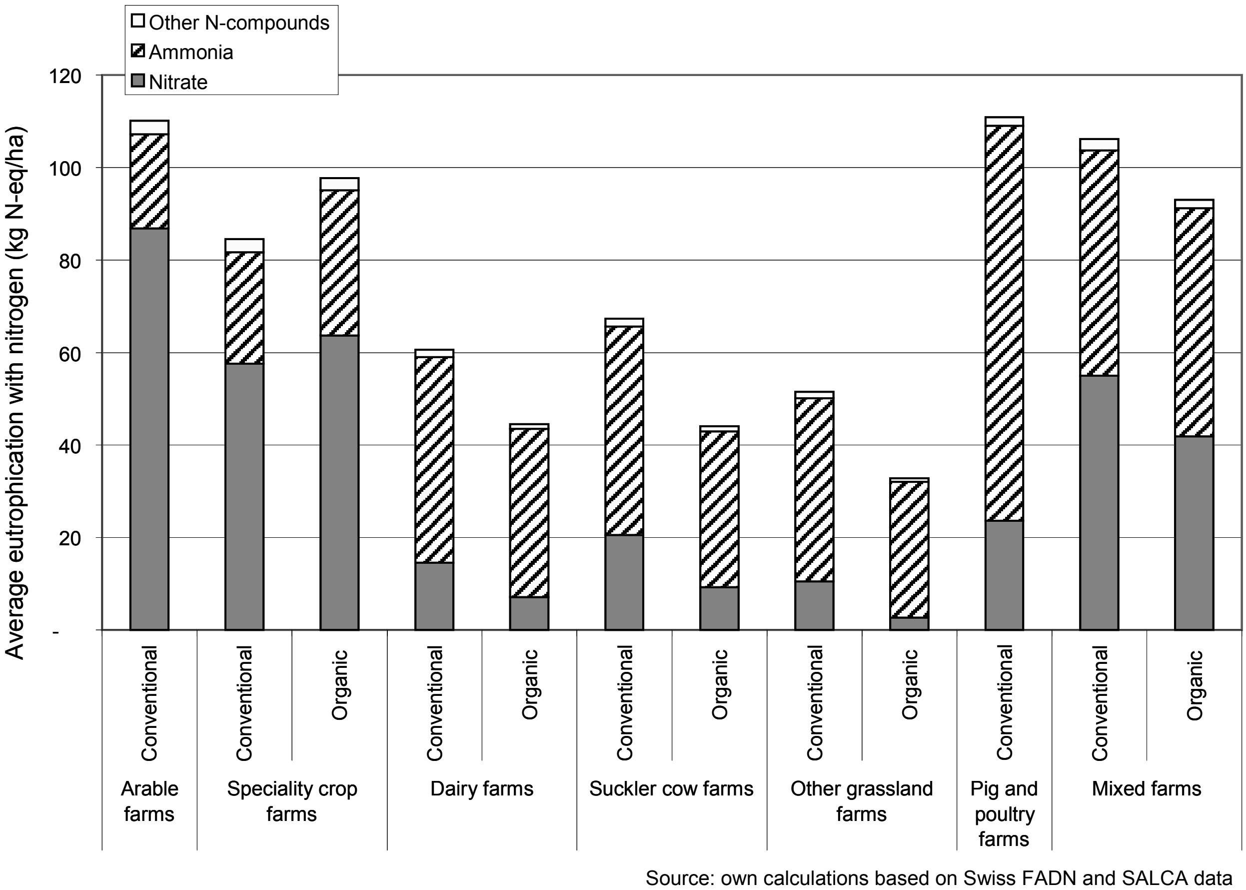


Figure 27 Nitrogen eutrophication per ha on conventional and organic farms by farm type (2006/07)

Table 40 Eutrophication per ha on conventional and organic farms by farm type (2006/07)

Indicator	Unit	Arable farms	Speciality crop farms		Dairy farms		Suckler cow farms		Other grassland farms		Pig / Poultry	Mixed farms	
		CON	CON	ORG	CON	ORG	CON	ORG	CON	ORG	CON	CON	ORG
Total eutrophication	**kg N-eq/ha**	116.0	90.0	**101.8**	69.2	**51.6**	75.6	**51.2**	60.1	**40.4**	119.1	113.5	**98.4**
N-eutrophication	**kg N-eq/ha**	110.1	84.6	**97.7**	60.6	**44.5**	67.3	**44.0**	51.5	**32.8**	110.9	106.2	**93.0**
NO_3-eutrophication	**kg N-eq/ha**	86.8	57.6	**63.7**	14.5	**7.1**	20.6	**9.2**	10.5	**2.7**	23.6	55.0	**41.9**
NH_3-eutrophication	**kg N-eq/ha**	20.4	24.1	**31.4**	44.5	**36.4**	45.1	**33.8**	39.6	**29.3**	85.4	48.7	**49.4**
Other N-eutrophication	**kg N-eq/ha**	2.9	2.9	**2.6**	1.6	**1.0**	1.7	**1.0**	1.4	**0.8**	1.9	2.5	**1.8**
P-eutrophication	**kg P-eq/ha**	5.9	5.4	**4.1**	8.6	**7.2**	8.3	**7.2**	8.6	**7.6**	8.2	7.3	**5.3**

CON = Conventional farms; ORG = Organic farms — Source: own calculations based on Swiss FADN and SALCA data

For each farm type, the organic farms cause less eutrophication per ha with nitrogen. The differences between organic and conventional farming systems are non-uniform for NH_3. Organic speciality crop farms show 31 kg NH_3-N eutrophication instead of 24 kg on conventional farms. Organic and conventional mixed farms have nearly the same NH_3 emissions per ha. Organic dairy, suckler cow and other grassland farms tend to have lower NH_3 emissions per ha.

Nitrate eutrophication is uniformly lower for all farm types and regions, except for speciality crop farms, where slightly higher nitrate emissions were modelled (64 kg NO_3-N as against 58 kg on conventional farms).

Phosphorus eutrophication

In terms of P eutrophication a different picture can be drawn. In total numbers, P eutrophication per ha is less severe than N eutrophication. On average, conventional Swiss farms an estimated average of 7.5 kg P-eq/ha are emitted, whereas organic farms emit 6.8 kg P-eq/ha.

Comparison by region

A regional comparison reveals that the highest P eutrophication rates can be found in hill and mountain areas (Figure 28). In both regions, P eutrophication averages around 8.5 to 9 kg P-eq for conventional farms and 7 to 7.5 kg for organic farms. However, in the lowlands P eutrophication is modelled to be much lower with 6.5 kg per ha on conventionally cultivated land and 4.2 kg P-eq on organically cultivated land (Table 39).

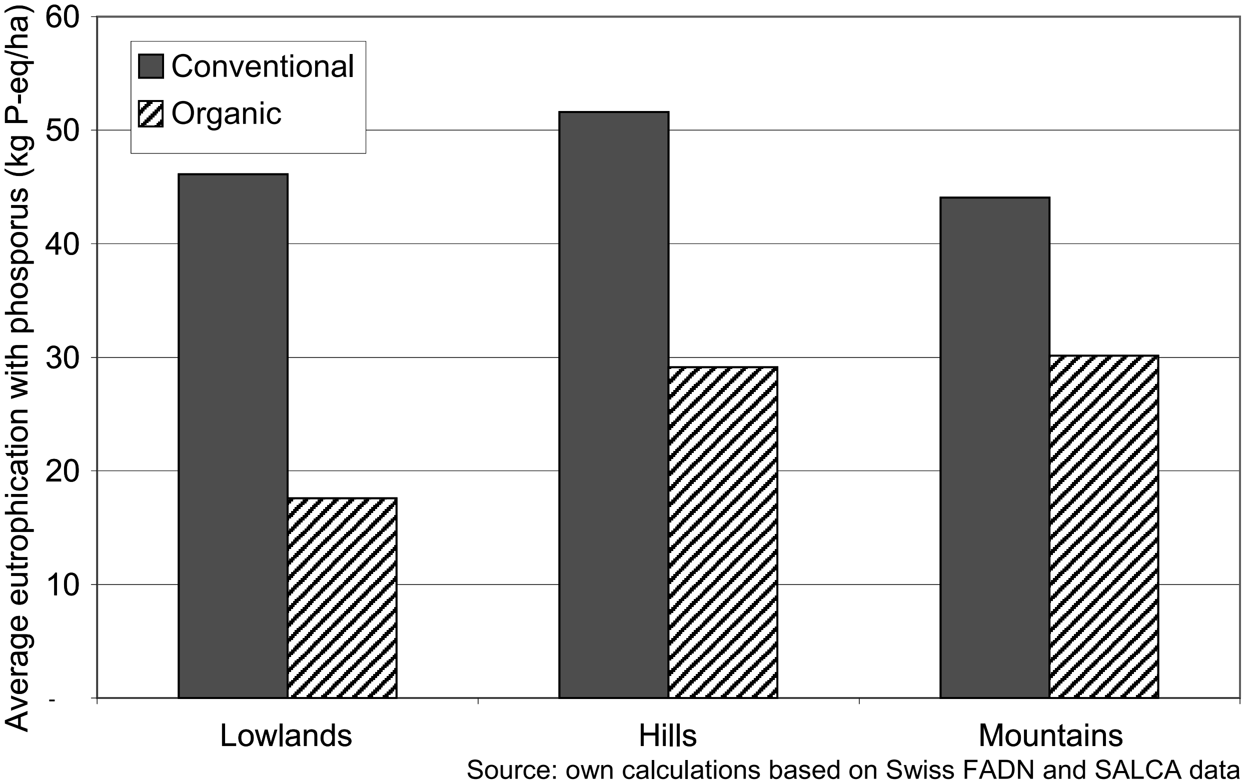
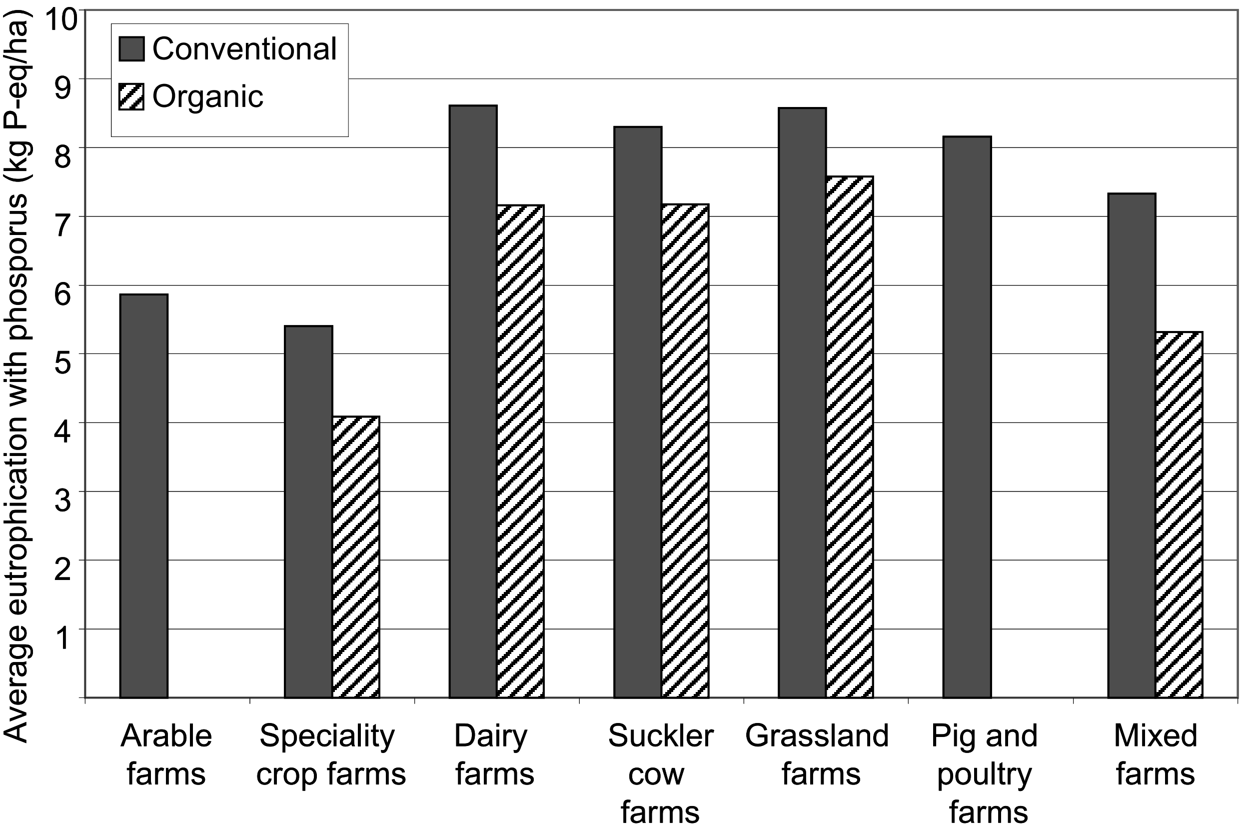


Figure 28 Phosphorus eutrophication on conventional and organic farms by region (2006/07)

Comparison by farm type

From the farm-type perspective, dairy, suckler cow, other grassland farms, and pig and poultry farms account for the highest phosphorus eutrophication rates per ha with more than 8 kg P-eq/ha (Figure 29). Mixed farms show lower eutrophication rates around 7 kg P-eq/ha, while arable and speciality crop farms have the lowest phosphorus eutrophication rates at less than 5 kg. For all farm types, organic farms have around 1 to 2 kg P-eq/ha lower eutrophication rates than their conventional counterparts (Table 40).



Source: own calculations based on Swiss FADN and SALCA data

Figure 29 Phosphorus eutrophication on conventional and organic farms by farm type (2006/07)

To sum up, both absolute and relative nitrogen and phosphorus eutrophication rates differ among regions and farm types. With regard to total farming systems, regions and farm types, all indicators for eutrophication were higher on conventional farms compared to organic. The only exception is NH_3 eutrophication in the lowlands, which is slightly higher on organic farms (49 kg N-eq/ha) than on conventional (45 kg N-eq/ha). Nitrogen eutrophication accounts for the biggest share in total eutrophication, while phosphorus eutrophication contributes only a minor share. Table 41 summarises the model results for total eutrophication, showing generally lower eutrophication rates per ha on organic farms. Eutrophication levels range from 11 (lowlands) to 32 % (suckler cow farms) for total eutrophication and from 9 (lowlands) to 12 % (suckler cow farms) for nitrogen eutrophication. The highest relative difference among farm types was modelled for mixed farms, while highest relative differences according to region occur in the lowland regions.

Table 41 Relative difference in eutrophication per ha between conventional and organic farms by region and farm type (2006/07)

Indicator species group	Unit*	Lowlands	Hill	Mountain	Dairy farms	Suckler cow farms	Mixed farms	Total farms
Total eutrophication	%	10.73	22.07	21.99	25.37	32.34	13.36	35.26
N-eutrophication	%	9.36	22.60	23.41	26.58	34.65	12.39	37.49
NO_3-eutrophication	%	22.80	44.33	55.60	51.21	55.20	23.84	61.78
NH_3-eutrophication	%	-8.35	9.40	16.58	18.21	25.05	-1.32	16.32
Other N-eutrophication	%	23.22	35.14	32.85	35.85	40.18	27.75	44.31
P-eutrophication	%	32.71	17.49	15.00	16.86	13.53	27.44	11.22

* positive values indicate a higher eutrophication of conventioal farms, negative values indicate a higher eutrophication on organic farms

Source: own calculations based on Swiss FADN and SALCA data

7.2.7 Public expenditure

Sections 7.2.4 to 7.2.6 have presented the model results for three land-related environmental indicators. This section now deals with differences between organic and conventional farms in terms of public expenditure. Both direct payments and policy-related transaction costs (PRTC) were included. First, the distribution of total direct payments among policy measures is analysed. Second, the public-expenditure indicators, which include total, farm level and public PRTC as well as average transfer efficiency, are presented for each farm group.

Distribution of direct payments

Table 42 shows the receipt of direct payments on average conventional and organic farms in different regions. In general, direct payments for organic farms are higher (64.6 kCHF) than for conventional farms (50.1 kCHF). Organic farms receive both more general direct payments (47.7 kCHF compared to 38.9 kCHF on conventional farms) and more ecological direct payments (12.4 kCHF compared to 7.8 kCHF on conventional farms) on average. Differences regarding general direct payments can be ascribed to higher RGVE, TEP and hillside contributions. The difference in ecological direct payments can be attributed fully to the additional OFASP received by organic farms (5.1 kCHF), while ECA, BTS and RAUS payments are about equally high in both farming systems.

Table 42 Distribution of direct payments on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountains		Total	
		CON	ORG	CON	ORG	CON	ORG	CON	ORG
Total direct payments	**kCHF/farm**	45.74	**52.00**	49.05	**57.98**	60.72	**71.60**	50.11	**64.56**
General direct payments	**kCHF/farm**	33.55	**31.42**	38.35	**41.65**	50.97	**55.95**	38.90	**47.74**
Area payments	**kCHF/farm**	28.46	**24.98**	23.43	**22.26**	22.77	**24.77**	25.76	**24.26**
RGVE payments	**kCHF/farm**	4.38	**6.14**	6.25	**9.13**	10.11	**11.43**	6.22	**9.83**
TEP payments	**kCHF/farm**	0.37	**0.19**	5.96	**7.09**	13.82	**14.76**	5.02	**10.07**
Hillside payments	**kCHF/farm**	0.34	**0.11**	2.71	**3.17**	4.27	**4.99**	1.90	**3.58**
Ecological direct payments	**kCHF/farm**	8.68	**17.99**	8.03	**12.85**	5.44	**10.26**	7.75	**12.43**
ECA payments	**kCHF/farm**	3.07	**3.49**	2.22	**2.46**	1.30	**1.82**	2.43	**2.30**
Extenso payments	**kCHF/farm**	0.95	**1.34**	0.59	**0.30**	0.05	**0.01**	0.65	**0.35**
OFSAP	**kCHF/farm**	0.01	**8.23**	0.03	**4.50**	0.09	**4.11**	0.03	**5.05**
BTS payments	**kCHF/farm**	1.51	**1.36**	1.33	**1.37**	0.63	**0.76**	1.26	**1.02**
RAUS payments	**kCHF/farm**	3.15	**3.58**	3.85	**4.22**	3.37	**3.56**	3.39	**3.71**
Crop-specific payments	**kCHF/farm**	1.85	**0.12**	0.31	**0.03**	0.00	**0.01**	1.00	**0.03**
Payments for alpine grazing	**kCHF/farm**	0.20	**-**	0.52	**0.81**	1.91	**2.71**	0.68	**1.73**
Other payments	**kCHF/farm**	1.45	**2.46**	1.84	**2.64**	2.39	**2.69**	1.78	**2.63**

RGVE = Roughage-consuming livestock
TEP = Animal husbandry under adverse conditions
ECA = Ecological compensation areas
OFASP = Organic farming area support payments
BTS = Particular animal-friendly stabling
RAUS = Livestock with outdoor exercise

Source: own calculations based on Swiss FADN

Comparison by region

Direct payments increase with farm altitude, from 45.7 on conventional farms in the lowlands to 60.7 kCHF on conventional farms in the mountain areas. Organic farms receive only slightly higher payments, with 52 kCHF in the lowlands, 58 kCHF in the hill areas and 71.6 kCHF in the mountain areas. These differences can be attributed most notably to higher receipts of ecological direct payments, while the general direct payments differ only slightly. Area payments are higher on conventional farms in the lowlands and hill regions compared to organic farms. In the mountain regions, area payments are slightly higher on organic farms. RGVE, TEP and hillside contributions are generally higher on organic farms, while the most substantial differences can be found in the lowlands and hill regions[69]. Ecological direct payments are roughly by 10 kCHF higher in the lowlands and approximately 5 kCHF higher in the hill and mountain regions. These differences are only slightly greater than the OFASP

[69] Except in the lowlands, where the absolute levels of TEP and hillside payments are marginal

received by organic farms. ECA payments are slightly higher in each region, while extenso payments are higher in the lowlands and lower in the hill and mountain areas. BTS payments are almost the same for both farming systems, while RAUS payments are higher throughout on organic farms.

Figure 38 (see Annex C) illustrates the relative shares of the direct payments graphically.

Comparison by farm type

Table 43 describes the distribution of direct payments on average organic and conventional farm types. The variation between the farm types is partly attributable to the different UAA among farm types, ranging from only 26.9 kCHF per speciality crop farm to 65.2 kCHF on suckler cow farms. An average amount of direct payments of approximately 50 kCHF goes to arable, mixed dairy, pig and poultry farms and other grassland farms.

Organic farms of all farm types receive higher total direct payments. The average differences are about 10 kCHF, with lowest differences found on suckler cow farms and the biggest differences on speciality crop and other grassland farms. While general direct payments are only slightly higher on organic farms, there are substantial differences in ecological direct payments between the farming systems. Generally, organic farms receive between 3 kCHF (suckler cow farms) and 9 kCHF (mixed farms) higher direct payments than their conventional equivalents. In relative terms, ecological direct payments are higher on organic farms than on conventional farms by 40 % (suckler cow farms), 80 % on mixed farms and 200 % on speciality crop farms. ECA payments are the same on suckler cow farms but higher on the other farm types. Extenso payments are mostly higher on conventional farm types, due to their higher share of arable land. Mixed farms and speciality crop farms are an exception, with organic farms receiving higher extenso payments than conventional equivalents.

Table 43 Distribution of direct payments on conventional and organic farms by farm type (2006/07)

Indicator	Unit	Arable	Speciality crops		Dairy farms		Suckler cow farms		Other grassland farms		Pig/ Poultry	Mixed farms	
		CON	CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	CON	**ORG**
Total direct payments	**kCHF/farm**	51.38	26.94	**40.04**	49.73	**60.26**	65.22	**70.94**	52.30	**74.27**	48.07	52.15	**63.32**
General direct payments	**kCHF/farm**	37.59	21.23	**23.06**	39.66	**44.94**	53.26	**55.17**	44.47	**57.99**	31.45	38.87	**40.13**
Area payments	**kCHF/farm**	35.70	19.94	**23.06**	22.78	**24.35**	23.69	**23.01**	18.87	**23.96**	21.19	30.35	**26.48**
RGVE payments	**kCHF/farm**	1.60	0.72	**-**	5.36	**6.21**	19.41	**16.29**	12.54	**14.24**	5.65	5.25	**8.46**
TEP payments	**kCHF/farm**	0.06	0.02	**-**	8.64	**10.61**	7.34	**11.35**	9.46	**15.23**	3.27	2.27	**3.65**
Hillside payments	**kCHF/farm**	0.23	0.55	**-**	2.88	**3.77**	2.82	**4.51**	3.60	**4.56**	1.34	1.01	**1.54**
Ecological direct payments	**kCHF/farm**	6.98	3.52	**16.57**	6.60	**11.24**	8.31	**11.20**	4.69	**10.34**	15.13	10.05	**18.80**
ECA payments	**kCHF/farm**	3.73	2.39	**3.40**	1.84	**1.92**	2.15	**2.14**	1.77	**2.26**	1.80	2.97	**3.37**
Extenso payments	**kCHF/farm**	1.98	0.87	**1.24**	0.16	**0.11**	0.20	**0.12**	0.14	**-**	0.20	1.00	**1.49**
OFSAP	**kCHF/farm**	-	-	**11.79**	0.04	**4.24**	0.30	**4.16**	0.03	**4.14**	0.03	0.02	**7.86**
BTS payments	**kCHF/farm**	0.50	0.12	**-**	0.79	**0.85**	1.65	**1.37**	0.26	**0.57**	5.78	1.92	**1.75**
RAUS payments	**kCHF/farm**	0.77	0.14	**0.14**	3.77	**4.11**	4.00	**3.41**	2.49	**3.37**	7.32	4.15	**4.31**
Crop-specific payments	**kCHF/farm**	5.62	1.54	**-**	0.02	**0.01**	0.09	**0.01**	0.03	**-**	-	1.25	**0.17**
Payments for alpine grazing	**kCHF/farm**	0.12	-	**-**	1.47	**1.40**	0.66	**1.83**	0.81	**3.54**	-	0.29	**1.00**
Other payments	**kCHF/farm**	1.06	0.65	**0.42**	1.98	**2.66**	2.90	**2.72**	2.30	**2.40**	1.50	1.68	**3.24**

RGVE = Roughage-consuming livestock
TEP = Animal husbandry under adverse conditions
ECA = Ecological compensation areas
OFASP = Organic farming area support payments
BTS = Particular animal-friendly stabling
RAUS = Livestock with outdoor exercise

Source: own calculations based on Swiss FADN

Public expenditure indicators

Table 44 shows the total public expenditure distributed among the policy measures for organic and conventional farms by region. Average public expenditure across all farms amounts to 2,579 CHF per ha for conventional farms and 3,265 CHF/ha for organic farms.

Comparison by region

Regional differences are considerable, with 2,254 CHF/ha for conventional farms in the lowlands (2,791 CHF/ha for organic farms), 2,622 CHF/ha in the hill areas (organic farms 3,152 CHF/ha), and 3,173 CHF in the mountain areas (3,479 CHF/ha for organic farms).

Average public PRTC are low in mountain regions (31 CHF/ha) and higher in the lowlands (55 CHF/ha). Farm-level PRTC are higher and vary more substantially, at 48 CHF/ha (mountains conventional) and 113 CHF/ha (lowlands organic). Transfer efficiency, *i.e.* share of total transaction costs from total public expenditure, ranges from 94 to 98 %.

Table 44 Public expenditure on conventional and organic farms by region (2006/07)

Indicator	Unit	Lowlands		Hills		Mountain		Total	
		CON	**ORG**	CON	**ORG**	CON	**ORG**	CON	**ORG**
Total public expenditure	**kCHF/ha**	2.25	**2.79**	2.66	**3.15**	3.17	**3.48**	2.58	**3.26**
Total direct payments	**kCHF/ha**	2.20	**2.74**	2.62	**3.11**	3.14	**3.44**	2.53	**3.22**
Total PRTC	**kCHF/ha**	0.14	**0.17**	0.11	**0.11**	0.08	**0.10**	0.12	**0.12**
Public PRTC	**kCHF/ha**	0.05	**0.05**	0.04	**0.04**	0.03	**0.04**	0.04	**0.04**
Farm-level PRTC	**kCHF/ha**	0.09	**0.11**	0.07	**0.08**	0.05	**0.06**	0.08	**0.08**
Transfer efficiency	**%**	93.59	**94.00**	95.89	**96.35**	97.51	**97.19**	95.36	**96.45**

CON = Conventional farms; ORG = Organic farms;
PRTC = Policy-related transaction costs Source: own calculations based on FADN and FSS

Comparison by farm type

Conventional farm types show an even higher variation in public expenditure parameters than the regions; as shown in Table 45. The lowest public expenditure per ha is recorded for arable and speciality crop farms (2,079 to 2,101 CHF/ha for conventional farms). Mixed farms entail only slightly higher public expenditure, at 2,364 CHF/ha. Higher public expenditure per ha is set off by farm types with higher stocking rates per ha, ranging from 2,632 CHF/ha (dairy farms) to 3,309 CHF/ha (other grassland farms), as shown in Table 45.

Public PRTC vary between 32 CHF/ha (other grassland farms) and 73 CHF/ha (pig and poultry farms), while farm-level PRTC range from 48 to 136 CHF/ha respectively. Transfer efficiency varies slightly between 93 % (conventional other grassland farms) and 98 % (pig and poultry farms).

Organic farms receive higher public expenditure on average, regardless of farm type. The most substantial differences were found for mixed farms (738 CHF/ha). Total transaction costs are higher on organic farms. While farm-level PRTC are most significant, public PRTC are only marginally different between the farming systems.

Table 45 Public expenditure on conventional and organic farms by farm type (2006/07)

Indicator	Unit	Arable	Speciality crops		Dairy farms		Suckler cow farms		Other grassland farms		Pig/ Poultry	Mixed farms	
		CON	CON	ORG	CON	ORG	CON	ORG	CON	ORG	CON	CON	ORG
Total public expenditure	**kCHF/ha**	2.10	2.08	**2.71**	2.63	**3.08**	3.30	**3.61**	3.31	**3.57**	2.80	2.36	**3.10**
Total direct payments	**kCHF/ha**	2.07	2.04	**2.68**	2.59	**3.04**	3.27	**3.57**	3.28	**3.53**	2.73	2.31	**3.05**
Total PRTC	**kCHF/ha**	0.10	0.09	**0.11**	0.12	**0.12**	0.09	**0.10**	0.08	**0.10**	0.21	0.14	**0.15**
Public PRTC	**kCHF/ha**	0.04	0.03	**0.04**	0.04	**0.04**	0.03	**0.04**	0.03	**0.04**	0.07	0.05	**0.05**
Farm-level PRTC	**kCHF/ha**	0.06	0.05	**0.08**	0.07	**0.08**	0.05	**0.06**	0.05	**0.06**	0.14	0.09	**0.10**
Transfer efficiency	**%**	95.44	95.69	**95.89**	95.62	**96.12**	97.36	**97.25**	97.60	**97.28**	92.57	93.88	**95.11**

CON = Conventional farms; ORG = Organic farms;
PRTC = Policy-related transaction costs

Source: own calculations based on Swiss FADN

Summing up the model results for public expenditure, it can be noted that the relative difference in public expenditure per ha between organic and conventional farm groups ranges from 9 % for suckler cow farms to 31 % for mixed farms. Organic dairy farms receive 17 % higher direct payments than conventional dairy farms. Regionally, differences in public expenditure are lowest in the mountain regions (10 %), medium in the hill areas (19 %) and highest in the lowlands (24 %).

As total PRTC account for less than 10 % of total public expenditure only, relative differences are not decisive for the total results. What is notable, however, are the relative differences between the farm groups, showing slightly lower public PRTC for the hill areas, dairy farms and mixed farms. Total transaction costs are, however, higher for organic farms, as slightly lower PRTC are overcompensated for by the higher farm-level PRTC, which include costs for private organic farming certification bodies. Transfer efficiency is not affected by organic farming, *i.e.* the relation between average payments and average transaction costs per ha is almost equal (Table 46).

Table 46 Difference in public expenditure between conventional and organic farms by region and farm type (2006/07)

Indicator	Unit*	Lowlands	Hill	Mountain	Dairy farms	Suckler cow farms	Mixed farms	Total farms
Total public expenditure	**% of conv.**	23.86	18.36	9.64	16.99	9.41	31.20	26.60
Total direct payments	**% of conv.**	24.32	18.76	9.58	17.36	9.41	32.02	27.22
Total PRTC	**% of conv.**	15.87	4.91	23.72	3.86	14.09	4.92	-3.14
Public PRTC	**% of conv.**	4.47	-7.35	15.53	-5.62	9.46	-4.78	-8.79
Farm-level PRTC	**% of conv.**	22.36	12.10	29.12	9.49	16.97	10.43	0.18
Transfer efficiency	**% of conv.**	0.44	0.49	-0.33	0.51	-0.12	1.31	1.14

CON = Conventional farms; ORG = Organic farms
PRTC = Policy-related transaction costs
* relative difference to conventional farms (=100 %)

Source: own calculations based on FADN and FSS

7.2.8 Cost-effectiveness

As described in Section 6, the relative effects of organic farms compared to conventional farms on the one hand, and the public expenditure on organic farms on the other hand, are related to each other in order to derive the cost-effectiveness of the public money spent. Cost-effectiveness and abatement/provision costs are the two central indicators for describing this relation. The data for these indicators has been obtained in the last sections and is summarised in Table 47.

If these figures are set in relation to each other, the cost-effectiveness ratio and the abatement costs can be derived as shown in Table 48 and Table 49, respectively.

Table 47 Public expenditure and relative environmental effects of organic farming by region and farm type (2006/07)

Indicator	Unit	Lowlands	Hill	Mountain	Dairy farms	Suckler cow farms	Mixed farms	Total farms
Public expenditure	**CHF/ha**	**537.71**	**489.00**	**305.80**	**447.16**	**310.54**	**737.65**	**685.99**
Total energy use	**%**	**55.25**	**40.85**	**33.21**	**30.44**	**35.99**	**48.92**	**54.39**
Habitat quality (all species)	**%**	**23.18**	**29.06**	**60.52**	**52.70**	**85.69**	**34.11**	**55.03**
Amphibians (red list)	%	24.82	17.63	42.83	31.04	61.39	30.86	63.27
Grasshoppers (red list)	%	23.20	17.36	35.22	24.54	50.59	21.71	41.72
Beetles (red list)	%	13.60	10.79	32.76	21.86	39.25	12.30	21.92
Butterflies (red list)	%	23.49	26.44	39.63	34.17	57.41	24.77	52.12
Spiders (red list)	%	16.62	14.72	34.88	25.54	46.38	17.45	34.87
Arable weeds	%	-11.95	-46.20	-79.13	-39.98	-52.20	-3.85	-61.98
Grassland weeds	%	45.66	9.95	35.27	15.29	35.27	21.86	42.97
Small mammals	%	12.12	9.26	6.83	7.58	16.34	15.39	29.61
Birds	%	51.14	35.66	34.95	36.37	51.46	46.89	65.64
Amphibians	%	15.89	14.26	28.99	21.11	44.56	21.02	43.38
Wild bees	%	23.35	16.32	27.37	19.41	43.45	25.12	48.20
Grasshoppers	%	22.84	17.60	34.24	24.37	50.19	22.18	43.04
Beetles	%	12.94	11.73	32.79	22.97	40.33	12.60	22.79
Butterflies	%	21.98	23.74	37.40	31.04	54.70	23.08	50.02
Spiders	%	20.81	20.71	40.23	32.23	52.00	22.04	41.69
Snails	%	7.33	0.47	16.80	7.97	14.64	2.73	1.29
Total eutrophication	**%**	**10.73**	**22.07**	**21.99**	**25.37**	**32.34**	**13.36**	**35.26**
N-eutrophication	%	9.36	22.60	23.41	26.58	34.65	12.39	37.49
NO_3-eutrophication	%	22.80	44.33	55.60	51.21	55.20	23.84	61.78
NH_3-eutrophication	%	-8.35	9.40	16.58	18.21	25.05	-1.32	16.32
Other N-eutrophication	%	23.22	35.14	32.85	35.85	40.18	27.75	44.31
P-eutrophication	%	32.71	17.49	15.00	16.86	13.53	27.44	11.22

Source: own calculations based on Swiss FADN and SALCA data

Cost-effectiveness ratio

Table 48 shows the cost-effectiveness of the policy measures, *i.e.* how much relative improvement could be achieved with an additional 100 CHF per ha, if all costs were related to only one environmental indicator[70]. The first row indicates the absolute additional public expenditure per ha on organic farms compared to conventional counterparts. Across all farms, the additional public expenditure per ha amounts to 686 CHF. This figure varies regionally

[70] Under the assumptions specified in Sections 6.3.9 and 7.1.

from 305.8 CHF/ha in mountain areas, 489 CHF/ha in hill areas to 537.7 CHF in the lowlands. Average per-ha costs in terms of additional public expenditure for organic farms per farm type range from 310.5 CHF for suckler cow farms to 737.7 CHF for mixed farms, while organic dairy farms entail an additional 447.2 CHF per ha.

Table 48 Cost-effectiveness of organic farming by region and farm type, expressed as percentage improvement of the indicator per 100 CHF of public expenditure (2006/07)

Indicator	Unit	Lowlands	Hill	Mountain	Dairy farms	Suckler cow farms	Mixed farms	Total farms
Public expenditure	**CHF/ha**	**537.71**	**489.00**	**305.80**	**447.16**	**310.54**	**737.65**	**685.99**
Total energy use	**%/100CHF***	**10.27**	**8.35**	**10.86**	**6.81**	**11.59**	**6.63**	**7.93**
Habitat quality (all species)	**%/100CHF***	**4.31**	**5.94**	**19.79**	**11.78**	**27.60**	**4.62**	**8.02**
Amphibians (red list)	%/100CHF*	4.62	3.60	14.01	6.94	19.77	4.18	9.22
Grasshoppers (red list)	%/100CHF*	4.32	3.55	11.52	5.49	16.29	2.94	6.08
Beetles (red list)	%/100CHF*	2.53	2.21	10.71	4.89	12.64	1.67	3.20
Butterflies (red list)	%/100CHF*	4.37	5.41	12.96	7.64	18.49	3.36	7.60
Spiders (red list)	%/100CHF*	3.09	3.01	11.41	5.71	14.93	2.37	5.08
Arable weeds	%/100CHF*	-2.22	-9.45	-25.88	-8.94	-16.81	-0.52	-9.04
Grassland weeds	%/100CHF*	8.49	2.03	11.53	3.42	11.36	2.96	6.26
Small mammals	%/100CHF*	2.25	1.89	2.23	1.69	5.26	2.09	4.32
Birds	%/100CHF*	9.51	7.29	11.43	8.13	16.57	6.36	9.57
Amphibians	%/100CHF*	2.95	2.92	9.48	4.72	14.35	2.85	6.32
Wild bees	%/100CHF*	4.34	3.34	8.95	4.34	13.99	3.41	7.03
Grasshoppers	%/100CHF*	4.25	3.60	11.20	5.45	16.16	3.01	6.27
Beetles	%/100CHF*	2.41	2.40	10.72	5.14	12.99	1.71	3.32
Butterflies	%/100CHF*	4.09	4.86	12.23	6.94	17.62	3.13	7.29
Spiders	%/100CHF*	3.87	4.24	13.16	7.21	16.74	2.99	6.08
Snails	%/100CHF*	1.36	0.10	5.49	1.78	4.72	0.37	0.19
Total eutrophication	**%/100CHF***	**1.99**	**4.51**	**7.19**	**5.67**	**10.41**	**1.81**	**5.14**
N-eutrophication	%/100CHF*	1.74	4.62	7.66	5.94	11.16	1.68	5.47
NO_3-eutrophication	%/100CHF*	4.24	9.07	18.18	11.45	17.78	3.23	9.01
NH_3-eutrophication	%/100CHF*	-1.55	1.92	5.42	4.07	8.07	-0.18	2.38
Other N-eutrophication	%/100CHF*	4.32	7.19	10.74	8.02	12.94	3.76	6.46
P-eutrophication	%/100CHF*	6.08	3.58	4.91	3.77	4.36	3.72	1.64

*% improvement of the indicator for 100 CHF/ha Source: own calculations based on Swiss FADN and SALCA data

Negative numbers indicate that the additional money spent entailed a negative environmental trend, as the environmental indicator was adversely affected. This is the case for the average habitat quality for arable weeds, if it was related to total UAA rather than to arable land (see Section 7.2.5). Furthermore, NH_3 emissions, if separated from total nitrogen eutrophication, are higher on organic farms in the lowlands and almost equal on mixed farms, resulting also in a negative cost-effectiveness. Apart from these indicators, cost-effectiveness figures are uniformly positive among all regions and farm types, indicating a positive relationship between additional money for organic farms on the one hand and environmental improvement on the other. The higher the percentages, the higher the relative improvement of the environment on organic farms.

Each individual indicator was improved by more than one percent per 100 CHF spent on organic farms, except for habitat quality for snails, which scores less than one percent in the lowlands, mountains, and for mixed farms.

Total energy use per ha was reduced on organic farms by between 6.6 and 11.6 % per 100 CHF spent, depending on region and farm type. Total habitat quality is improved by 4.6 % on mixed farms to 28 % in the mountain regions per 100 CHF public expenditure per ha and year. Total eutrophication was reduced by 1.8 % on mixed farms and by 10.4 % on suckler cow farms.

Abatement and provision cost

As summarised in Table 49, the cost-effectiveness figures can be transformed into abatement/provision costs by taking the reciprocal. Abatement costs express the costs that were spent on achieving a 1 % improvement in the respective environmental indicator. It shows negative values for habitat quality regarding arable weeds for ammonia eutrophication on mixed farms and lowlands.

With regard to regions, abatement costs for total energy use per ha range from 9.2 CHF/ha to 9.7 CHF/ha and 12 CHF/ha for mountains, lowlands, and hills respectively. Regarding farm types, abatement costs range from 8.6 CHF/ha on suckler cow farms to 15.1 CHF/ha on mixed farms, while costs on dairy cow farms are 14.7 CHF/ha.

Provision costs for habitat quality vary markedly among the regions. In particular, low provision costs were calculated for mountain regions (5.9 CHF/ha) and hill regions (8.1 CHF/ha), while in the lowlands provision costs of 18.5 CHF/ha were incurred. Farm-type differences were even higher than regional differences, with 5.6 CHF/ha for mountain regions, 8.1 CHF/ha for hills and 18.5 CHF/ha in the lowlands.

Abatement costs for eutrophication range from 13.9 in the mountain areas to 22.2 CHF/ha in the hill areas and 50.1 CHF/ha in the lowlands. Mixed farms have the highest eutrophication abatement costs at 55.2 CHF/ha. Abatement costs on dairy farms amount to 17.6 CHF/ha followed by 9.6 CHF/ha on suckler cow farms.

Table 49 Abatement and provision costs of organic farming by region and farm type, expressed as CHF/ha for a 1% improvement of the indicator (2006/07)

Indicator	Unit	Lowlands	Hill	Mountain	Dairy farms	Suckler cow farms	Mixed farms	Total farms
Public expenditure	**CHF/ha**	**537.71**	**489.00**	**305.80**	**447.16**	**310.54**	**737.65**	**685.99**
Total energy use	**CHF/1%***	**9.73**	**11.97**	**9.21**	**14.69**	**8.63**	**15.08**	**12.61**
Habitat quality (all species)	**CHF/1%***	**23.20**	**16.83**	**5.05**	**8.49**	**3.62**	**21.63**	**12.47**
Amphibians (red list)	CHF/1%*	21.66	27.74	7.14	14.41	5.06	23.90	10.84
Grasshoppers (red list)	CHF/1%*	23.17	28.17	8.68	18.22	6.14	33.98	16.44
Beetles (red list)	CHF/1%*	39.55	45.30	9.33	20.46	7.91	59.98	31.30
Butterflies (red list)	CHF/1%*	22.89	18.50	7.72	13.09	5.41	29.78	13.16
Spiders (red list)	CHF/1%*	32.36	33.22	8.77	17.51	6.70	42.27	19.67
Arable weeds	CHF/1%*	-45.01	-10.58	-3.86	-11.19	-5.95	-191.44	-11.07
Grassland weeds	CHF/1%*	11.78	49.15	8.67	29.25	8.80	33.75	15.97
Small mammals	CHF/1%*	44.35	52.80	44.78	59.03	19.01	47.93	23.17
Birds	CHF/1%*	10.52	13.71	8.75	12.29	6.03	15.73	10.45
Amphibians	CHF/1%*	33.85	34.30	10.55	21.18	6.97	35.09	15.81
Wild bees	CHF/1%*	23.03	29.97	11.17	23.03	7.15	29.37	14.23
Grasshoppers	CHF/1%*	23.54	27.78	8.93	18.35	6.19	33.25	15.94
Beetles	CHF/1%*	41.55	41.70	9.32	19.46	7.70	58.55	30.11
Butterflies	CHF/1%*	24.46	20.60	8.18	14.41	5.68	31.96	13.72
Spiders	CHF/1%*	25.84	23.61	7.60	13.87	5.97	33.46	16.46
Snails	CHF/1%*	73.34	1,035.06	18.20	56.09	21.21	269.91	533.21
Total eutrophication	**CHF/1%***	**50.13**	**22.16**	**13.91**	**17.63**	**9.60**	**55.22**	**19.45**
N-eutrophication	CHF/1%*	57.43	21.64	13.06	16.82	8.96	59.56	18.30
NO_3-eutrophication	CHF/1%*	23.59	11.03	5.50	8.73	5.63	30.94	11.10
NH_3-eutrophication	CHF/1%*	-64.36	52.04	18.44	24.56	12.40	-560.34	42.04
Other N-eutrophication	CHF/1%*	23.16	13.92	9.31	12.47	7.73	26.58	15.48
P-eutrophication	CHF/1%*	16.44	27.96	20.38	26.53	22.96	26.89	61.12

*CHF/ha*1%improvement of the indicator   Source: own calculations based on Swiss FADN and SALCA data

In summary, organic farming delivered relative improvements in habitat quality most cost-effectively, reductions in fossil energy use less cost-effectively, and reductions in eutrophication least cost-effectively.

The mountain areas and lowlands are least cost-effective for attaining relative improvements in energy use via organic farming. Regarding farm types, the cheapest effects were achieved on suckler cow farms, while dairy and mixed farms are similarly expensive.

Achieving relative improvements in habitat quality due to organic farming was cheapest in mountain regions, slightly more expensive in the hill areas and most expensive in the low-lands. On suckler cow farms, the habitat-quality effects were attained at lowest cost. Effects on dairy farms were achieved at medium cost, while effects on mixed farms were achieved at the highest relative cost.

Regarding relative improvements in eutrophication, the lowest costs were calculated for the mountain regions, medium costs in the hill regions and highest costs in the lowlands. Eutrophication was reduced on suckler cow farms at the lowest cost, while costs for dairy farms were medium and reduction costs for eutrophication on mixed farms were the highest.

7.3 Cost-effectiveness of agri-environmental policy measures

In order to relate the cost-effectiveness of organic farming calculated in Section 7.2 to other policy measures, policy scenarios were calculated using FARMIS. The scenarios consisted solely of abolishing the respective agri-environmental payment, as described in Section 6.3.9[71]. Results are presented, concerning all indicators for

a) all farms, in order to show the total sector-level effect, and

b) the farming systems in order to reveal possible interactions between organic farming and the agri-environmental policy measures.

Finally, farm-group specific differences in cost-effectiveness are presented. The structure of this section follows the same sequence as in Section 7.2. Four scenarios were analysed. Scenario A assumed the abolition of extenso payments. Scenario B assumed the abolition of payments for less intensive meadows, while Scenario C consisted of the abolition of payments for extensive meadows. Finally, in Scenario D all the above mentioned payments were abolished simultaneously.

[71] The cost-effectiveness of OFASP was not calculated by the same procedure because the approach is unable to adequately model conversion to and from organic agriculture.

7.3.1 Farm structure

This section presents the effects of the scenario on crop production. It then shows livestock and labour-related structural indicators.

Crop production

Table 50 presents the structural changes modelled in the Scenarios A, B, C, and D for organic, conventional and all farms. UAA, arable land, grassland, and permanent land do not change substantially. With regard to all farms, Scenario A affects mostly arable land, as with cereals and rape (included in oilseeds) only the relative profitability of arable activities is altered directly. Conversely, Scenarios B and C result primarily in farm responses in terms of grassland activities. Finally, Scenario D, as a combination of both direct payment modifications, affects both arable and grassland activities simultaneously.

Only in Scenario D, where payments for extenso, less intensive and extensive meadows were abolished, does total UAA decrease slightly by 0.9 percent and grassland share fall by 1.3 % compared to the base year.

Concerning arable crops, Scenario A, C and D show noticeable effects. The abolition of extenso payments in Scenarios A and D leads to lower cereal shares (about 2 % for Scenario D), whereas other arable crops off-set this decline. For instance, pulses increase by 4 % in Scenario D. Fallows[72] increase markedly by 7 % (Scenario A), 9 % (Scenario C) and 32 % (Scenario D). This is due to a) increasing relative economic profitability compared to cereals and b) the need for farms to hive off a share of 7 % of ecological compensation areas as for the total UAA. Due to the low absolute share of fallows in total UAA (0.3 %), the relative numbers mean only slight absolute changes.

While in Scenario A grassland activities are affected only to a marginal extent, Scenarios B, C and D lead to a marked decrease in meadows (Scenario B -1.5 %, C: -2.7 %, D: -4.1 %) and an increase in pastures by 4.8 % (Scenario B), 7.0 % (Scenario C), and 9.1 % (Scenario D). Ley shares also slightly increase up to 2.5 % in Scenario D. Permanent crops are fixed in the model and do not change in the scenarios.

[72] Sum of mixed and rotational fallows

Table 50 Relative responses in crop levels of conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
			relative change (%)					relative change (%)					relative change (%)			
UAA	**ha/farm**	20.2	-0.0	-0.0	-0.4	-0.7	19.9	-0.0	-0.5	-1.9	-3.2	20.2	-0.0	-0.1	-0.6	-0.9
Share open arable land	**%**	29.4	0.0	0.0	-0.0	0.0	7.8	-0.0	-0.0	0.0	-0.0	27.3	-0.0	0.0	0.0	-0.0
Share bread cereals	**%**	8.7	-1.3	-0.0	-0.6	-2.1	3.6	-0.9	-0.0	-0.6	-1.4	8.2	-1.3	-0.0	-0.6	-2.1
Share fodder cereals	**%**	6.0	-1.0	0.0	-0.6	-1.7	0.8	-0.8	-0.0	-0.7	-1.4	5.5	-1.0	0.0	-0.6	-1.7
Share maize	**%**	6.2	1.1	-0.1	1.5	2.2	1.4	1.2	0.0	1.5	2.4	5.7	1.1	-0.1	1.5	2.2
Share root crops	**%**	4.1	1.4	0.0	-0.2	1.1	0.6	0.9	-0.0	-0.2	0.8	3.8	1.4	0.0	-0.2	1.1
Share pulses	**%**	0.6	4.9	-0.0	-0.7	3.9	0.1	7.0	-0.1	-1.5	5.7	0.5	4.9	-0.0	-0.7	4.0
Share oilseeds	**%**	2.7	-0.0	-0.0	-0.8	-1.1	0.0	-0.8	-0.0	-0.9	-1.4	2.4	-0.0	-0.0	-0.8	-1.1
Share fallow	**%**	0.3	6.9	1.2	8.9	32.1	0.1	11.5	0.6	12.7	24.7	0.3	7.0	1.2	8.9	31.9
Share other arable land	**%**	0.7	0.1	-0.0	-0.0	0.1	1.2	-	-	-	-	0.8	0.1	-0.0	-0.0	0.1
Share total grasland	**%**	67.4	-0.0	-0.0	-0.7	-1.0	90.8	-	-0.5	-2.1	-3.5	69.7	-0.0	-0.1	-0.8	-1.3
Share ley	**%**	13.3	-0.1	2.2	0.8	2.4	7.7	-0.0	2.8	3.4	5.1	12.8	-0.1	2.2	0.9	2.5
Share permanent meadows	**%**	46.0	0.1	-1.5	-2.5	-3.8	71.8	-0.0	-1.6	-3.8	-6.0	48.5	0.1	-1.5	-2.7	-4.1
Share permanent pastures	**%**	8.1	-0.3	4.9	7.4	9.5	11.3	0.1	3.9	4.4	6.0	8.4	-0.2	4.8	7.0	9.1
Share permanent crops	**%**	3.2	-	-	-	-	1.4	-	-	-	-	3.0	-	-	-	-

Scenarios:
A: Abolition of extenso payments
B: Abolition of payments for less intensive meadows
C: Abolition of payments for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

Source: own calculations based on Swiss FADN

Compared to conventional farms, organic farms commonly respond similarly to the four scenarios but to a different extent, indicating a different level of economic dependence on different ecological direct payments. In contrast to conventional farms, arable crops on organic farms are less severely affected than on conventional farms (e.g. fallows). However, pulses and oilseeds, constituting a much lower absolute share of UAA on organic farms, show more elastic responses. Grassland activities generally show more flexible responses to the policy scenarios, revealing a higher susceptibility of organic farms to changes in grassland-related direct payments than conventional farms.

Livestock activities and labour requirements

On all farms, livestock activities are only marginally affected in the different scenarios (Table 51). Both Scenario A and Scenario B do not lead to substantial changes in the agricultural sector. Scenario C and Scenario D increase total livestock units by 1.7 and 1.8 %, attributable to greater numbers of dairy cows and ruminants. The higher density of livestock is associated with more intensive grassland and entails a lower fodder area per roughage-consuming livestock (-3.5 % in Scenario C and -4.2 % in Scenario D). Pig and poultry stocking densities are not affected by the Scenarios. The number of working units rises by 1.1 % in Scenarios C and D due to an increasing number of cattle. The results indicate that the abolition of the policy measure ‘extensive grassland’ affects mainly ruminant stocking density, while both extenso payments and less intensive meadows do not affect livestock.

The responses do not differ substantially between organic and conventional farms. However, the intensification response on the farms is more pronounced on organic farms, due to the higher average shares of land under the respective agri-environmental policies (Table 51).

Table 51 Relative responses in livestock units and labour requirements of conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
			relative change (%)					relative change (%)					relative change (%)			
Total livestock units	**LU/farm**	25.4	0.0	-0.0	1.8	1.7	20.3	0.0	0.1	1.8	1.5	24.9	0.0	-0.0	1.8	1.7
Share ruminants	**%**	79.8	0.0	-0.0	2.2	2.1	95.8	0.0	0.1	1.9	1.6	81.1	0.0	-0.0	2.2	2.0
Share dairy	**%**	51.1	0.1	-0.1	1.8	1.7	44.2	0.0	0.1	1.9	1.6	50.5	0.1	-0.1	1.8	1.7
Share beef	**%**	8.3	0.2	0.1	2.5	2.7	21.3	0.0	0.1	2.2	1.8	9.3	0.2	0.1	2.5	2.5
Share pigs	**%**	15.2	-0.0	-0.0	-0.0	-0.0	2.4	-0.0	-0.0	0.0	0.0	14.2	-0.0	-0.0	-0.0	-0.0
Share poultry	**%**	4.7	0.0	0.0	-0.0	-0.0	1.7	0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.0
Share oth. animals	**%**	0.2	-	-	-	-	0.0	-	-	-	-	0.2	-	-	-	-
Forage area per RLU	**ha**	0.1	0.0	0.0	-3.4	-4.1	0.5	0.1	-0.5	-3.7	-4.4	0.2	0.0	-0.1	-3.5	-4.2
LU per ha UAA	**LU/ha**	1.3	0.0	-0.0	2.2	2.3	1.0	0.0	0.6	3.8	4.9	1.2	0.0	0.0	2.4	2.6
Average working units	**AWU/farm**	1.6	0.0	0.1	1.1	1.1	1.7	0.0	0.2	0.9	0.8	1.6	0.0	0.1	1.1	1.1
Family working units	**FWU/farm**	1.2	-	-0.0	0.0	0.0	1.3	-	-	-	-	1.2	-	-0.0	0.0	0.0

RLU = roughage-consuming livestock,
AWU = average working units, FWU = family working units
Scenarios:
A: Abolition of extenso payments
B: Abolition of payments for less intensive meadows
C: Abolition of payments for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

Source: own calculations based on Swiss FADN

7.3.2 Financial performance

The financial performance of the farms is generally influenced only slightly by the Scenarios, as the ecological direct payments account for only a small share of total direct payments (Table 52).

Most prominently, ecological direct payments are reduced in the scenarios by 7.3 % (Scenario A), 9.3 % (Scenario C) and 25.5 % (Scenario D), due to the abolition of the respective direct payment. As a consequence, compensating for the loss of less intensive meadows by means of extensive meadows in order to maintain compliance with the PEP, the total ecological direct payments remain almost constant in Scenario B for conventional farms as do the other financial indicators in this scenario. Organic farms, on the contrary, suffer losses in total ecological direct payments in a similar dimension to that in Scenario A (-2.8 %). Apart from Scenario D, income reductions are marginal. In Scenario D, farm income decreases by 1.6 %, while farm income per AWU and family farm income decrease by 2.6 to 2.9 %, respectively. Due to intensification of farms and an increase in ruminant livestock units, the production value of livestock rises by 1.2 %, partly compensating for the 4.1 % losses in total direct payments.

Table 52 Relative responses in financial parameters of conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
			relative change (%)					relative change (%)					relative change (%)			
Farm income	**kCHF**	114.7	-0.5	-0.2	-0.1	-1.5	111.4	-0.4	-0.5	-0.8	-2.4	114.3	-0.5	-0.2	-0.2	-1.6
Farm income per FWU	**kCHF**	70.6	-0.5	-0.2	-1.2	-2.6	65.8	-0.4	-0.7	-1.7	-3.1	70.1	-0.5	-0.3	-1.3	-2.6
Farmily farm income	**kCHF**	87.7	-0.7	-0.3	-1.0	-2.9	90.8	-0.5	-0.8	-1.7	-3.6	88.0	-0.7	-0.4	-1.1	-2.9
Farmily farm income per FWU	**kCHF**	63.4	-0.6	-0.3	-1.3	-2.8	62.6	-0.4	-0.7	-1.8	-3.3	63.3	-0.6	-0.3	-1.3	-2.9
Total costs	**kCHF**	154.3	0.2	0.2	1.1	1.3	112.5	0.1	0.3	1.4	1.3	150.2	0.2	0.2	1.1	1.3
Total revenues	**kCHF**	242.0	-0.2	-0.0	0.3	-0.2	203.3	-0.2	-0.2	-0.0	-0.9	238.2	-0.2	-0.0	0.3	-0.3
Total direct payments	**kCHF**	50.1	-1.2	0.0	-1.2	-4.1	64.6	-0.5	-0.7	-1.8	-4.3	51.5	-1.1	-0.0	-1.3	-4.1
General direct payments	**kCHF**	38.9	-0.0	0.0	0.3	0.1	47.7	-0.0	-0.3	-0.5	-1.5	39.8	-0.0	-0.0	0.2	-0.1
Ecological direct payments	**kCHF**	7.8	-8.0	0.2	-9.6	-27.1	12.4	-2.8	-2.4	-7.6	-16.6	8.2	-7.3	-0.2	-9.3	-25.5
Total prodcution value	**kCHF**	151.5	0.2	-0.0	0.9	1.0	96.6	0.0	0.0	1.2	1.0	146.1	0.2	-0.0	0.9	1.0
Production value of crops	**kCHF**	36.5	0.5	0.0	-0.2	0.2	23.3	-0.0	-0.0	-0.1	-0.1	35.2	0.5	0.0	-0.2	0.2
Production value of livestock	**kCHF**	115.0	0.1	-0.0	1.3	1.2	73.3	0.0	0.1	1.6	1.4	110.9	0.1	-0.0	1.3	1.2

Scenarios:
A: Abolition of extenso payments
B: Abolition of payments for less intensive meadows
C: Abolition of payments for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

Source: own calculations based on Swiss FADN

7.3.3 Policy uptake

Policy uptake was identified as one of the key determinants of the cost-effectiveness of agri-environmental policies in Section 6.1.1. Accordingly, model reactions to the Scenarios are indicative for cost-effectiveness.

Extenso payments

Extenso uptake is substantially affected only by Scenario A and D. Table 53 shows about equal responses for both Scenarios, which can be attributed almost entirely to the abolition of extenso payments. While the share of intensive oilseed rape and grains grows by about 10 %, shares of extensive grains decrease by 10.5 %. Extensive rape shares of total rape land decrease by 25.8 %, overcompensating in absolute terms the increase in extensive rape. The modest decline in extenso uptake for grains in particular implies substantial windfall profits from extenso payments, at least for some of the modelled farm groups.

Total extenso area is reduced by 12.6 % due to the elimination of the payment. As organic farms do not grow intensive variants of grains and rape, shares in total UAA decrease slightly due to the generally declining profitability of organic cereals and rape compared to other arable crops. Noticeable, however, is the very slight decrease as compared to conventional farms, suggesting the particularly high deadweight effect of extenso payments for organic farms.

Table 53 Relative responses in extenso uptake of conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
Crop shares			relative change (%)					relative change (%)					relative change (%)			
Intensive grains	**%**	50.2	11.1	0.0	0.0	11.2	-	-	-	-	-	48.6	11.1	0.0	0.0	11.1
Extenso grains	**%**	49.8	-11.1	-0.0	-0.0	-11.2	100.0	-	-	-	-	51.4	-10.5	-0.0	-0.0	-10.5
Intensive rape	**%**	72.4	9.9	0.0	0.2	10.3	-	-	-	-	-	72.3	9.9	0.0	0.2	10.3
Extenso rape	**%**	27.6	-26.0	-0.0	-0.6	-27.1	100.0	-	-	-	-	27.7	-25.8	-0.0	-0.6	-26.9
UAA shares																
Intensive grains	**%**	7.4	9.7	0.0	-0.2	9.7	-	-	-	-	-	6.7	9.7	0.1	-0.0	10.0
Extenso grains	**%**	7.3	-12.2	0.0	-0.2	-12.4	4.4	-0.9	0.5	1.4	1.9	7.1	-11.5	0.1	-0.0	-11.5
Intensive rape	**%**	1.5	8.1	-0.0	-0.1	8.1	-	-	-	-	-	1.4	8.1	0.0	0.1	8.4
Extenso rape	**%**	0.6	-27.2	-0.0	-0.9	-28.5	0.0	-0.8	0.5	1.1	1.8	0.5	-27.0	0.0	-0.7	-28.1
Total extenso	**%**	7.9	-13.3	0.0	-0.2	-13.6	4.4	-0.9	0.5	1.4	1.9	7.6	-12.6	0.1	-0.1	-12.6

Source: own calculations based on Swiss FADN

Scenarios:
A: Abolition of extenso payments
B: Abolition of payments for less intensive meadows
C: Abolition of payments for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

Ecological compensation areas

Ecological compensation areas (ECA) are strongly affected in Scenarios B, C, and D (Table 54). Although marginal effects occur for grassland in Scenario A, marked increases in mixed and rotational fallows must be noted, as described above.

The total share of ECA in total UAA falls in Scenario B by 10.5 %. In Scenario C, a stronger decrease of 12.9 % occurs, while the strongest decrease was modelled for Scenario D, with 21.8 %. Organic farms show more sensitive responses, as the base-year share of ECA is higher and the farms are more dependent on ecological direct payments.

In Scenarios B, C and D, intensive meadows experience an increase of 3.1 %, 4.4 % and 7.2 % respectively compared to total meadows[73]. A substantial reduction in less intensive meadows can be noted in Scenario B (78.2 %), accompanied by a 25.7 % increase in extensive meadows. In Scenario C, the area of less intensive meadows almost doubles, while extensive meadows decrease by 83.9 %. In Scenario D, where both payments are abolished, the model shows an increase in less intensive meadows and a decrease in extensive meadows. The increase in less intensive meadows may be attributed to the PEP restriction, which postulates 7 % of ECA area for cross-compliance. As the relative profitability of less intensive meadows is higher compared to extensive meadows, the farms try to minimise their financial losses by increasing less intensive meadows. In absolute terms, however, intensive meadows show the strongest increase among meadows. In this regard, the response of organic farms differs fundamentally from the response of conventional ones. Organic farms cultivate both less extensive and intensive meadows in Scenario D. However, they reduce extensive meadows less drastically. There are several reasons for this difference between the farming systems. First, organic farms have higher shares of ECA and hence have a greater scope for reducing total ECA. Second, a larger proportion of organic farms are located in the mountain areas, where the payment rates for less intensive meadows and extensive meadows are similar.

[73] alpine meadows not included

Table 54 Relative responses in ECA uptake of conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
Share relative to crop area			relative change (%)					relative change (%)					relative change (%)			
Intensive meadows	**%**	80.6	-0.1	3.0	4.0	6.6	75.0	0.0	4.0	6.2	11.2	79.8	-0.1	3.1	4.4	7.2
Less intensive meadows	**%**	6.9	1.5	-84.7	111.8	46.0	10.2	0.0	-51.9	42.0	-10.5	7.4	1.2	-78.2	98.0	34.8
Extensive meadows	**%**	12.5	0.1	27.7	-88.5	-68.5	14.8	-0.1	15.5	-60.3	-49.1	12.8	0.1	25.7	-83.9	-65.4
Intensive pastures	**%**	84.9	0.1	0.1	-2.0	-1.5	83.1	0.0	0.4	-0.5	-0.3	84.6	0.1	0.2	-1.8	-1.4
Extensive pastures	**%**	15.1	-0.6	-0.8	11.0	8.6	16.9	-0.2	-2.1	2.6	1.4	15.4	-0.5	-1.0	9.7	7.5
Share relative to UAA																
Intensive meadows	**%**	37.0	-0.1	1.4	1.9	3.3	53.8	0.0	2.8	4.3	8.0	38.6	-0.1	1.6	2.2	3.8
Less intensive meadows	**%**	3.2	1.6	-84.9	107.5	41.4	7.3	0.0	-52.4	39.3	-13.1	3.6	1.3	-78.5	93.9	30.5
Extensive meadows	**%**	5.7	0.2	25.8	-88.8	-69.5	10.6	-0.1	14.2	-61.1	-50.6	6.2	0.2	23.8	-84.2	-66.5
Intensive pastures	**%**	6.8	-0.2	5.1	5.8	8.6	9.4	0.1	4.9	5.9	9.2	7.1	-0.1	5.1	5.8	8.6
Extensive pastures	**%**	1.2	-0.9	4.1	19.7	19.7	1.9	-0.1	2.2	9.2	11.1	1.3	-0.8	3.8	18.1	18.3
Mixed fallows	**%**	0.2	6.4	1.4	10.7	32.6	0.0	11.8	1.4	16.7	31.8	0.2	6.6	1.4	11.0	32.9
Rotational fallows	**%**	0.1	7.6	0.9	7.3	33.5	0.0	10.7	0.5	10.7	21.9	0.1	7.7	0.9	7.5	33.6
Other ECA	**%**	0.9	0.0	0.0	0.4	0.7	1.0	0.0	0.5	2.0	3.3	0.9	0.0	0.1	0.6	0.9
Total ECA	**%**	11.4	0.7	-10.4	-12.1	-20.2	20.9	-0.0	-10.8	-16.3	-29.0	12.3	0.6	-10.5	-12.9	-21.8

Source: own calculations based on Swiss FADN

Scenarios:
A: Abolition of extenso payments
B: Abolition of payments for less intensive meadows
C: Abolition of payments for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

On all farms, fallows and extensive pastures are used to compensate for the lost ECA through the reduction in the share of ECA meadows. While extensive pastures increase in both Scenario C and D by about 18 %, the share of mixed and rotational fallows rises by 33 % in Scenario D in contrast to 7.5 to 11 % in Scenario C.

7.3.4 Fossil energy use

The above mentioned model responses regarding farm structure and policy uptake result in only marginal changes in fossil energy use (Table 55). Fossil energy use decreases most in Scenarios C and D, while Scenario A and B show only minor effects on energy use.

The difference in total energy use between Scenario A and the base year is 0.1 %. The abolition of extenso payments leads to a 0.9 % rise in energy use for crop protection.

Scenario B even results in a decrease in total energy use due to the stronger uptake of extensive meadows, which gives rise to a lower energy use per ha and a marginal reduction in ruminant stocking rates, which overcompensate for the meadows being intensified. Scenarios C and D lead to a stronger increase in energy use by about 1.5 %. The fact that these two scenarios respond very similarly indicates that the effects in Scenario D are caused almost exclusively by the abolition of extensive meadows. This is attributable to the intensification of meadows and the accompanying increase in ruminant livestock. This intensification triggers a higher energy use in the following categories: buildings (2.6 %), animal husbandry (1.4 %), crop protection (2.3 %), fertilising (2.6 %), and harvesting (2.0 %).

The responses of organic farms to policy scenarios B, C and D differ significantly from the responses of conventional farms. Organic farms show higher relative increases in energy use as a response to the changes assumed in the policy scenarios.

Neither on organic nor on conventional farms is energy use affected substantially when less intensive meadows are abolished (Scenario B). While conventional farms show a slight increase in energy use (0.1 %) when less intensive meadows are abolished (Scenario B), organic farms display the opposite response. Organic farms respond more sensitively due to a) a lower absolute energy use per ha, b) a higher share of grassland, which is the type of land directly affected by Scenarios B, C, and D, and c) higher uptake levels of the policy measures ‘less intensive meadows’ and ‘extensive meadows’ in the base year.

Table 55 Relative response in fossil energy use per ha of conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
			relative change (%)					relative change (%)					relative change (%)			
Total fossil energy use (FEU)	**GJ/ha**	44.3	0.1	-0.1	1.3	1.4	20.2	0.0	0.1	3.1	3.8	41.9	0.1	-0.1	1.4	1.5
FEU for buildings	**GJ/ha**	7.4	0.0	-0.5	2.6	2.4	6.4	0.0	0.0	4.1	4.8	7.3	0.0	-0.4	2.7	2.6
FEU for animal husbandry	**GJ/ha**	9.4	0.0	-0.1	1.2	1.3	5.3	0.0	0.4	3.0	3.9	9.0	0.0	-0.0	1.3	1.4
FEU for purchased fodder	**GJ/ha**	16.0	0.0	-0.0	0.4	0.5	3.4	0.1	0.2	1.3	1.8	14.8	0.0	-0.0	0.5	0.5
FEU for tillage	**GJ/ha**	1.0	-0.3	0.2	0.2	0.1	0.4	-0.2	0.5	1.6	1.7	0.9	-0.3	0.2	0.3	0.2
FEU for seeding	**GJ/ha**	0.4	0.0	0.2	0.3	0.5	0.1	0.2	0.6	1.7	2.3	0.4	0.0	0.3	0.4	0.6
FEU for plant protection	**GJ/ha**	0.5	1.0	0.1	0.9	2.3	0.1	0.2	0.0	2.8	3.0	0.5	0.9	0.1	1.0	2.3
FEU for fertilising	**GJ/ha**	4.7	0.2	0.1	2.1	2.5	0.8	0.0	-0.5	5.8	5.9	4.4	0.2	0.1	2.1	2.6
FEU for harvesting	**GJ/ha**	4.4	0.2	-0.4	1.9	1.9	3.4	0.0	-0.1	2.9	3.3	4.3	0.2	-0.4	2.0	2.0
FEU for other	**GJ/ha**	0.5	0.0	-0.1	2.0	2.6	0.3	0.0	-0.3	4.9	5.4	0.5	0.0	-0.1	2.1	2.8

Scenarios:
A: Abolition of extenso payments
B: Abolition of payments for less intensive meadows
C: Abolition of payments for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

Source: own calculations based on Swiss FADN and SALCA data

7.3.5 Habitat quality

Effects regarding habitat quality are more marked than for energy use (Table 56). The exception is Scenario A, which had almost no impact on habitat quality apart from a 0.8 % decrease in habitat quality of arable weeds. Total habitat quality is improved in Scenario B by 2.2 % due to the partial substitution of less intensive meadows by extensive meadows, which perform substantially better on habitat quality. As for species groups, amphibians (-4.4 %), butterflies (-2.5 %), birds (-2.3 %) and snails (-2.0 %) react most sensitively. Scenarios C and D both have a strong negative effect on habitat quality of about -18 %. With respect to the species group indicators, most species are strongly affected – amphibians, grassland weeds, grasshoppers, and butterflies in particular. Only arable weeds remain almost unchanged.

It is particularly notable that Scenario C affects habitat quality more severely than Scenario D does. This effect may be explained by farms reducing their extensive meadows in both scenarios substantially. However, while in Scenario C farms opt for less intensive meadows due to the higher relative profitability of this activity and in order to fulfil the requirements for proof of ecological performance (PEP), this option is not available in Scenario D. Instead, the modelled farms try to compensate for the losses in ECA by a higher uptake of fallows and extensive pastures. The increased use of both activities results in lower losses of habitat quality than in Scenario C, in which the reductions in extensive meadows are primarily compensated by the uptake of less intensive meadows.

In contrast to the energy use indicators, habitat quality is affected more severely with regard to both relative and absolute scale. This is particularly notable because organic farms have a 55 % higher habitat quality on average compared to conventional farms. Thus when payments for less intensive meadows and extensive meadows are abolished, the differences in habitat quality between the farming systems are amplified.

There are only small differences in model response by organic farms compared to conventional farms concerning single species. Most indicator species groups are more strongly affected by Scenarios C and D on organic farms than on conventional farms. An exception is arable weeds, which experience an improvement in average habitat quality due to the increasing share of arable land as a share of total land.

Table 56 Relative response in habitat quality on conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
			relative change (%)					relative change (%)					relative change (%)			
AHQ, all species	%	16.6	-0.1	2.5	-18.7	-18.1	25.7	-0.0	0.7	-16.5	-17.9	17.4	-0.0	2.2	-18.4	-18.0
AHQ for amphibians	%	14.2	0.2	4.8	-23.0	-20.3	23.2	0.0	2.1	-18.9	-19.0	15.1	0.1	4.4	-22.3	-20.1
AHQ for locusts	%	18.4	0.4	1.9	-13.2	-12.3	26.0	0.0	0.5	-12.8	-13.9	19.1	0.4	1.7	-13.1	-12.5
AHQ for carabids	%	24.4	-0.0	1.0	-8.8	-8.5	29.7	-0.0	0.3	-9.7	-10.4	24.9	-0.0	0.9	-8.9	-8.7
AHQ for butterflies	%	16.0	0.4	2.7	-14.5	-12.9	24.3	0.0	1.4	-12.7	-12.9	16.8	0.4	2.5	-14.3	-12.9
AHQ for spiders	%	18.9	-0.1	1.1	-10.7	-10.5	25.4	-0.0	0.2	-10.8	-11.7	19.5	-0.1	1.0	-10.7	-10.7
AHQ for arable weeds	%	5.0	-0.8	0.0	0.3	-0.1	1.9	-0.2	0.0	1.1	0.9	4.7	-0.8	0.0	0.3	-0.1
AHQ for grassland weeds	%	18.4	0.1	1.8	-11.8	-11.3	26.3	-0.0	0.4	-11.2	-12.5	19.2	0.1	1.6	-11.7	-11.5
AHQ for small mammals	%	40.9	0.1	-1.0	-1.3	-2.1	53.0	0.0	-1.3	-2.3	-3.9	42.1	0.1	-1.0	-1.4	-2.4
AHQ for birds	%	14.6	0.1	2.4	-11.1	-9.8	24.3	0.0	1.4	-8.5	-8.1	15.6	0.1	2.3	-10.7	-9.5
AHQ for wild bees	%	16.9	0.1	-0.2	-8.9	-9.6	25.0	-0.0	-1.2	-9.0	-11.5	17.7	0.1	-0.3	-8.9	-9.9
AHQ for snails	%	42.2	0.1	2.0	-6.8	-5.4	42.7	0.0	2.0	-8.1	-6.8	42.3	0.1	2.0	-6.9	-5.6

AHQ = Average habitat quality

Source: own calculations based on Swiss FADN and SALCA data

Scenarios:
A: Abolition of extenso contributions;
B: Abolition of contributions for less intensive meadows;
C: Abolition of contributions for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

7.3.6 Eutrophication

The effects of the policy scenarios on eutrophication range from generally marginal (Scenario A), minor and non-uniform (Scenario B), a medium increase (Scenario C) to a marked increase (Scenario D). Total eutrophication levels remain almost unchanged when extenso payments are abolished.

The abolition of less intensive meadows leads to an increase in nitrate eutrophication by 0.7 % and a decrease in ammonia eutrophication by 0.6 %, with total eutrophication remaining almost unchanged. The increase in nitrate eutrophication can be attributed to an increase in leys (Table 50), while ammonia eutrophication goes down slightly, driven by reductions in livestock density and accumulation of organic manure (Table 57).

More extreme responses of overall eutrophication take place in Scenario C (1.9 %) and D (2.2 %). Increases in nitrate eutrophication are slightly less severe in Scenario C (0.5 %) than in Scenario B (0.7 %), while Scenario D results in the strongest increase (1.6 %). Due to the higher stocking rates entailed by Scenarios C and D, ammonia eutrophication increases by 3.3 % (Scenario C) and 2.9 % (Scenario D).

Phosphorus eutrophication increases by 1.2 % (Scenario D) to 1.3 % (Scenario D), while remaining almost unchanged in Scenarios A and B.

Similarly as for energy use, organic farms tend to react more sensitively to the scenarios in terms of total eutrophication than conventional farms. The higher relative responses are driven by increases in ammonia and nitrate eutrophication. In Scenario C, nitrate eutrophication increases by 2.3 % as opposed to 0.4 % on conventional farms, while eutrophication involving ammonia shows near equal responses for both farming systems, 3.2 % and 3.3 %. In Scenario D nitrate eutrophication increases by 3.0 % as against 1.5 % on conventional farms, while ammonia eutrophication increases by 3.9 % compared with 2.8 % on conventional farms. The model demonstrates less pronounced responses on organic farms than on conventional farms regarding phosphorus and other nitrogen eutrophication.

Table 57 Relative response in eutrophication with nitrogen and phosphorus from conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
			relative change (%)					relative change (%)					relative change (%)			
Total eutrophication	**kg N-eq / ha**	90.6	0.2	-0.1	1.9	2.1	58.7	-0.1	0.6	2.7	3.2	87.5	0.2	-0.0	1.9	2.2
N-eutrophication	**kg N-eq / ha**	82.9	0.2	-0.1	1.9	2.2	51.8	-0.1	0.7	2.9	3.5	79.9	0.2	-0.0	2.0	2.3
NO_3-eutrophication	**kg N-eq / ha**	37.4	0.6	0.7	0.4	1.5	14.3	-0.3	1.4	2.3	3.0	35.1	0.6	0.7	0.5	1.6
NH_3-eutrophication	**kg N-eq / ha**	43.5	-0.1	-0.7	3.3	2.8	36.4	0.0	0.4	3.2	3.9	42.8	-0.0	-0.6	3.3	2.9
Other N-eutrophication	**kg N-eq / ha**	2.1	0.0	0.3	0.4	0.6	1.2	-0.0	0.5	-0.5	-0.1	2.0	0.0	0.3	0.3	0.6
P-eutrophication	**kg P-eq / ha**	7.7	0.1	-0.3	1.3	1.3	6.8	-0.0	-0.2	1.0	0.8	7.6	0.1	-0.3	1.3	1.2

Scenarios:
A: Abolition of extenso payments
B: Abolition of payments for less intensive meadows
C: Abolition of payments for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

Source: own calculations based on Swiss FADN and SALCA data

7.3.7 Public expenditure

Total public expenditure is affected only slightly in the Scenarios (Table 58). A 1.1 % decrease in total direct payments can be noted if extenso payments are abolished (Scenario A). The decrease in direct payments is relatively greater for conventional farms, as they have higher shares of cereals and are less dependent on other direct payments than organic farms. Scenario B has almost no effect on public expenditure. The abolition of less intensive meadows even led to a marginal increase in public expenditure, since farms partly switched to extensive meadows, and received slightly higher payment rates. Scenario C led to a decrease in public expenditure of 0.5 % as a direct consequence of the abolition of payments for extensive meadows, which is partly compensated for by increased general direct payments for animal husbandry. In Scenario D, there is a more substantial reduction in payments by 2.8 % compared to the base year situation. The reduction in overall direct payments can be attributed to the decline in ECA payments and extenso payments. A compensation of direct payment losses is induced by an increase in RGVE payments, TEP payments, BTS and RAUS payments.

Farm-level transaction costs remain almost unaffected, showing responses of less than 0.3 % to 0.4 %. It is remarkable, however, that farm level PRTC increase in Scenario C, in contrast to public PRTC, which show a more significant decrease by up to 3.1 % in Scenario D. The payment efficiency remains almost unaffected by the abolition of the agri-environmental payments in the scenarios.

Payments for ecological compensation areas are not affected by Scenario A and even show a slight increase in Scenario B due to the higher uptake levels of extensive meadows in this scenario. In Scenarios C and D, ECA payments go down by 46.9 % and 86.8 % respectively. Extenso payments, by contrast, are affected only by Scenarios A and D, which include a complete abolition of this payment.

Table 58 Relative response in public expenditure parameters of conventional, organic and all farms to Scenarios A-D

Indicator	Unit	Conventional farms					Organic farms					All farms				
		Base year	Scenarios				Base year	Scenarios				Base year	Scenarios			
			A	B	C	D		A	B	C	D		A	B	C	D
			relative change (%)					relative change (%)					relative change (%)			
Total public expenditure	CHF/ha	2,579.0	-1.1	0.0	-0.6	-3.0	3,265.0	-0.6	-0.2	0.3	-0.7	2,646.4	-1.1	0.0	-0.5	-2.8
Direct payments	CHF/ha	2,534.7	-1.2	0.1	-0.6	-3.1	3,224.6	-0.6	-0.1	0.4	-0.6	2,602.5	-1.1	0.0	-0.5	-2.8
Total policy-related transaction costs	CHF/ha	119.7	-0.4	-0.9	-0.2	-1.2	116.0	-0.0	-1.3	-1.2	-2.6	119.4	-0.3	-0.9	-0.3	-1.4
Public policy-related transaction costs	CHF/ha	44.3	-0.3	-1.7	-1.0	-2.7	40.4	-0.0	-2.9	-3.3	-6.6	43.9	-0.3	-1.8	-1.3	-3.1
Farm-level policy-related transaction costs	CHF/ha	75.4	-0.4	-0.4	0.3	-0.4	75.6	-0.0	-0.4	-0.1	-0.5	75.5	-0.3	-0.4	0.3	-0.4
Payment efficiency	%	95.4	0.0	-0.0	0.0	0.1	96.4	0.0	-0.0	-0.1	-0.1	95.5	0.0	-0.0	0.0	0.1
Total direct payments	kCHF	50.1	-1.2	0.0	-1.2	-4.1	64.6	-0.5	-0.7	-1.8	-4.3	51.5	-1.1	-0.0	-1.3	-4.1
General direct payments	kCHF	38.9	-0.0	0.0	0.3	0.1	47.7	-0.0	-0.3	-0.5	-1.5	39.8	-0.0	-0.0	0.2	-0.1
Area payments	kCHF	25.8	-0.0	-0.0	-0.4	-0.6	24.3	-0.0	-0.5	-1.9	-3.2	25.6	-0.0	-0.1	-0.5	-0.8
Payments for roughage-consuming livestock	kCHF	6.2	-0.0	0.1	2.6	2.5	9.8	-0.0	0.1	1.8	1.4	6.6	-0.0	0.1	2.5	2.4
Payments for TEP	kCHF	5.0	-0.0	0.3	2.0	2.0	10.1	-0.0	0.0	1.5	1.0	5.5	-0.0	0.2	1.9	1.8
Hillside payments	kCHF	1.9	0.0	-0.9	-2.0	-3.1	3.6	-0.0	-1.3	-3.4	-5.6	2.1	0.0	-1.0	-2.2	-3.5
Ecological direct payments	kCHF	7.8	-8.0	0.2	-9.6	-27.1	12.4	-2.8	-2.4	-7.6	-16.6	8.2	-7.3	-0.2	-9.3	-25.5
ECA payments	kCHF	2.4	1.2	1.1	-45.9	-85.8	2.3	0.0	-16.6	-55.6	-97.1	2.4	1.1	-0.6	-46.9	-86.8
Extenso payments	kCHF	0.6	-100.0	0.0	-0.7	-100.0	0.3	-100.0	-0.0	-0.6	-100.0	0.6	-100.0	0.0	-0.7	-100.0
OFSAP	kCHF	0.0	0.0	-0.1	-0.7	-1.1	5.0	-0.0	-0.4	-1.5	-2.5	0.5	-0.0	-0.4	-1.5	-2.4
Payments for BTS	kCHF	1.3	-0.0	-0.1	1.4	1.4	1.0	0.0	0.1	1.9	1.6	1.2	-0.0	-0.0	1.5	1.4
Payments for livestock with outdoor exercise	kCHF	3.4	0.0	0.0	1.8	1.8	3.7	0.0	0.1	1.8	1.5	3.4	0.0	0.0	1.8	1.7
Crop-specific payments	kCHF	1.0	0.8	-0.0	-0.7	-0.2	0.0	4.4	-0.1	-1.4	3.2	0.9	0.8	-0.0	-0.7	-0.2
Payments for alpine summer grazing	kCHF	0.7	0.0	0.5	2.4	2.5	1.7	0.0	-0.0	1.4	0.8	0.8	0.0	0.3	2.2	2.1
Other payments	kCHF	1.8	-	-	-	-	2.6	-	-	-	-	1.9	-	-	-	-

Source: own calculations based on Swiss FADN

TEP = Animal husbandry under adverse conditions
BTS = Particular animal-friendly stabling
Scenarios:
A: Abolition of extenso payments
B: Abolition of payments for less intensive meadows
C: Abolition of payments for extensive meadows
D: Abolition of payments for extenso, less intensive and extensive meadows

Total public expenditure for organic farms is affected to a smaller extent by the policy scenarios than for conventional farms (-0.7 % compared with -3.0 %). ECA payments are more strongly affected by the policy scenarios on organic farms than on conventional farms. Scenario C results in a decrease of 55.6 % on organic farms and an even smaller decrease of 45.9 % on conventional farms. Scenario D entails a 97 % reduction in ECA payments on organic farms, but only an 85.8 % reduction on conventional farms. Moreover, while Scenario B leads to additional public expenditure on conventional farms, it entails a reduction in ECA payments of 16.6 % to organic farms.

7.3.8 Cost-effectiveness

As described in Section 6.3.9, cost-effectiveness was calculated on the basis of environmental effects (Sections 7.3.4 to 7.3.6) and public expenditure (Section 7.3.7) for the policy measures. This section presents, first, cost-effectiveness ratios. It then outlines abatement and provision costs of policies.

Cost-effectiveness ratio

Table 59 shows the cost-effectiveness of the agri-environmental payments ‘extenso payments’, ‘less intensive meadows’, ‘extensive meadows’ and of the combination of all three payments. The figures indicate the relative improvement that would be possible with 100 CHF per ha[74]. In terms of all farms, extenso payments have a relatively low cost-effectiveness, with less than a 1 % improvement in the environmental indicators per 100 CHF/ha. The cost-effectiveness of less intensive meadows is not defined, because at sector level this measure leads to cost reductions of 1.1 CHF/ha. Payments for extensive meadows have a high cost-effectiveness, as they lead to a significant improvement in environmental indicators compared to the public expenditure entailed with these payments. Finally, the combination of the three payments has a high cost-effectiveness. It leads to

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[74] These cost-effectiveness values are only a calculative figure, as they implicitly assume a linear relationship between environmental effects and public expenditure. However, this relation is not given. So the cost-effectiveness values refer only to the payment as it is implemented currently, in terms of payment level and policy environment (including the interactions with other policy measures and the given producer price levels). This aspect is addressed by sensitivity analyses regarding the payment level (see Section 7.5.1).

slightly higher environmental effects than the payments for extensive meadows, however, at higher public expenditure.

The cost-effectiveness of policies regarding the use of fossil energy ranges from 0.3 %/100 CHF for extenso to 2 %/100 CHF for the combined payments, and to 10.8 %/100 CHF for extensive meadows.

The same ranking of the measures as for energy use can be observed for habitat quality. Extenso payments lead to even lower improvements in habitat quality (0.17 %/100 CHF). Most notable is the improvement in habitat quality for arable weeds due to extenso payments, which, at 2.66 %/100 CHF, is even higher than for the other agri-environmental payments analysed. Both the combination of payments and the payments for extensive meadows have a higher cost-effectiveness regarding average habitat quality than the extenso payments, with 144 %/100 CHF for extensive meadows and 25 %/100 CHF for the combined payments.

Extenso payments have a higher cost-effectiveness regarding eutrophication than regarding habitat quality and energy use (0.7 %/100 CHF). This relatively high cost-effectiveness regarding eutrophication (compared to the other environmental impact categories) is driven in particular by its potential for nitrogen reduction, which is in a similar range compared to the other two payments (extensive meadows 3.9 %/100 CHF; combination of agri-environmental policies: 2.2 %/100 CHF). However, total eutrophication could be improved more significantly with the same public expenditure by supporting the extensive meadows (15.1 %/100 CHF) or by combining the agri-environmental payments (4 %/100 CHF).

Organic farms are less cost-effective than conventional farms when implementing extenso payments. The cost-effectiveness of less intensive meadows is defined for organic farms as opposed to all farms and conventional farms. However, as this payment causes negative environmental impacts, the cost-effectiveness ratio becomes negative. For organic farms, the cost-effectiveness of extensive meadows in not defined due to negative public expenditure. These budget savings (10.4 CHF/ha) can be attributed to lower general direct payments and in particular to lower stocking rates.

Table 59 Cost-effectiveness of agri-environmental policy measures regarding the analysed environmental indicators

Indicator	Unit	Conventional farms				Organic farms				All farms			
		Extenso	Less intensive meadows	Exten-sive meadows	Com-bined AEM	Extenso	Less intensive meadows	Exten-sive meadows	Com-bined AEM	Extenso	Less intensive meadows	Exten-sive meadows	Com-bined AEM
Total fossil energy use	**%/100 CHF**	0.3	n.d.	8.5	1.8	**0.2**	**2.5**	**n.d.**	**16.3**	0.3	n.d.	10.8	2.0
Total habitat quality	**%/100 CHF**	0.18	n.d.	122.27	22.99	**0.12**	**-13.41**	**n.d.**	**77.36**	0.17	n.d.	143.94	24.67
Habitat quality for amphibians	**%/100 CHF**	-0.55	n.d.	150.21	25.88	**-0.06**	**-42.08**	**n.d.**	**82.26**	-0.49	n.d.	175.12	27.53
Habitat quality for locusts	**%/100 CHF**	-1.49	n.d.	86.22	15.58	**-0.23**	**-10.78**	**n.d.**	**60.03**	-1.36	n.d.	102.88	17.04
Habitat quality for carabids	**%/100 CHF**	0.00	n.d.	57.50	10.84	**0.06**	**-5.95**	**n.d.**	**44.80**	0.01	n.d.	69.65	11.95
Habitat quality for butterflies	**%/100 CHF**	-1.51	n.d.	95.16	16.41	**-0.23**	**-27.07**	**n.d.**	**55.73**	-1.37	n.d.	111.95	17.63
Habitat quality for spiders	**%/100 CHF**	0.28	n.d.	69.99	13.39	**0.01**	**-4.83**	**n.d.**	**50.45**	0.26	n.d.	83.90	14.58
Habitat quality for arable weeds	**%/100 CHF**	2.64	n.d.	-1.69	0.18	**1.25**	**-0.16**	**n.d.**	**-4.02**	2.66	n.d.	-2.28	0.14
Habitat quality for grassland weeds	**%/100 CHF**	-0.23	n.d.	77.12	14.41	**0.16**	**-7.03**	**n.d.**	**54.16**	-0.20	n.d.	91.70	15.70
Habitat quality for small mammals	**%/100 CHF**	-0.35	n.d.	8.21	2.71	**-0.07**	**26.15**	**n.d.**	**16.98**	-0.32	n.d.	10.82	3.21
Habitat quality for birds	**%/100 CHF**	-0.23	n.d.	72.37	12.41	**-0.10**	**-28.81**	**n.d.**	**34.88**	-0.21	n.d.	83.54	12.98
Habitat quality for wild bees	**%/100 CHF**	-0.47	n.d.	57.95	12.20	**0.01**	**23.06**	**n.d.**	**49.90**	-0.42	n.d.	69.58	13.48
Habitat quality for molluscs	**%/100 CHF**	-0.27	n.d.	44.52	6.89	**-0.00**	**-40.93**	**n.d.**	**29.30**	-0.25	n.d.	54.35	7.59
Total eutrophication	**%/100 CHF**	0.77	n.d.	12.26	2.68	**-0.35**	**11.47**	**n.d.**	**13.95**	0.73	n.d.	15.10	2.98
Eutrophication with nitrogen	**%/100 CHF**	0.81	n.d.	12.60	2.78	**-0.40**	**13.61**	**n.d.**	**15.32**	0.77	n.d.	15.57	3.10
Nitrate eutrophication	**%/100 CHF**	2.00	n.d.	2.78	1.95	**-1.51**	**27.58**	**n.d.**	**12.77**	1.96	n.d.	3.91	2.18
Ammonia eutrophication	**%/100 CHF**	-0.18	n.d.	21.52	3.58	**0.03**	**8.24**	**n.d.**	**16.82**	-0.17	n.d.	25.73	3.97
Other nitrogen eutrophication	**%/100 CHF**	0.10	n.d.	2.41	0.81	**-0.09**	**10.07**	**n.d.**	**-0.23**	0.09	n.d.	2.50	0.81
Phosphorus eutrophication	**%/100 CHF**	0.27	n.d.	8.65	1.59	**-0.01**	**-4.72**	**n.d.**	**3.56**	0.25	n.d.	10.12	1.66

n.d.: cost-effectiveness of this policy is not defined, as the payment induces negative cost

Source: own calculations based on Swiss FADN and SALCA data

In contrast to the payments for less intensive meadows to conventional farms, this policy measure entails positive environmental effects when paid to organic farms. This means that payments lead both to cost reductions and to marked environmental effects on organic farms.

The combination of all three payments results in higher cost-effectiveness on organic farms than on conventional farms. This is particularly notable because the absolute state of the environmental indicators on organic farms is already significantly higher, which would normally imply a lower cost-effectiveness. Cost-effectiveness regarding energy use is 16.3 %/100 CHF as opposed to 1.8 %/100 CHF on conventional farms. Cost-effectiveness regarding habitat quality is 77.4 %/100 CHF as against 23 %/100 CHF on conventional farms. Finally, cost-effectiveness regarding eutrophication is 14 %/100 CHF compared with 2.7 %/100 CHF on conventional farms.

Abatement and provision cost

Table 60 presents the cost-effectiveness of the measures expressed as abatement and provision costs, which is the reciprocal value of the cost-effectiveness ratio. It shows that achieving relative improvements is most expensive with extenso payments. The indicator shows that habitat-quality improvements are particularly costly. By contrast, extensive meadows achieve very low values for abatement costs of 9.3 CHF/% for energy use and 0.7 CHF/% for habitat quality. Abatement costs for eutrophication, on the other hand, are relatively high at 6.4 CHF/%.

The combination of agri-environmental measures entails higher abatement costs, as the environmental effects are only slightly higher and the costs substantially exceed those of the payments for extensive meadows. Abatement costs regarding energy use are 48.9 CHF/%, while provision costs regarding habitat quality are 4.1 CHF/%. Eutrophication can be abated for 33.6 CHF/% using the combination of agri-environmental measures.

By correspondence to the higher cost-effectiveness of combined payments on organic farms, the abatement costs are substantially lower as compared to conventional farms. With combined payments energy use could be reduced at 6.1 CHF/% as compared to 56.9 CHF/%. Provision costs for habitat quality are only 1.3 CHF/% on organic farms compared to 4.3 CHF/% on conventional farms. With regard to eutrophication, the abatement costs are 7.2 CHF/% compared to 37.4 CHF/% on conventional farms.

Table 60 Abatement and provision costs of agri-environmental policy measures regarding the analysed environmental indicators

Indicator	Unit	Conventional farms				Organic farms				All farms			
		Extenso	Less intensive meadows	Exten-sive meadows	Com-bined AEM	Extenso	Less intensive meadows	Exten-sive meadows	Com-bined AEM	Extenso	Less intensive meadows	Exten-sive meadows	Com-bined AEM
Total fossil energy use	**CHF/%**	386.3	n.d.	11.8	56.9	**631.3**	**39.8**	**n.d.**	**6.1**	383.3	n.d.	9.3	48.9
Total habitat quality	**CHF/%**	543.0	n.d.	0.8	4.3	**859.0**	**-7.5**	**n.d.**	**1.3**	573.1	n.d.	0.7	4.1
Habitat quality for amphibians	**CHF/%**	-182.7	n.d.	0.7	3.9	**-1,773.1**	**-2.4**	**n.d.**	**1.2**	-204.9	n.d.	0.6	3.6
Habitat quality for locusts	**CHF/%**	-67.3	n.d.	1.2	6.4	**-426.1**	**-9.3**	**n.d.**	**1.7**	-73.7	n.d.	1.0	5.9
Habitat quality for carabids	**CHF/%**	27,522.0	n.d.	1.7	9.2	**1,702.4**	**-16.8**	**n.d.**	**2.2**	12,809.6	n.d.	1.4	8.4
Habitat quality for butterflies	**CHF/%**	-66.3	n.d.	1.1	6.1	**-425.6**	**-3.7**	**n.d.**	**1.8**	-73.2	n.d.	0.9	5.7
Habitat quality for spiders	**CHF/%**	354.5	n.d.	1.4	7.5	**13,987.4**	**-20.7**	**n.d.**	**2.0**	390.8	n.d.	1.2	6.9
Habitat quality for arable weeds	**CHF/%**	37.9	n.d.	-59.1	546.0	**80.2**	**-607.4**	**n.d.**	**-24.9**	37.6	n.d.	-43.9	722.5
Habitat quality for grassland weeds	**CHF/%**	-426.7	n.d.	1.3	6.9	**607.1**	**-14.2**	**n.d.**	**1.8**	-510.0	n.d.	1.1	6.4
Habitat quality for small mammals	**CHF/%**	-287.6	n.d.	12.2	37.0	**-1,536.2**	**3.8**	**n.d.**	**5.9**	-311.0	n.d.	9.2	31.1
Habitat quality for birds	**CHF/%**	-433.6	n.d.	1.4	8.1	**-954.1**	**-3.5**	**n.d.**	**2.9**	-468.9	n.d.	1.2	7.7
Habitat quality for wild bees	**CHF/%**	-213.3	n.d.	1.7	8.2	**14,142.0**	**4.3**	**n.d.**	**2.0**	-239.0	n.d.	1.4	7.4
Habitat quality for molluscs	**CHF/%**	-370.2	n.d.	2.2	14.5	**-51,839.9**	**-2.4**	**n.d.**	**3.4**	-395.7	n.d.	1.8	13.2
Total eutrophication	**CHF/%**	130.5	n.d.	8.2	37.4	**-282.5**	**8.7**	**n.d.**	**7.2**	137.4	n.d.	6.6	33.6
Eutrophication with nitrogen	**CHF/%**	123.1	n.d.	7.9	36.0	**-250.4**	**7.3**	**n.d.**	**6.5**	129.3	n.d.	6.4	32.2
Nitrate eutrophication	**CHF/%**	49.9	n.d.	36.0	51.2	**-66.1**	**3.6**	**n.d.**	**7.8**	51.1	n.d.	25.6	46.0
Ammonia eutrophication	**CHF/%**	-568.8	n.d.	4.6	27.9	**3,563.3**	**12.1**	**n.d.**	**5.9**	-603.3	n.d.	3.9	25.2
Other nitrogen eutrophication	**CHF/%**	1,043.4	n.d.	41.6	123.9	**-1,075.1**	**9.9**	**n.d.**	**-436.9**	1,106.8	n.d.	40.0	123.0
Phosphorus eutrophication	**CHF/%**	373.2	n.d.	11.6	62.8	**-10,410.5**	**-21.2**	**n.d.**	**28.1**	395.1	n.d.	9.9	60.3

n.d.: cost-effectiveness of this policy is not defined, as the payment induces negative cost

Source: own calculations based on Swiss FADN and SALCA data

7.4 Comparison of cost-effectiveness of organic farming with agri-environmental policy measures

In this section, the cost-effectiveness of organic farming is compared with the cost-effectiveness of agri-environmental measures by environmental impact category.

Both public expenditure and environmental effects of organic farms were expressed relative to the utilised agricultural area (UAA) of all farms, unlike the calculations in Section 7.2. This transformation results in a better comparability of organic farming with the agri-environmental policy measures, as both environmental effects and public expenditure were also calculated relative to total UAA for the latter policies. However, this transformation leaves both cost-effectiveness and abatement costs unaffected.

In Sections 7.4.1 to 7.4.3 it is implicitly assumed in each case that only a single policy goal is pursued. Therefore, all the costs of the policies are allocated towards this one environmental impact.

7.4.1 Fossil energy use

Table 61 shows the public expenditure, relative reduction in energy use, the corresponding cost-effectiveness and the abatement costs of the policies for Switzerland as a whole, *i.e.* all farms, without any differentiation by region or farm-type.

Table 61 Cost-effectiveness of organic farming compared to agri-environmental measures for reducing energy use

Indicator	Unit	Organic agri-culture	Extenso	Less intensive meadows	Extensive meadows	Combined AEM on all farms	Combined AEM on organic farms	Combined AEM on conven-tional farms
Public expenditure	CHF / ha	66.58	28.24	-0.49	12.76	73.17	23.11	78.62
Reduction of energy use	%	5.28	0.07	-0.11	1.38	1.50	3.76	1.38
Cost-effectiveness	% / 100 CHF	7.93	0.26	n.d.	10.80	2.04	16.29	1.76
Abatement cost	CHF / %	12.61	383.26	n.d.	9.26	48.94	6.14	56.88

n.d. = not defined

Source: own calculations based on Swiss FADN and SALCA data

With 66.6 CHF/ha of total Swiss UAA, public expenditure for organic farming exceeds public expenditure for the single policies. For these latter, public expenditure ranges from -0.5 CHF/ha for less intensive meadows to 12.8 CHF/ha for extensive meadows and to

28.2 CHF/ha for extenso payments. In combination, all agri-environmental measures account for public expenditure of 73.2 CHF/ha, which is slightly above the costs for organic farming.

The much higher public expenditure for organic farming is accompanied by much higher relative environmental effects in terms of energy reduction (5.3 %) compared to payments for extensive meadows (1.4 %) and the combination of the agri-environmental measures (1.5 %). Only marginal environmental effects were identified for extenso payments (-0.1 %) and payments for less intensive meadows (-0.1 %).

For organic farming the above mentioned indicators result in cost-effectiveness that is approximately four times higher and abatement costs that are four times lower than the combination of agri-environmental measures (2 %/100CHF; 48.9 CHF/% improvement). However, a higher cost-effectiveness and consequently lower abatement cost were calculated for the payments for extensive meadows (10.8 %/100 CHF and 9.3 CHF/% improvements respectively). Cost-effectiveness for extenso payments was 0.3 %/100CHF, resulting in theoretical abatement costs involving a 383.3 CHF/% improvement. Finally, cost-effectiveness of the less intensive meadows is not defined, as both public expenditure and relative reduction in energy use were negative.

7.4.2 Habitat quality

Similar to impacts on energy use, organic farming yields a 5.3 % improvement in habitat quality at sector level, *i.e.* relative to total UAA (Table 62). The effects of extenso payments (0.1 %) are at a similar level as for energy use. Less intensive meadows lead to a more marked decline in habitat quality of 2.2 %.

Contrary to energy use, payments for extensive meadows lead to an improvement in habitat quality of 18.4 %, which is an even slightly higher increase than for the combination of agri-environmental payments (18.1 %).

This high effectiveness of extensive meadows entails a calculative cost-effectiveness of 143.9 %/100CHF and ha. Accordingly, a theoretical improvement in the average habitat quality at 144 %/100CHF was possible. The cost-effectiveness of extenso payments, by contrast, is low, at 0.2 %/100CHF, while the cost-effectiveness of less intensive meadows is not defined. The cost-effectiveness of organic farming is, at 8 %/100CHF, about three times lower than the cost-effectiveness of the combined agri-environmental payments (24.7 %/100CHF).

The high cost-effectiveness of extensive meadows results in very low provision costs for improved habitat quality of 0.7 CHF/ha for a 1 % improvement. Habitat quality can also be improved at low costs (4.1 CHF/ha and % improvement) with combined payments, whereas the extenso payments entail costs of 573.1 CHF/% improvement in habitat quality. Finally, organic farming entails provision costs of 12.5 CHF/ha and % improvement, which are about three times higher than the provision costs of combined AEM support.

Table 62 Cost-effectiveness of organic farming compared to agri-environmental measures for improving habitat quality

Indicator	Unit	Organic agri-culture	Extenso	Less intensive meadows	Extensive meadows	Combined AEM on all farms	Combined AEM on organic farms	Combined AEM on conven-tional farms
Public expenditure	**CHF / ha**	66.58	28.24	-0.49	12.76	73.17	23.11	78.62
Improvement of habitat quality	**%**	5.34	0.05	-2.21	18.37	18.05	17.88	18.08
Cost-effectiveness	**% / 100 CHF**	8.02	0.17	n.d.	143.94	24.67	77.36	22.99
Abatement cost	**CHF / %**	12.47	573.11	n.d.	0.69	4.05	1.29	4.35

n.d. = not defined

Source: own calculations based on Swiss FADN and SALCA data

7.4.3 Eutrophication

Organic farming entails a relative reduction in eutrophication of 3.4 % per ha of total UAA (Table 63). Relative reductions in eutrophication are lower for all other policies and for the combination of AEM. Extenso payments lead to a reduction of only 0.2 %, whereas payments for less intensive meadows again result in negative environmental effects. However, extensive meadows show a remarkable decline in eutrophication of 1.9 % in the model. Finally, the combination of the three agri-environmental measures leads to a reduction in eutrophication of 2.2 %.

Cost-effectiveness values are highest for extensive meadows as a single payment (15.1 %/100CHF), while combined agri-environmental measures achieve only 3 %/100CHF. As for energy use, the cost-effectiveness of organic farming exceeds the combination of agri-environmental policies, at 5.1 %/100CHF as opposed to 3 %/100CHF. The cost-effectiveness of extenso payments is higher than for habitat quality and energy use but still to a marginal extent.

Expressed as abatement costs, extensive meadows are the most efficient measure, at 6.6 CHF/%. Organic farming follows, with 19.5 CHF/% lower costs than the combined agri-environmental payments (33.6 CHF/%). Extenso payments reduce eutrophication by 137.4 per CHF/%, while abatement costs for less intensive meadows are not defined due to the cost-saving effect of this measure.

Table 63 Cost-effectiveness of organic farming compared to agri-environmental measures for reducing eutrophication

Indicator	Unit	Organic agri-culture	Extenso	Less intensive meadows	Extensive meadows	Combined AEM on all farms	Combined AEM on organic farms	Combined AEM on conven-tional farms
Public expenditure	**CHF / ha**	66.58	28.24	-0.49	12.76	73.17	23.11	78.62
Reduction of total eutrohication	**%**	3.42	0.21	-0.03	1.93	2.18	3.22	2.10
Cost-effectiveness	**% / 100 CHF**	5.14	0.73	n.d.	15.10	2.98	13.95	2.68
Abatement cost	**CHF / %**	19.45	137.39	n.d.	6.62	33.60	7.17	37.37

n.d. = not defined

Source: own calculations based on Swiss FADN and SALCA data

7.4.4 Average cost-effectiveness

The average cost-effectiveness of the three indicators was calculated as a non-weighted mean according to Equations 15, 16, and 17 (page 105). The highest relative environmental effects have been found for the combined agri-environmental measures and extensive meadows (7.2 %) (Table 64). The relative environmental effect of organic farming is only slightly lower at 4.7 %. Average environmental effects of both extenso payments and less intensive meadows are insignificant.

The cost-effectiveness ratio, which has been calculated using the average relative improvement, is highest for extensive meadows (56.6 %/100CHF). Lower values were calculated for the combined agri-environmental measures (9.9 %/100CHF) and organic farming (7 %/100CHF). As for the single environmental impacts, the average cost-effectiveness for extenso payments is marginal. The cost-effectiveness of less intensive meadows is not defined.

Abatement costs are, consequently, lowest for extensive meadows at 1.8 CHF/%. The combination of agri-environmental measures costs 10.1 CHF/ha per % of environmental improvement, while organic farming costs 14.2 CHF/%. Abatement costs of extenso payments are highest, at 257.9 CHF/%, while the abatement costs for less intensive meadows are not defined.

Table 64 Average cost-effectiveness of organic farming compared to agri-environmental measures

Indicator	Unit	Organic agri-culture	Extenso	Less intensive meadows	Extensive meadows	Combined AEM on all farms	Combined AEM on organic farms	Combined AEM on conven-tional farms
Public expenditure	CHF / ha	66.58	28.24	-0.49	12.76	73.17	23.11	78.62
Average improvement	%	4.68	0.11	-0.78	7.22	7.24	8.29	7.19
Average cost-effectiveness	% / 100 CHF	7.03	0.39	n.d.	56.61	9.90	35.87	9.14
Average abatement cost	CHF / %	14.22	257.89	n.d.	1.77	10.10	2.79	10.94

n.d. = not defined

Source: own calculations based on Swiss FADN and SALCA data

7.5 Sensitivity analysis

The model that has been developed in order to generate the results presented in Sections 7.2, 7.3 and 7.4 was built on a set of assumptions. Therefore, the results for cost-effectiveness are only valid in relation to the above mentioned assumptions. Hence, in order to test the validity and stability of the main results, key assumptions related to the determinants of cost-effectiveness of the policies in the model were modified. This sensitivity analysis serves to provide additional information on the most important determining factors of cost-effectiveness.

The results of the sensitivity analyses are described for variations in payment levels (Section 7.5.1), policy uptake (Section 7.5.2), weightings of policy goals (Section 7.5.3) and number of policy goals (Section 7.5.4).

7.5.1 Variation in payment levels

Apart from the main scenarios in which payments are assumed to be abolished completely, payment levels were varied in order to investigate whether the main results regarding cost-effectiveness would change if the current payment rate was higher or lower. Moreover, this sensitivity analysis was conducted in order to confirm that the model is able to respond in a plausible way to different assumptions in the policy scenarios. Furthermore, it serves to analyse where on the cost-effectiveness curve (see Figure 13 on page 102, south-western quadrant) the current payment levels are located. If the current payment level is below PL_1, an increase in payment level results in an increase in cost-effectiveness of the payments, because the marginal environmental effects, increase with higher payment rates between 0 and PL_1. If the current payment level is beyond PL_2, increases in payment level will lead to lower cost-effectiveness, because the marginal environmental effects decrease with higher payment levels. If payment rates are between PL_1 and PL_2, both marginal effects and marginal environmental impacts will be almost constant, resulting in a stable cost-effectiveness for different payment rates.

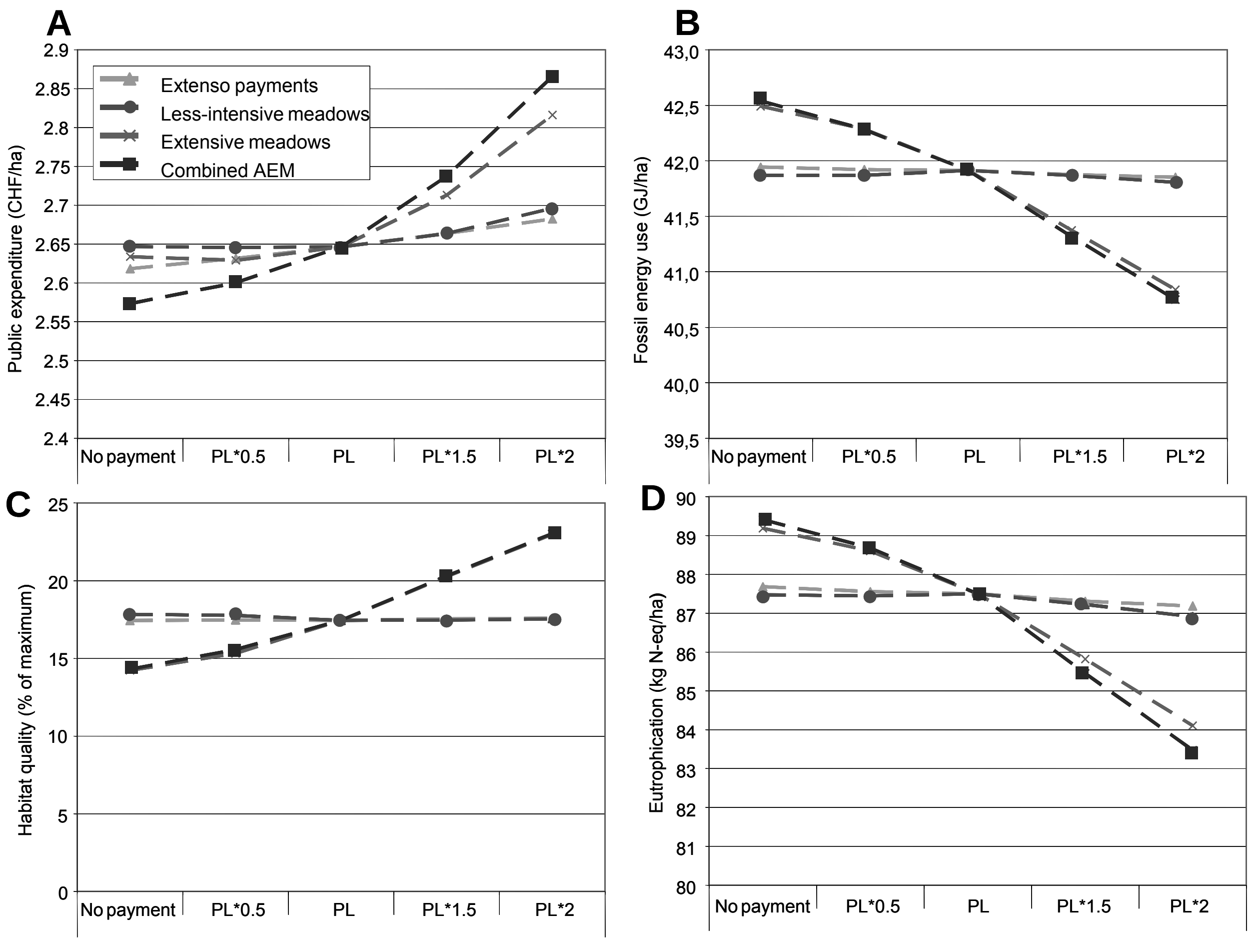
In addition to the standard situation in the base year and the complete abolition of the payments in the policy scenarios, a 50 % reduction, a 50 % increase and a 100 % increase in payment levels (PL) were imposed on each of the three agri-environmental payments and their combination.

The model responses to these scenarios regarding both average public expenditure per ha and average environmental effects are presented in Figure 30. The graphs show sound model responses for average public expenditure (Figure 30A). Total public expenditure rose with increasing payment rates for each of the four scenarios. As presented above, less intensive meadows were the exception, with slightly increasing public expenditure when abolished or reduced to 50 % of the base-year payment rate. However, for higher payments than in the base year, public expenditure rose slightly to 2.7 kCHF per ha when the payment rate was doubled.

Extenso payments showed a more gradual, minor increase as a function of payment rate. When payment rates were doubled, the response of total public expenditure was even lower than for less intensive meadows. All parameters responded in a generally plausible way. The responses were most sensitive to changes in payment rates for extensive meadows. This confirms that the payments for extensive meadows were also the main driver for combined changes in payment rates of all the agri-environmental measures analysed.

If payments for extensive meadows are reduced to 50 % rather than being abolished, public expenditure falls even more markedly due to the substitution effect with less intensive meadows (described in Section 7.3). Payments beyond the base-year level resulted in up to 2.8 kCHF per ha when the payment rates were doubled. As for variations in the combined agri-environmental payments, the model responded most sensitively and in an exponential way.

This suggests the conclusion with respect to Figure 13 (page 102) that the current payment levels of agri-environmental policies are higher than PL_1.



Source: own calculations based on Swiss FADN and SALCA data

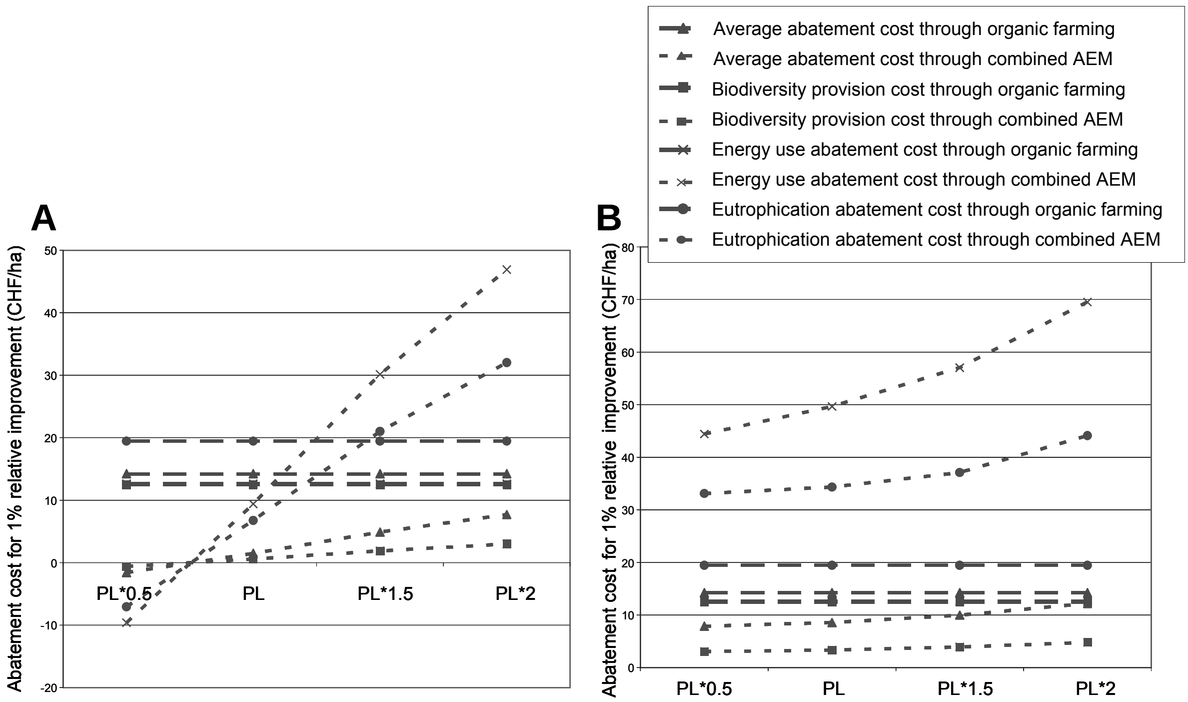
Figure 30 Responses of public expenditure (A), energy use (B), habitat quality (C), and eutrophication (D) to different payment levels for agri-environmental policy measures

The environmental indicators (Figure 30B-D), alike the parameter 'public expenditure', showed plausible responses to the different payment rates. While higher extenso payments resulted in a marginal decline of energy use per ha, less intensive meadows showed a marginally increasing energy use between zero payments and the base-year level (PL) (Figure 30B). Payment rates beyond the base-year level led to marginal decreases in energy use. Steeper decreases in energy use were the result of the increase in payments for extensive meadows by 100 % compared to base-year levels. However, energy use remained at 40.7 GJ/ha at a relatively constant level compared to total energy use.

Habitat quality (Figure 30C) responded positively to changes in payment rates for extensive meadows. Average habitat quality reaches up to 23 %, although the habitat quality responds in a completely inelastic manner to changes in payments for less intensive meadows and extenso payments. Almost the same responses as for extensive meadows were observed for simultaneous variations in all three payments, indicating the strength of impacts of the payments for extensive meadows as compared to the other policy measures.

The responses for eutrophication (Figure 30D) are equivalent to those for energy use. In the scenarios in which payments for extensive meadows are assumed to be doubled, eutrophication drops to 84 kg N-eq / ha as compared to its level of 87.5 kg N-eq / ha in the base year.

Figure 31 presents the responses of the total abatement costs to variations in payment levels for extensive meadows (Figure 31A) and the combined agri-environmental payments (Figure 31B).



Source: own calculations based on Swiss FADN and SALCA data

Figure 31 Responses of abatement cost parameters to different payment levels (PL) for extensive meadows (A) and combined agri-environmental support (B)

The model shows sound responses for both policies resulting in higher abatement costs with higher levels of agri-environmental support. This implies that the cost-effectiveness regarding each environmental indicator declines with higher payment levels due to more pronounced marginal increases in public expenditure than marginal increases in environmental effects. This can be explained by the larger increase in public expenditure than for the increases in environmental effectiveness with rising payment rates. This effect was found for all payments and implies that the current base year payment rates are either somewhere between PL_1 and

PL_2 or beyond PL_2 with respect to Figure 13 (page 102). Consequently, further increases in payment rates lead to relatively lower increases in environmental performance.

It should be pointed out that the theoretical abatement costs become negative when the payment rates are cut by 50 %. This is because, compared to the counterfactual situation, extensive meadows result in a lower total public expenditure when the payment is non-existent.

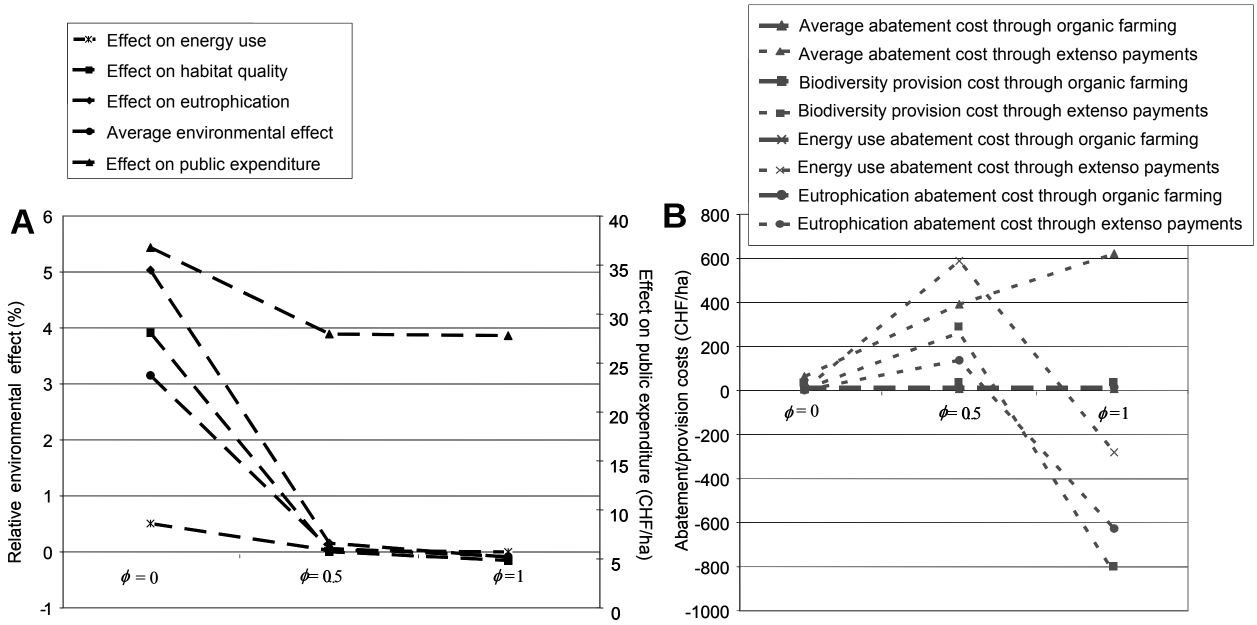
7.5.2 Variation in policy uptake

Uptake is one of the main determinants of cost-effectiveness. Using the Röhm-Dabbert approach (RDA), uptake sensitivity can be changed by adjusting parameter ϕ (described in Section 6.3.6). If parameter ϕ equals 1, the model responds as if the intensity levels were separate activities. If parameter ϕ equals 0, the model treats the intensity levels as a linear programme. Given that there were no empirical data and in order to validate the sensitivity adjustment of the model, a medium sensitivity of $\phi = 0.5$ was opted for.

For sensitivity analysis the parameter was set to both extreme points in order to observe the effect of the elasticity parameter on cost-effectiveness. Figure 32 shows the model's responses to changes in uptake elasticity for Scenario A, in which extenso payments are abolished. It reveals that variations in environmental effects and public expenditure to lower uptake sensitivity are marginal. Both effectiveness regarding all environmental categories and public expenditure remain almost constant (Figure 32A).

However, assuming a higher uptake sensitivity, additional public expenditure rises from 28 to 37 CHF/ha. Relative environmental effects rise from almost 0 to 5 % for eutrophication and 4 % for habitat quality. The relative reduction in fossil energy use, however, remains at a level below 1 %. These changes result in a strong decrease in abatement costs of the extenso payments when uptake is assumed to be more sensitive (Figure 32B). Abatement cost goes down to 11.6 CHF/ha in relation to the average of the environmental impacts. Thus, highly elastic uptake responses led to competitive abatement and provision costs of extenso payments.

At the same time, if uptake sensitivity is assumed to be low, abatement and provision costs rise (or even convert to negative values) as environmental effects become negative.

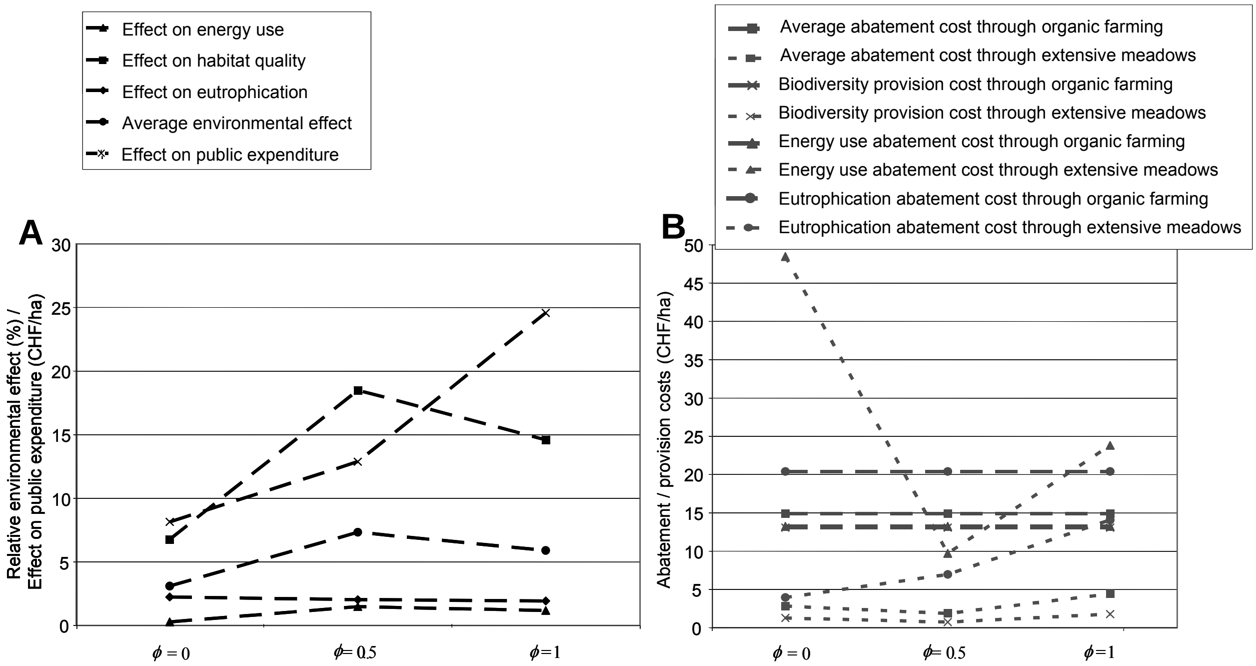


Source: own calculations based on Swiss FADN and SALCA data

Figure 32 Model responses to changes in uptake elasticity of extenso payments in terms of relative environmental effects (A) and abatement/provision costs (B)

Figure 33 deals with the model's responses to changes in uptake elasticity for Scenario C, in which payments for extensive meadows are abolished. With lower uptake sensitivity public expenditure rises from 8 CHF (ϕ = 0) to 24 CHF (ϕ = 1) (Figure 33A). By contrast, relative effects on habitat quality and energy use peak at ϕ = 0.5, whereas the relative effectiveness of extensive meadows regarding the reduction in energy use falls to 11.6 CHF/ha for an average improvement in the environmental indicators of 1 %. Eutrophication stays constant and is not susceptible to changes in uptake sensitivity.

This results in a high variability in abatement and provision costs regarding energy use, while the abatement and provision costs regarding habitat quality and eutrophication as well as average cost-effectiveness rise steadily with lower uptake sensitivity, as public expenditure experiences a steeper rise than environmental effects (Figure 33B).



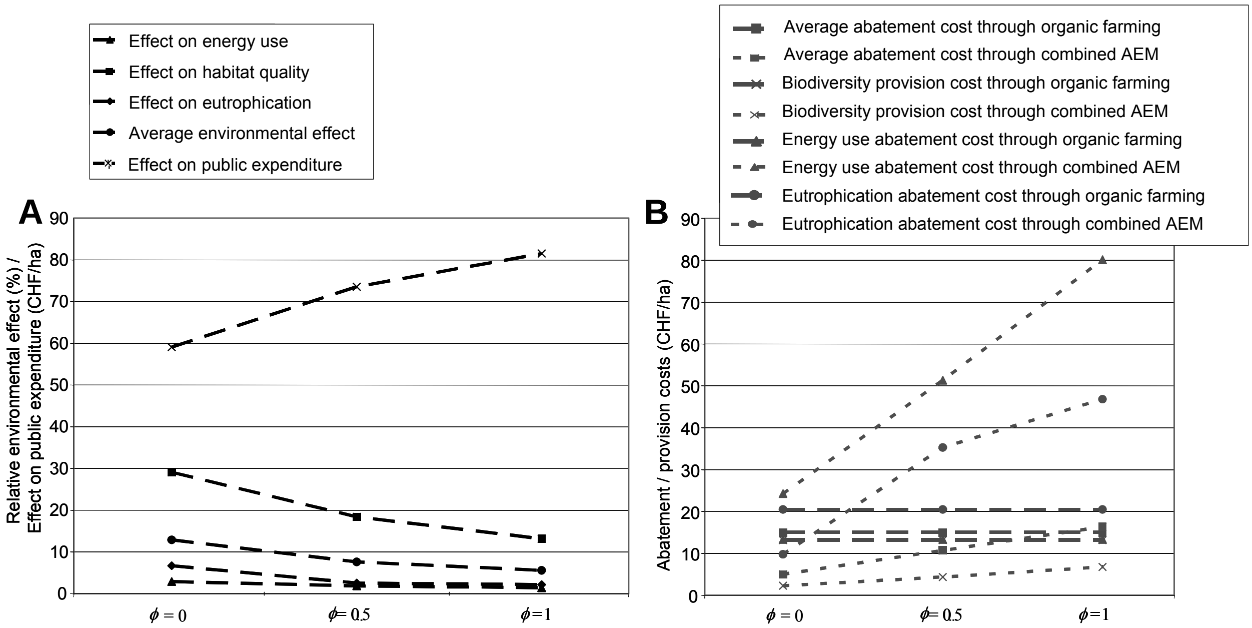
Source: own calculations based on Swiss FADN and SALCA data

Figure 33 Model responses to changes in uptake elasticity of payments for extensive meadows in terms of relative environmental effects (A) and abatement/provision costs (B)

Figure 34 shows the model's responses to variations in uptake elasticity in Scenario D, in which the three agri-environmental payments are abolished simultaneously. The graphs show rising public expenditure with lower uptake sensitivity, whereas the environmental effects fall gradually (Figure 34A).

The abatement and provision costs of the combined agri-environmental measures rise steadily from 23 to 76 CHF/ha for energy use, from 9 to 44 CHF/ha for eutrophication, and from 3 to 8 CHF/ha for habitat quality. This results in average abatement costs of 5 to 15 CHF/ha (Figure 34B).

In sum, changes in elasticity of uptake have a significant impact on both environmental effects and public expenditure. This results in a relative high variability of the cost-effectiveness of the different policy measures. The uptake of extenso payments in particular responds elastically if uptake sensitivity is increased.



Source: own calculations based on Swiss FADN and SALCA data

Figure 34 Model responses to changes in uptake elasticity of the combined agri-environmental payments in terms of relative environmental effects (A) and abatement/provision costs (B)

7.5.3 Variation in the weightings of environmental goals

Determining the overall cost-effectiveness of all policy goals is an awkward matter, since there are no data available on the relative importance of policy goals. This is why, in the main analysis, an equal weight was assumed for the three policy goals reduction in fossil energy use, improvement in habitat quality and reduction in eutrophication involving nitrogen and phosphorus. In the sensitivity analysis, the relative weights were varied. Apart from the standard weights, each of the environmental impact categories received double the weight of the other two.

Table 65 reveals that the ranking of policy measures in terms of cost-effectiveness remains stable, no matter which weighting is chosen. This means that the highest abatement costs were calculated for extenso payments. The lowest costs were obtained for extensive meadows as a separate measure.

Table 65 Results of the sensitivity analysis regarding the weights of agri-environmental goals

Indicator	Unit	Energy use (%)	Habitat quality (%)	Eutrophi-cation (%)	Organic agriculture	Extenso	Less intensive meadows	Extensive meadows	Combined AEM
		Relative weight of impact (%)							
Average improvement	%	*33*	*33*	*33*	4.68	0.11	-0.78	7.22	7.24
	%	*50*	*25*	*25*	4.83	0.10	-0.61	5.76	5.80
	%	*25*	*50*	*25*	4.85	0.09	-1.14	10.01	9.94
	%	*25*	*25*	*50*	4.37	0.13	-0.59	5.90	5.98
Average cost-effectiveness-ratio	% / 100 CHF	*33*	*33*	*33*	7.03	0.39	n.a.	56.61	9.90
	% / 100 CHF	*50*	*25*	*25*	7.25	0.36	n.a.	45.16	7.93
	% / 100 CHF	*25*	*50*	*25*	7.28	0.33	n.a.	78.44	13.59
	% / 100 CHF	*25*	*25*	*50*	6.56	0.47	n.a.	46.23	8.17
Average abatement or provision cost	CHF / %	*33*	*33*	*33*	14.22	257.89	n.a.	1.77	10.10
	CHF / %	*50*	*25*	*25*	13.78	280.86	n.a.	2.21	12.61
	CHF / %	*25*	*50*	*25*	13.74	299.01	n.a.	1.27	7.36
	CHF / %	*25*	*25*	*50*	15.25	211.51	n.a.	2.16	12.25

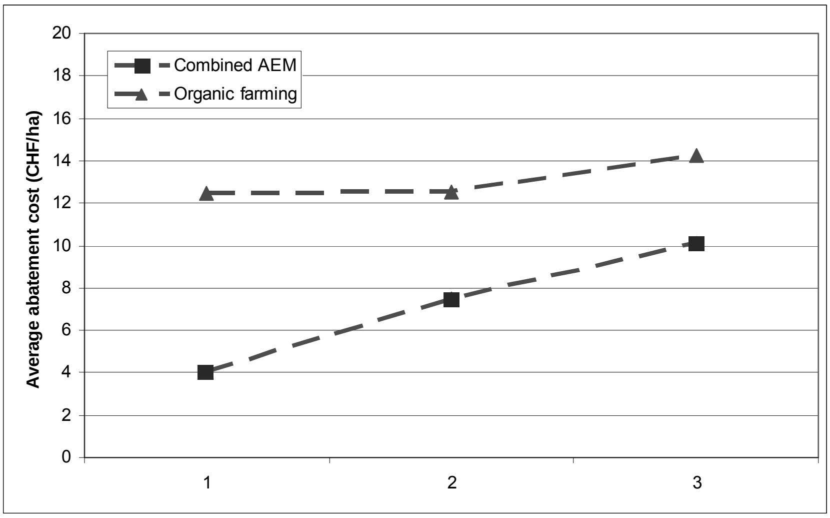
n.d. = not defined

Source: own calculations based on FADN and SALCA data

The combination of AEM gave rise to lower abatement costs than for organic farming, with all the varied weight distributions chosen. While the cost-effectiveness of organic farming remained stable between 13.7 CHF (assuming the weight for habitat quality was double) to 15.3 CHF (assuming the weight for eutrophication was double) over all weightings, it fluctuated significantly between 7.4 CHF (assuming the weight for habitat quality was double) and 12.6 CHF (assuming the weight for energy use was double) for the combined AEM.

7.5.4 Variation in the number of environmental goals

In addition to sensitivity analyses in respect of the weighting of the goals, the number of policy goals was varied in order to examine the model's response if one, two or three policy goals were considered. Figure 35 presents the cost-effectiveness of organic farming and the variation in cost-effectiveness of the other agri-environmental support measures in combination. The figure demonstrates comparatively constant abatement/provision costs of 12 to 14 CHF / ha for organic farming, while for the combined AEM the costs rise the more policy goals are considered. This response occurs because organic farming pursues all the environmental impacts analysed at a similar cost between 12.5 and 19.5 CHF/ha per % of improvement, while the costs of AEM for pursuing the goals are more variable, between 4.1 and 48.9 CHF/ha per % of improvement.



Source: own calculations based on Swiss FADN and SALCA data

Figure 35 Abatement cost for organic farming and combined agri-environmental payments depending on the number of policy goals

The various sensitivity analyses showed that the results are relatively stable even if different assumptions are made regarding payment levels, policy uptake and number and weights of environmental goals. The most sensitive model responses were found in relation to the Röhm-Dabbert approach. Depending on which uptake sensitivity is assumed in the model, both relative effects and public expenditure vary. Thus, the sensitivity analyses did not confirm a definitely higher level of cost-effectiveness of the combined agri-environmental measures.

7.6 Summary of the chapter

Conventional and organic farms were compared a) for Switzerland as a whole, b) by region (lowlands, hills, mountains) and c) by farm type (dairy, suckler cow and mixed farms). Comparisons by farm also included other grassland farms and speciality crop farms in the base year, although no cost-effectiveness indicators were calculated for the latter types due to the weak representation of organic farm groups in the FADN sample. The comparisons included structural, financial and environmental parameters.

The cost-effectiveness of organic farms was derived by comparing the above mentioned farm groups (as the 'treatment') with their respective conventional counterparts (as the 'counterfactual'). For this purpose, structural differences, financial performance, policy uptake, environmental performance and public expenditure for the farming systems were analysed.

The cost-effectiveness of agri-environmental policies was derived by modelling policy scenarios in which the respective payments were abolished. The additionality of the policy measures was obtained by comparing the above mentioned indicators in the policy scenarios (as the 'counterfactual') with the indicator values in the base year ('treatment').

The comparisons of structural and financial indicators revealed notable differences between organic and conventional farms. This necessitated the region and farm type-specific comparisons, which largely levelled off the structural gaps, particularly for the farm-type comparisons. However, important structural differences in terms of cost-effectiveness remain regarding a) a higher uptake of agri-environmental policies, namely extenso and ECA measures, and b) a lower stocking density, most prominently for pig and poultry livestock.

Organic farms had, on average, 54 %, lower energy use per ha than their conventional counterparts. This gap became smaller as farm altitude became higher (lowlands 49 %, mountains 27 %). At 38.2 % mixed farms showed the most significant differences among the farm types.

These differences could be attributed mainly to the lower purchase of fodder, the lower animal production-related emissions (including buildings), and the ban on mineral fertilisers.

The habitat quality on organic farms was higher for all the farm groups being compared. On average, habitat quality on organic farms exceeded the scores on conventional farms by 55 %. The biggest differences were found on suckler cow farms (86 %) and in the lowlands (56 %).

Organic farms had 35 % lower eutrophication rates than conventional farms. Eutrophication involving nitrogen was 37 % lower, whereas eutrophication involving phosphorus was 11 % lower. Differences in nitrogen eutrophication rates were particularly notable for hill and mountain regions (22 % each) and for suckler cow farms (32 %). Eutrophication involving phosphorus differed most in the lowlands (33 %) and for mixed farms (27 %).

Public expenditure for organic farms was higher than for conventional farms. Public expenditure per ha on organic farms was 3,265 CHF/ha, compared to 2,579 CHF on conventional farms. The differences in public expenditure are variable between farm types and regions. The largest differences were calculated for the lowlands (24 %) and mixed farms (31 %). Public policy-related transaction costs are at a similar level for both farming systems. However, farm-level policy-related transaction costs are higher on organic farms, due to the costs of private certification.

The abatement costs of organic farms regarding energy use are 12.6 CHF per ha and year. Habitat quality provision costs are 12.5 CHF per ha. Eutrophication was abated with organic agriculture for 19.5 CHF/ha. Abatement costs vary markedly by region and farm type. The highest abatement/provision costs were found for mixed farms and in the lowlands. On suckler cow farms and in the mountain regions improvements were achieved at the lowest costs.

Of the agri-environmental measures analysed (extenso payments, payments for less intensive meadows and payments for extensive meadows), the measure that performed best in terms of cost-effectiveness was extensive meadows. Both extenso and less intensive meadows rendered only small or no environmental improvements and therefore had high abatements costs. The combination of these agri-environmental measures led to an improvement in the indicators – triggered predominantly by the payments for extensive meadows – of 1.5 % (energy use), 18 % (habitat quality) and 2.2 % (eutrophication) at an additional cost of 73 CHF per ha.

The cost-effectiveness of the agri-environmental policies was higher on organic farms than on conventional farms, except for habitat quality. Thus these policies gave rise to improvements in the environmental categories at far lower costs on organic farms than on conventional farms.

Organic farming gave rise to slightly higher average abatement costs (14.2 CHF/ha) compared to the combination of agri-environmental measures with 10.1 CHF/ha. However, extensive meadows had the highest cost-effectiveness, with abatement costs of only 1.8 CHF/ha. The cost-effectiveness of less intensive meadows was not defined because the policy, as currently implemented, actually resulted in negative environmental effects.

A sensitivity analysis revealed that the results are relatively stable even if assumptions are varied. Only variations in uptake elasticity revealed a significant influence on the ranking of the policy measures. Assuming a more elastic uptake response, the model calculated a higher cost-effectiveness of the measures, while the assumption of more inelastic uptake led to lower cost-effectiveness. When the Röhm-Dabbert approach is not applied, *i.e.* assuming a standard elasticity of the policy measures, the combination of AEM leads to slightly higher abatement (15.5 CHF) and provision costs than organic farming.

8 Discussion

This chapter serves to discuss the research approach and the results of the previous chapters. Section 8.1 addresses the methods used in this thesis. The main aim here is to identify the strengths and limitations of a) the general modelling approach and b) the specific determinants of cost-effectiveness. Section 8.2 focuses on the results generated by using the research approach. It discusses the validity, relevance and generalisability of the results against the background of a) the methodological limitations of the approach and b) the existing body of literature.

8.1 Discussion of the research approach

This section is structured according to the main features of the approach. First, it addresses the general modelling approach, particularly the level of analysis and the comparability of farm groups. Second, it discusses aspects related to the determinants and the calculation of cost-effectiveness, including the general conceptual cost-effectiveness model, modelling policy uptake, environmental effects and public expenditure.

General cost-effectiveness framework

This study employed a cost-effectiveness framework rather than a cost-benefit framework for the economic evaluation of organic farming. The framework chosen can be regarded as suitable for the research question: **How cost-effective is organic farming in providing environmental services under the current Swiss agricultural policy scheme?** In Section 2.2.2, it was stated that a cost-benefit analysis (CBA) is not required in pursuing the research objective, as the question is not whether or not agri-environmental policies should be implemented, but which policy measures or mixes of policy measures deliver more environmental benefits in relation to public money spent. Therefore, only relative weightings and not absolute weightings of different environmental impacts were necessary. However, welfare economists in particular stress the necessity of expressing benefits in monetary terms. Especially in the UK and US CBA evaluations of agri-environmental programmes are more common than in continental Europe, including Switzerland (EC, 2005; Jones-Walters and Mulder, 2009; Pearce, 2005). Particularly with regard to multi-objective policies like organic farming, monetisation is a complex procedure, which can probably be conducted in a manageable way only by using methods of benefit transfer.

However, the relative weighting of environmental indicators for considering multiple environmental effects is a further methodological challenge, since either there are no applicable results available from prioritisation studies or those that exist show highly variable results. For instance, Huber *et al.* (2007) found different prioritisations in rural and urban areas of Switzerland. Schader *et al.* (2009a) also identified heterogeneous demand patterns for non-commodity outputs in different European regions. Therefore, instead of allocating specific weights for each environmental indicator, equal weights were allocated to the effects and varied in a sensitivity analysis. The sensitivity analysis revealed that different weights did not affect the main findings of the comparison of cost-effectiveness.

Instead, changing the number of policy goals had a more marked impact on cost-effectiveness. This may lead to the conclusion that for an exact calculation of the cost-effectiveness of organic farming, it is more important to include further environmental impact categories than to identify exact weights. The methodological works in this thesis constitute a good basis for including further environmental impact categories. Besides a reduced use of fossil energy, improvements in habitat quality and a reduced eutrophication with N and P also other environmental impacts, in particular eco-toxicity and global warming potential would be relevant with respect to organic farming. With more environmental impact categories, however, the sensitivity analysis regarding the weights of environmental impact categories becomes more difficult. Uncertainty analysis provides an efficient alternative to sensitivity analysis, since the results for many different weightings can be tested in a very short time. The distribution of weights could even be defined according to available empirical WTP data. An alternative approach to WTP weightings could be the distance-to-target approach, which weights the impact categories according to the distance of the current environmental state from the targeted one (Wenzel *et al.*, 2000).

Level of analysis

Contrary to existing evaluation approaches, the level of the analysis was set to sector-representative farm groups, which were average farms formed on the bases of FADN, FSS, and normative data and were representative of a number of real farms. This approach is advantageous because it can lay claim to sector representativeness on the one hand and allows of an intuitive interpretation of results on the other, as these can be expressed at farm level (Jacobs, 1998).

In order to use the model for sector-level energy and nutrient balances, either the sector should not be grouped into farms but optimised as a whole, or the interactions between farm groups need to be covered explicitly, as is the case in multi-agent models. Since optimising the sector as a whole would contradict a flexible stratification of farms into relatively homogeneous groups, applying multi-agent modelling features, could be an elegant solution. Such an adaptation of the extended FARMIS model seems to be most important for purchased fodder, due to its major significance in relation to environmental impacts. Solutions similar to those implemented by Bertelsmeier (2005) or Happe *et al.* (2006) would make it possible to model such relations.

Environmental problems, particularly regarding eutrophication and habitat quality, depend largely on site-specific characteristics. It was thus necessary to make some rough assumptions for the purpose of working at sector-level. For this reason -, and despite the availability of a large amount of data for this model – site- and farm-specific analysis, as performed by Alig and Baumgartner (2009), for example, can complement sector-level analyses for a full understanding of the site- and farm-specific impacts of different policies. Nevertheless, in contrast to linear programming models, PMP calibration can be used to take account of site- and farm-specific aspects indirectly via hidden costs.

Furthermore, the PMP is superior to linear programming models as it solves the problem of overspecialisation and calibrates precisely according to an empirically founded base year period without any calibration constraints (Howitt, 1995; Umstätter, 1999). At the same time, PMP is criticised for its lack of empirical validation, as the base year period which determines the level of the shadow prices of the model activities usually stems from a single observation, rather than a number of observations, which would be necessary for an econometric estimation (Heckelei, 2002). Therefore, one proposal emerging from this is that the elements – also non-diagonal elements – of the Q-matrix should be estimated econometrically, based on data from several years.

Comparison of farm groups

A further important general feature is the comparison of farm groups; existing sector-level models commonly neglect this type of sector differentiation. Most models permit only regional comparisons. However, only a few of the approaches developed so far differentiate between organic and conventional farms (Offermann *et al.*, 2009; Sanders, 2007; Schmid and Sinabell, 2005).

The level of aggregation of farm types and regions chosen for the study and the comparisons facilitated by this is a compromise resulting from the trade-off between representativeness and differentiation. The comparisons by region and farm type are particularly useful for levelling off differences in the distribution between organic and conventional farms among regions and farm types. Thus this grouping permitted a basic comparability of organic farms. However, while the regional differentiation into three groups permitted a full representation of all classes, not all farm-type strata could be modelled. Particularly specialised crop production farms (arable and speciality crop farms) could not be modelled at all or only with an insufficient number of organic sample farms from the Swiss FADN.

Moreover, due to this low number of organic farms in each FADN strata, it was not possible to conduct a simultaneous comparison by region and farm type. The remaining structural differences can be fully eliminated only by econometric methods (Frondel and Schmidt, 2005), such as propensity score matching (Pufahl, 2007), which derive the unknown 'counterfactual' according to a combination of different criteria. The selection of these criteria, however, is another challenge, because it is difficult to identify clearly those criteria which are exclusively caused by the self-selection bias but do not contain differences which could be attributed to farming system-inherent factors. Furthermore, it is unlikely that a matching procedure would generate fundamentally different results because in the present analysis the cost-effectiveness figures in the different types of comparisons (Switzerland as a whole, by region and by farm-type) were relatively stable.

Nevertheless, the comparison of farming systems across all farm types and regions proved to be useful for the given research problem. As this study has shown, it is crucial to include regional farm distributions for evaluating the environmental impacts of organic farming. Moreover, in product-related LCAs such considerations are also beneficial. For instance, if organic meat or dairy products are compared with each other, the comparison should ac-knowledge that an organic product will be more likely to come from the mountain region instead of from the lowlands than a conventional product. Furthermore, the organic product will more likely be from a mixed farm than from a specialised dairy or suckler cow farm. Finally, the organic product will most likely be produced on a farm with a lower farming intensity, in terms of stocking density, and on more extensively cultivated meadows. Therefore, the approach used can be described as comparing the average organic farm with the average non-organic farm as they can be found currently in Switzerland, rather than quantifying the exact environmental effect of organic agriculture as such.

Derivation of cost-effectiveness

Conceptual model of cost-effectiveness

In Chapter 6, a simplified framework for cost-effectiveness at sector level was developed in order to show the interlinkages between the main determinants of cost-effectiveness at sector level, which are uptake, environmental effects and public expenditure. This new framework proved to be a useful tool for understanding the dependencies between and theoretically deriving the consequences of variations in payments levels, uptake, environmental effectiveness and public expenditure. However, two important aspects have been identified in the model analysis that the conceptual framework does not consider: First, only a single policy measure is taken into account. This means that positive and negative feedback effects with other policy measures are not considered. For instance, any two policies may have mutually enhancing or hindering effects. Second, other relevant indirect effects of policies found with the FARMIS model were neglected in the theoretical framework. For instance, extenso payments increase not only the relative profitability of extenso grains and rape compared to intensive grains and rape but also the relative profitability of total grains and rape as compared to more environmentally friendly activities like extensive grassland.

However, given that the three determinants of cost-effectiveness are addressed by existing literature in largely isolated form (Dupraz, 2007; Falconer and Saunders, 2002), this integrated framework offers a useful basis for conceptualising the cost-effectiveness of agri-environmental policies.

As a further contribution to economic theory, this thesis has been able to clarify the implications of the Tinbergen Rule for organic farming and multi-objective policies in general by designing a small linear programming model which minimises public expenditure under consideration of fixed policy targets. It is particularly notable that the model is built on very simple assumptions which can be derived from existing literature. Due to its simplicity, the model can be a very useful tool for many different research questions concerning the analysis of co-benefits (Feng and Kling, 2005) and economies of scope, not least in the context of the multifunctionality of agriculture (OECD, 2001a).

The comparison of the cost-effectiveness of organic farming with the cost-effectiveness of agri-environmental measures was conducted by applying two different procedures. The cost-effectiveness of organic farming, as a farming system[75], was derived by comparisons of entire farms by region and farm type. Thus the approach defined the total public expenditure for organic farms as the budget spent on organic farms additionally to the spending for conventional farms. The assumption made here was that if organic farms were non-organic, they would cultivate their land in the same way as the respective equivalent conventional farms. A calculation of the 'counterfactual' by econometric means could be a methodological alternative to this assumption. The cost-effectiveness of agri-environmental policy measures was obtained by calculating policy scenarios in which the respective policy measures were abolished. The negative differences in environmental effects between the base year and the policy scenario were interpreted as the additionality or the policy outcome, while the difference in public expenditure represented the additional costs entailed by the policy measure.

It is important to note that, in taking this approach, the cost-effectiveness of organic farming is understood as a function of the combination of all agri-environmental policy measures, rather than the impact of organic farming area support payments only. Therefore, the calculated cost-effectiveness of organic farming comprises the effects and costs of the taken up agri-environmental policies indirectly. In other words, the higher uptake rates of organic farms are conceptualised as farming system-inherent features of organic farming.

Using this approach had the advantage of avoiding the need to model conversion from and to organic farming, in addition, the whole farming system was considered, rather than taking into account only support payments specific to organic agriculture (OFASP). This is particu-

[75] In Switzerland, organic farming addresses the farm as a whole. With very few exceptions, staged conversion is not applied.

larly important because in Switzerland organic farming area support payments constitute only a minor share (8 %) of total ecological direct payments and only 1.2 % of the total direct payments. Furthermore, the current structural differences between organic and conventional farms in Switzerland cannot be attributed solely to OFASP but to the combined framework of ecological direct payments.

In this regard, it is significant that the levels of environmental effectiveness on the agricultural sector as a whole are similar to organic farming and the combined agri-environmental measures. Otherwise, the indicator 'cost-effectiveness' can be misleading, as a measure can be cost-effective but still lead to only marginal environmental effects, if it also entails only marginal cost increases. At the same time, a measure may have low cost-effectiveness but lead to very pronounced environmental effects. Thus, when comparing measures with very different levels of effectiveness, a linear relationship between environmental effects and uptake has to be assumed (see Figure 12A, page 101).

Nevertheless, when dividing the total additional public expenditure for organic farms by total UAA, rather than by the UAA of all organic farms, they were at about the same level as the combined agri-environmental payments considered in this study. This implies that the cost-effectiveness in each case is comparable. However, it should be noted that in the present study organic farming was compared to only one single combination of agri-environmental measures. As discussed in Chapter 2, different policy options could be analysed using the established approach. It should be mentioned, however, that the PMP approach has important methodological limitations when modelling new policy measures, as the shadow prices used in the scenarios are based on empirical data in the base year period (Heckelei and Britz, 2005). Therefore, only existing policies were taken into account for the comparison with organic farming in this study.

Modelling policy uptake

Coverage of policy uptake is a key feature of this approach and was elaborated especially for the research question. The uptake of three important agri-environmental measures was included in the model: extenso payments, payments for less intensive meadows, and payments for extensive meadows. However, as the assumption of uptake elasticity proved to be a crucially determining factor in the evaluation of the extenso payments, an econometric estimation of the elasticity parameter for the Röhm-Dabbert approach seems advisable.

Conversion to organic farming, which could be understood as 'uptake of organic farming' if organic farming is conceptualised as an agri-environmental policy, was excluded from the model approach, as reliable concepts for modelling the conversion to and from organic farming are not yet available. The available body of literature implies that conversion should be examined further by both qualitative means and by econometric modelling, taking into account investment dynamics and soft factors such as personal attitudes or personal product price expectations (Kerselaers *et al.*, 2007; Odening *et al.*, 2004; Padel, 2008). The knowledge thus generated could be used in a second step to model conversion to organic agriculture using mathematical economics (Hollenberg, 2001). Currently, a straightforward way of addressing the conversion issue in a mathematical programming model is via assumptions and sensitivity analysis. For the development of such assumptions a recent empirical farm survey on conversion in Switzerland conducted by Ferjani *et al.* (2009) might deliver sufficient data.

Modelling environmental effects

Three different environmental impact categories – fossil energy use, biodiversity and eutrophication – were modelled from a sector-level perspective. The approach showed that plausible results were generated for all three impact categories analysed. However, due to the broad scope of the analysis, the degree of detail cannot be similar to that in a site- or farm-specific analysis. So it was necessary to base farm management (e.g. levels and methods of fertiliser application) on normative data. Nevertheless, the relative environmental effects can be considered valid, since potentially unidentified systematic flaws affect both organic and conventional farming systems. Furthermore, the environmental data used for the model can be regarded as being of good quality in terms of both representativeness for Swiss agriculture and consistency between the environmental impact categories, compared to alternative data sources (Nemecek and Erzinger, 2005).

Technically, the environmental indicators were linked largely to the model activities and intensity levels. Thus the indicators are influenced primarily as a function of farm groups opting for different activities and intensity levels. This approach is tailored to the research question, since the direct payments affect primarily the relative profitability of farm activity. Due to the quantitative importance of meadows and the implications for stocking density, the shift between the different intensities of meadows received most attention. Since stocking density is an important determining factor, quantities of purchased fodder were based on a maximum entropy model which calculated the fodder needs of farms in the FARMIS feed-module.

However, model-endogenous differences in fertiliser use were not considered for estimating the environmental effects. Instead, the impact assessment was based on the LCA inventory data from the representative SALCA activity data. The main advantage of this procedure was that the environmental impacts were based on the same assumptions regarding farm management. The differences between farming systems in terms of environmental performance can be regarded as conservative compared to alternative LCAs. The main disadvantage of this procedure was a less flexible model response. However, a direct linkage would not have been possible for all three environmental impact categories within a reasonable time frame, due to the complexity of the ecological models. For example, the biodiversity indicator is determined by very detailed farm management specifications, about 3,000 options in total. As a sector-level model, FARMIS is not able to represent farm activities in such detail.

The way environmental indicators were linked to the model implies a linear relationship between uptake and effect, or a constant marginal increase. Since this is particularly questionable for local and regional environmental indicators, such as biodiversity and eutrophication, the indicators were expressed as hectare averages. Average habitat quality is an environmental impact category which was modelled here at sector level for the first time. The indicator performed well in terms of interpretability and relevance. The indicator values could be attributed logically to the modelled farm responses and to structural differences between organic and conventional farms.

Modelling public expenditure

As shown in Chapter 2, different approaches exist for assessing the costs of agri-environmental policy measures on the one hand and organic farming on the other. Public expenditure was the key component for modelling the abatement costs of agri-environmental policies. Thus public expenditure was understood as the costs entailed by policy measures in order to solve or prevent an environmental problem or to improve environmental performance. As a new parameter in sector models policy-related transaction costs were explicitly included in the calculation of cost-effectiveness. Labour costs as the most important component of policy-related transaction costs were considered. Infrastructure (e.g. costs for buildings or consumables) was neglected due to the relatively low quantitative importance of policy-related transaction costs compared to the high payment rates in Switzerland. Furthermore, the transaction cost data are relatively uncertain as a) they were gathered some time ago in 2005, and b) they are only estimates which refer to experts' assessments from a limited number of cantonal and federal administrations. Nevertheless, against the background of the limited relevance of policy-related transaction costs in Switzerland in absolute terms, the data can be considered adequate.

The approach chosen for calculating public expenditure alone, rather than taking into account producer and consumer surplus, does not follow the welfare economic framework. However, it is useful for analysing policies from a budgetary perspective, as the approach was used for calculating the amount of public money spent on generating a particular level of environmental impacts through different policy measures.

Strengths and limitations of the approach

The research approach developed in this thesis represents a new method for analysing both the cost-effectiveness for organic farming and of agri-environmental measures, covering policy uptake, environmental effects and public expenditure. In addition, a simple theoretical model was developed to show that multi-objective policies do not necessarily contradict the Tinbergen Rule.

By way of a summary, the particular methodological strengths of this thesis are:

- The design of an integrated conceptual model of cost-effectiveness of agri-environmental policies using the determinants policy uptake, environmental effects and public expenditure.

- Design and application of a linear programming model for analysing the implications of the Tinbergen Rule for multi-objective policies, such as organic farming area support payments.

- The novel linkage between a farm-group specific PMP approach and life cycle assessments for analysing the cost-effectiveness of organic farming and agri-environmental measures. Average habitat quality as an impact indicator for biodiversity has not been modelled before in a similar methodological framework.

- The transfer of the Röhm-Dabbert approach into CH-FARMIS for modelling agri-environmental policy measures with known uptake levels in the base year.

- The consideration of policy-related transaction costs into the model framework for a full assessment of public expenditure related to the payments.

The following aspects are regarded as the most important methodological limitations:

- One general drawback is the lack of a sufficient empirical foundation for deriving shadow prices when using a PMP approach, including the limited empirical basis of the uptake elasticity of the agri-environmental measures modelled.

- The exclusion of conversion to and from organic agriculture from the modelling framework and the use of strong, simplifying assumptions regarding the self-selection bias of organic farms in the general comparison and in the comparison by farm type and region.

- The limited number of environmental impact categories considered (fossil energy use, biodiversity and N and P eutrophication) and the limited number of agri-environmental instruments taken into consideration for the model analysis.

8.2 Discussion of the results of the thesis

This section discusses the validity, relevance and generalisability of the results against the background of a) the methodological limitations of the approach and b) the existing body of literature.

Corresponding to the structure of Chapter 7, the results pertaining to the cost-effectiveness of organic farming are discussed first, followed by those pertaining to the cost-effectiveness of the agri-environmental measures. The final section addresses the comparison between the cost-effectiveness of organic farming and the cost-effectiveness of agri-environmental measures.

Cost-effectiveness of organic farming

In this section, the most important results regarding a) farm structure and financial performance, b) fossil energy use, c) biodiversity, d) eutrophication, e) public expenditure, and f) abatement cost are examined.

Farm structure, financial performance, and uptake

Since there are noticeable differences in the distribution of organic and conventional farms, specific comparisons by region and farm type were required. These comparisons widely levelled out structural gaps between the farming systems. The remaining structural differences influencing the cost-effectiveness of organic farming are a) a higher uptake of agri-environmental policies, namely extenso and ECA measures and b) a lower stocking density, most prominently for pig and poultry livestock, on organic farms compared to conventional farms. These differences can be understood as a further determinant of cost-effectiveness alongside policy uptake, effectiveness and public expenditure.

These results contradict assertions that there are only marginal differences between organic and conventional grassland farms compared with arable farms (Bengtsson *et al.*, 2005; Köpke, 2003; Pfiffner and Luka, 2003). Structural differences, including differences in policy uptake, were shown to be quite pronounced in this study, particularly for farm types which supposedly work extensively in general (e.g. suckler cow farms). The same applies to the regional comparison; the most pronounced differences in policy uptake between the farming systems were found in mountain regions, which already work relatively extensively, even in conventional systems.

Higher uptake levels of agri-environmental measures on organic farms in the model are confirmed by studies which analyse statistical datasets. For instance, Jurt (2003) found higher uptake levels on organic farms additionally for other agri-environmental measures such as hedges and 'extensive meadows on wet sites' when analysing farm structure data. These differences can be only partly attributed to economically rational behaviour, because the opportunity costs of these measures should be of a similar magnitude for both farming systems. It can therefore be hypothesised that these differences in uptake levels are partly farming-system inherent, as the lower stocking rate and the higher grassland share enable a lower stocking density.

Moreover, qualitative studies suggest a different understanding of nature conservation among organic farmers compared to conventional farmers. While conventional farmers interpret nature conservation as activities particularly relevant on areas excluded from production, organic farmers see their whole farm – including the productive areas – in terms of nature conservation (Jurt, 2003; Stotten, 2008). This suggests a different attitude towards nature conservation, which could be a reason for the greater policy uptake by organic farms.

However, the model is built mainly on FADN sample data rather than on population data. Therefore, the results may be expected to show a certain amount of bias on the basis of deviations in the FADN sample. However, as comparisons with the above studies and FSS data have shown, these deviations are not severe and do not affect the main results of this thesis. Furthermore, uptake levels of only a selection of agri-environmental measures were analysed.

Nevertheless, the modelled differences in uptake of agri-environmental policies between the farming systems are plausible and in line with current scientific literature and farm structure surveys. However, the notable differences between the farming systems regarding grassland intensity contradict the findings of previous studies.

Fossil energy use

According to the results of the model used in this study, organic farms have, on average, a 54 % lower energy use per ha than their conventional counterparts. This gap becomes smaller with higher farm altitude (lowlands 49 %[76], mountains 27 %). Mixed farms show the most significant differences among the farm types, with a 38 % lower fossil energy use on organic farms. These differences can be attributed mainly to the lower purchase of feedstuffs, the lower animal production-related emissions (including buildings), and the ban on mineral fertilisers in organic systems.

A large proportion of the relative difference in energy use per ha between the farming systems can be attributed to differences in the allocation of farms among regions and farm types. For

[76] This figure is lower than the reduction across all farms due to the higher concentration of organic farms in the mountain regions where energy use is lower in general and on farm types with a generally lower energy use per ha (dairy farms and suckler cow farms).

example, organic farms are more concentrated in the mountain areas, where energy use is generally lower. Furthermore, organic lowland farms do not include specialised pig and poultry farms, which are the primary consumers of concentrate feedstuffs. Nevertheless, as the greater pig and poultry stocking density is also evident in the comparison by farm type, this is conceived as a farming-system inherent characteristic.

The results on reduction in fossil energy use of organic farms are in accordance with existing literature, as similar reduction rates can be found in studies analysing farm-level energy use. Nemecek and Gaillard (2004) calculated a slightly higher energy use on average for both conventional and organic farms. Furthermore, the difference in energy use per ha between the farming systems was less distinct (33 %) than the values calculated in the present study. The relative differences among farm types and regions were at a similar level compared to the present analysis. A more recent study, which linked SALCA data to the SILAS model, found an average energy use in Swiss agriculture of only 36 GJ/ha (Zimmermann, 2008). Slightly lower figures were obtained by Mack *et al.* (2007). Neither Zimmermann (2008) nor Mack *et al.* (2007) distinguished between organic and non-organic farms. The difference in absolute numbers is attributable to the different perspective taken by the study. While the present study analyses energy use per ha for representative farm types, the studies cited above employ a full sector-level perspective. The main difference lies in the calculations for intermediate products, namely feedstuffs, sold within the optimised farm group. This study optimised 94 farm groups simultaneously instead of a single or a few regional farms, as done by Mack *et al.* (2007) and Zimmermann (2008). Hence, feedstuffs produced on one farm but used for the other were counted twice, *i.e.* for both the producing and purchasing farm groups. Therefore, the energy use calculated in this study should not be taken as a figure for the whole agricultural sector. One way to overcome this methodological drawback is to model factor markets, e.g. as modelled by Bertelsmeier (2005) for the land market. Multi-agent models could provide a sound answer to this problem, as implemented e.g. by Happe *et al.* (2006).

To sum up, the study results indicating that the increased implementation of organic farming led to a decreased fossil energy use by the Swiss agricultural sector can be confirmed despite the methodological shortcomings described above. Both the calculated absolute energy use levels and the relative differences between the farm types and regions are broadly in line with the results found in the existing literature.

Biodiversity

In contrast to the impact category ‘energy use’, large-scale assessments of impacts on biodiversity in terms of habitat quality are rare. The model results of this study show a higher habitat quality on organic farms for all the farm groups compared. On average, habitat quality scores on organic farms exceeded the scores on conventional farms by 55 %. The biggest differences were found on suckler cow farms (86 %) and in the lowlands (56 %).

Using the driving force-state-impact-response indicator framework, the IRENA operation (EEA, 2005) suggested several indicators for assessing the impact of agriculture on biodiversity. These include response indicators such as ‘area under organic farming’ or ‘area under other agri-environmental support’, and pressures such as ‘mineral fertiliser consumption’, ‘cropping and livestock patterns’. For instance, the Shannon index is used to describe habitat diversity (Holzschuh *et al.*, 2007; Sipiläinen *et al.*, 2008). State and impact indicators such as ‘population trend in farmland birds’ and ‘impacts on habitats and biodiversity’ are also suggested (EEA, 2005). A framework covering habitat quality comprehensively or which at least includes state and impact indicators in a sector-level approach was not found in the literature. Therefore, validation of the values obtained in this study is not possible due to the lack of comparative benchmarks.

The results on average habitat quality can be considered to be approximations of real habitat quality. A well-founded empirical base of evidence on the impacts of agricultural practices on habitat quality for different species exists from previous evaluations of Swiss direct payments (Herzog, 2005; Knop *et al.*, 2006). Therefore, although habitat quality is highly variable on each field the model assumptions are based on solid data. Moreover, the SALCA-BD model, which was used for this analysis, has been validated on various farms (Jeanneret *et al.*, 2008). Despite this, other comparisons of biodiversity between organic and conventional farming systems have come to different results. Current international meta-studies suggest that farming-system differences in scores for habitat quality on arable land have been somewhat underestimated in the SALCA-BD model (Bengtsson *et al.*, 2005; Hole *et al.*, 2005).

Farm structure data used for the model are based on FADN-datasets. The deviation between the modelled uptake levels of agri-environmental measures and the FSS datasets (BfS, 2009) is small. The results confirm the findings from previous studies including Jurt (2003) and Steiner (2006).

However, further improvements in the modelling quality could potentially be achieved by consistently linking the production inventories of the SALCA-BD tool with FARMIS assumptions. This means that each model activity in each farm group would receive a separate score for habitat quality, depending on the input use and the stocking density calculated endogenously by FARMIS. Furthermore, a re-weighting of the FADN farm weights by a maximum-entropy model could be conducted for calibrating FARMIS to the uptake levels of specific agri-environmental measures.

Moreover, as ECA elements have a substantial impact on biodiversity, the quality of the results would improve if further ECA elements, such as hedgerows and high-stem fruit trees, were explicitly included. Such an enlargement of the model would potentially lead to even more distinct differences in habitat quality between organic and conventional farms, since, as Jurt (2003) showed, implementation of these nature-conservation elements is higher on organic farms. However, modelling these nature-conservation elements at sector level using an economic model is complex because the uptake decision is influenced by the farmers' personal attitudes and by site-specific considerations, which can be modelled only indirectly using a PMP approach.

Thus, despite the above mentioned methodological constraints, the scores for habitat quality are considered to be robust. Differences between the farming systems have been somewhat underestimated.

Eutrophication

Similar to biodiversity, eutrophication is an environmental impact category which is not frequently modelled at sector level. Instead, pressure indicators such as nutrient balances are most frequently used (see Table 16). According to the model results, organic farms have 35 % lower eutrophication levels than conventional farms. The biggest differences in phosphorus eutrophication were found in the lowlands (33 %) and for mixed farms the most (27 %). Differences in nitrogen eutrophication rates were particularly notable for hill and mountain regions (22 % each) and for suckler cow farms (32 %). Phosphorus eutrophication differed in the lowlands (33 %) and for mixed farms the most (27 %).

Official evaluations undertaken by the Swiss Federal Government showed potential loss of nitrogen from the system of 74,000 t N in 2005 (surplus 86,000 t N) (Herzog and Richner, 2005). If the hectare averages are summed up for the whole sector, slightly higher values with (79,700 t N-equivalents) are obtained. With 7,600 t of potential P eutrophication the calculated value was slightly higher than 6,000 t as calculated in the official evaluations (Herzog and Richner, 2005). These marginally higher values may be attributed to the fact that eutrophication is also taken into account in preceding steps, e.g. during the production of mineral fertilisers. However, for the impact category eutrophication, the impacts of imported feedstuffs were not taken into account for the purchasing farm. This deviation from the standard LCA methodology was chosen because of the regional character of this study, with area as a functional unit. Including eutrophication related to purchased feedstuffs would have increased the difference in eutrophication between organic and conventional farms due to the higher quantities of purchased feedstuffs per ha on conventional farms.

Nonetheless, the calculated difference between organic and conventional farms is slightly higher in the present study, at 37 % on average compared to 27 % according to Nemecek and Gaillard (2004). Most international studies have also calculated higher differences between organic and conventional farming systems (Haas *et al.*, 2001; Stolze *et al.*, 2000). However, these studies are only partly comparable to Swiss conditions, because in Switzerland conventional farms have to comply with PEP requirements (Nitsch and Osterburg, 2005). Therefore, the baseline for the comparison is different from that in other countries.

The most substantial methodological drawback of the approach used in this study is the normative character of the generated eutrophication values. The production inventories for the SALCA data assume general compliance of farms with the standards. However, in reality this is often not given, as N and P eutrophication especially occur when guidelines, e.g. for fertiliser application, are not followed by the farmer. Furthermore, FARMIS-endogenous fertiliser balances suggest differences in fertilisation rates between farm types. These have not been explicitly taken into account in the FARMIS model. Therefore, the methodological considerations regarding the linkage of SALCA data to FARMIS set out above in the context of biodiversity, also apply to eutrophication.

Although the calculated total eutrophication rates are slightly higher than the eutrophication potential calculated in the official evaluations, the values are generally plausible. Relative differences between the farming systems are also within a plausible range and can be traced back to the assumptions made in the model. However, the relative differences between the farming systems may be slightly underestimated due to the normative linkage of eutrophication to FARMIS.

Public expenditure

The public expenditure calculated using the model comprised payments to farmers and public policy-related transaction costs. It should be emphasised that there are different approaches to calculating the costs of organic farming, as discussed in Section 3.3.

Total public expenditure per ha on organic farms was 3.3 kCHF/ha compared to 2.6 kCHF on conventional farms. Thus average public expenditure per ha for organic farms exceeded the public expenditure for conventional farms by 26.6 %. The difference was smallest for suckler cow farms (9.4 %) and farms in the mountain regions (9.6 %). The highest differences were evident for mixed farms (31.2 %) and farms in the lowlands (23.9 %). In absolute numbers, the difference in direct payments ranged between 305 and 738 CHF/ha. Public transaction costs accounted for less than 5 % of total costs. However, farm level policy-related transaction costs are higher on organic farms due to the costs of private certification.

The calculated differences in public expenditure can be regarded as robust, since the model validation showed that these figures deviate only slightly from official datasets. In contrast to comparisons in the EU, where at least first-pillar support is generally higher on conventional farms (Häring and Offermann, 2005), in Switzerland both general and ecological direct payments to organic farms per ha tend to exceed payments to conventional farms. Furthermore, the Swiss direct payment scheme has a high transfer efficiency, partly because of the higher absolute levels of direct payments (Buchli and Flury, 2005). As a consequence, potential differences in transaction costs are less significant in relation to the overall results. Nevertheless, relative differences in public and farm level transaction costs are in line with previous studies (Hagedorn *et al.*, 2003). It should be emphasised that the analysed transaction costs comprise only the labour costs incurred in administering and implementing the policies. Neither the cost of supplies nor market transaction costs are covered.

As in the case of structural differences and the environmental indicators, the varying distribution of farms among regions and types makes a sound comparison between the farming systems difficult. Nevertheless, in the comparison by farm type, structural differences which are not farming-system inherent can be assumed to be largely excluded, as discussed in Section 7.1.

The inclusion of differences in producer surplus would lead to different results, as the higher direct payments would count as benefits (or cost reductions). According to Jacobs (1998), the farm-level parameter farm income corresponds to net value added at sector level. The net value added can be taken as a figure for producer surplus. As the average net added value per ha on conventional farms slightly exceeds the net added value on organic farms, total cost is marginally higher than calculated using the approach applied in this study.

However, due to the varied distribution of organic farms among the regions and farm types, the comparisons by region and farm type are significant. These comparisons reveal lower costs of organic farming in a welfare-economic perspective for all farm groups and regions in contrast to the purely budgetary perspective. Only suckler cow farms show a marginally higher producer surplus on conventional farms (Table 66).

It is particularly notable that even without considering external costs, both in the lowlands and hill regions organic farming results in welfare gains of 973 CHF/ha (lowlands) and 295 CHF/ha (hills). For mixed farms a welfare gain of 149 CHF/ha was calculated. At the same time, welfare losses occur in the mountain regions (53 CHF/ha) and on dairy (173 CHF/ha) and suckler cow farms (414 CHF/ha). But it should be noted that potential changes in consumer surplus were not assessed.

Table 66 Cost of organic farming in Switzerland from a welfare-economic perspective

Indicator	Unit	Lowlands	Hill	Mountain	Dairy farms	Suckler cow farms	Mixed farms	Total farms
Change in public expenditure (-ΔST)	CHF per ha	538	489	306	447	311	738	686
Change in producer surplus (ΔPS)	CHF per ha	1,511	784	253	274	-103	887	-84
Change in consumer surplus (ΔCS)	CHF per ha	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Cost (welfare-economic perspective (ΔST+ΔPS))	CHF per ha	973	295	-53	-173	-414	149	-770

n.a. = not assessed

Source: own calculations

To sum up, organic farms receive higher direct payments per ha. These higher direct payments were conceptualised as the societal costs of organic farming. If producer surplus was included in the calculations, the regional and farm type-specific cost figures for different farm types and regions would change substantially, while the costs for total farms would be only marginally affected.

Abatement cost

The cost-effectiveness of organic farming was expressed as the abatement cost. According to the calculations in this study, the abatement cost of organic farms regarding energy use is 12.6 CHF per 1 % improvement, ha and year. Habitat quality provision cost is 12.5 CHF per ha. Eutrophication is abated by organic agriculture for 19.5 CHF/ha. Abatement cost varies strongly by region and farm type. The highest abatement cost was detected for mixed farms and for farms in the lowlands. On suckler cow farms and in the mountain regions improvements were achieved at lowest cost.

The cost-effectiveness of organic farming in relation to providing environmental services has not been calculated in a similar approach before. The existing literature concentrates on specific aspects of cost-effectiveness but does not opt for an integrated framework including uptake, environmental effects and public expenditure. The sole exception to this is Ziolkowska (2008), who analysed the cost-effectiveness of organic farms with respect to biodiversity indicators in a Polish *Voivodship* using expert assessments as part of an Analytical Hierarchy Process approach. Ziolkowska (2008) expressed the effectiveness of agri-environmental measures as the ratio of benefits and cost. Corresponding to the results of the present study, Ziolkowska (2008) obtained both high costs and high environmental benefits of organic farming compared to agri-environmental measures. Although the benefit-cost ratio for organic farming was low compared to agri-environmental measures, Ziolkowska (2008) concludes '[…] *for the benefit maximization the measures: 'Extensive meadow farming', 'Ground and water protection' and 'Organic farming' are recommendable, however, high costs have to be taken into account for the last measure.*'

Due to the relatively robust and plausible values calculated for the environmental impact categories and public expenditure, the calculated abatement cost can also be judged as relatively robust. However, the results could be substantially influenced by the type of approach used for calculating abatement cost. For example, differences in agricultural income could be included in the calculation on either the cost or benefit side. Furthermore, different environmental categories, such as eco-toxicity or different indicators for the categories, such as energy use per net added value, would affect the results for cost-effectiveness significantly.

To sum up, the calculated figures for cost-effectiveness of organic farming can be considered to be realistic estimates. As the environmental effects particularly for biodiversity and eutrophication are likely to be underestimated, the calculated cost-effectiveness of organic agriculture is regarded to be rather too low than too high. Against the background of the methodological choices, the potential impacts of different system boundaries and other assumptions for the cost-effectiveness comparison, the values for cost-effectiveness of organic farming are considered both plausible and robust.

Cost-effectiveness of agri-environmental policies

In this section, the most important results regarding a) farm structure and financial performance, b) the environmental impacts (fossil energy use, biodiversity and eutrophication), c) public expenditure, and d) abatement cost are examined.

Farm structure, financial performance, and uptake

Like organic farming, the agri-environmental measures ‘extenso payments’, ‘less intensive meadows’, and ‘extensive meadows’ have an impact on farm structure and financial parameters.

According to the model results, the general farm structure is affected to only a marginal extent by single agri-environmental measures. The strongest effect is induced by extensive meadows which lead to a decrease in stocking rate by 2.6 %. It is notable that the responses of organic farm groups to the different scenarios are more elastic compared to conventional ones. The agri-environmental payments have positive effects on farm income (0.2 to 1.6 %).

The model results suggest high windfall profits for extensive grains and rape, as an abolition of payments lead to only slight decreases in extenso area. In contrast to extenso payments, both less intensive and extensive meadows would be implemented on only 20 % of the current area. However, there are strong interactions between less intensive and extensive meadows. Less intensive meadows compete mainly with both extensive and intensive meadows. An abolition of less intensive meadows results in an increase in extensive meadows by 23.8 % and an increase in intensive meadows by 1.6 %. The abolition of extensive meadows leads to almost a doubling of less intensive meadows area.

Zgraggen (2005), modelling a reduction in general ECA payment rates, found a constant share of less intensive meadows. Among meadows, the reduction in payment rates affected only the uptake of extensive meadows (Zgraggen, 2005). It should be stated here, however, that the comparability of Zgraggen's analysis and the present study is limited as Zgraggen modelled the policy impacts for a specific geographic region in Switzerland (*Greifensee*) rather than for the entire sector.

Similar to the model results of this study, Zgraggen (2005) also found two general effects of extenso payments. First, extenso payments lead to a substitution of intensive grains and rape by extensive grains and rape. Second, extenso payments lead to an increased cultivation of grains in general compared to other arable crops and to permanent grassland. In Zgraggen's model the abolition of extenso payments led to a complete substitution of extensive grains and rape by intensive activities, which contradicts the results of this study.

As the sensitivity analysis showed, such a complete substitution is only realistic if extreme assumptions are made regarding the elasticity of the intensity levels, *i.e.* if LP-like model behaviour for policy uptake is assumed. By contrast, the results of the present study reveal strong windfall profits due to extenso payments, which seems plausible since extenso grains are often cultivated in combination with special supply contracts with a private label for integrated agriculture. Thus the abolition of extenso payments is unlikely to result in a complete substitution of extensive activities unless the termination of the integrated farming label is assumed at the same time. However, since assumptions regarding uptake elasticity have proven to be a crucially determining factor for the evaluation of extenso payments, an econometric estimation of the elasticity parameter for the Röhm-Dabbert approach seems to be advisable.

Nevertheless, the model results from the present study on farmers' responses to changes in direct payments are fully in line with economic theory. However, there is only limited literature available which analyses the uptake levels of agri-environmental payments using an appropriate, *i.e.* non-linear programming, modelling approach.

Environmental impacts and public expenditure

The structural impacts of the policies resulted in generally marginal changes in environmental performance and public expenditure. Of the analysed agri-environmental measures (extenso payments, payments for less intensive meadows and payments for extensive meadows), the best performing measure in terms of environmental effectiveness was extensive meadows.

Both extenso and less intensive meadows rendered only small or no environmental improvements and therefore had high abatements costs. The combination of these agri-environmental measures led to an improvement in the indicators (triggered predominantly by the payments for extensive meadows) of 1.5 % (energy use), 18 % (habitat quality) and 2.2 % (eutrophication) at an additional cost of 73 CHF per ha.

The choice of functional unit has a particular influence on the outcomes of the comparison between the farming systems regarding fossil energy use. If the energy efficiency of agricultural production were assessed (e.g. using the indicator 'energy input per energy output' or 'energy input per net added value'), differences between farming systems would be lower, due to both lower productivity in physical units and a lower production value on organic farms. However, as discussed in Chapter 6, in order to obtain a consistent functional unit and because the fact that direct payments, (which are also linked to area) are to be evaluated, the indicator 'energy use per ha' is preferred. Moreover, in the Swiss policy context with an emphasis on the multifunctionality of agriculture, the productive function of agriculture has become one among others. The relative importance of agricultural production decreased, while the importance of provision of multifunctional services increased. Furthermore, as organic farming is compared with other production-reducing policies, the cost-effectiveness of these policies would decrease as well if the indicator was changed. This puts the problem of the functional unit into perspective.

There is generally a lack of studies comparing the environmental effects of the selected Swiss agri-environmental policies at sector level. Zimmermann (2008) analysed the effects of both fertiliser taxes and energy taxes of 70 % and 91 %, respectively, on the energy use in the agricultural sector. These scenarios resulted in only marginal changes in total energy use (1-2 %). Zgraggen (2005) found an overall negative effectiveness of extenso payments regarding nutrient losses, with an increase of 3 % in both N and P losses, while the results of the present study show almost constant eutrophication rates, given the abolition of extenso payments.

At the same time, nitrate and phosphorus losses are 10 % and 6 % higher, respectively, when payments for ecological compensation areas are set to zero (Zgraggen, 2005). Such a scenario was not calculated in the present study. However, the combined abolition of all three agri-environmental payments analysed in this study showed reductions in total eutrophication of 2.2 % (nitrogen eutrophication 2.3 %; phosphorus eutrophication 1.2 %). Results for policy-related transaction costs have been taken directly from Buchli and Flury (2005) and Mann (2003a).

The results of the present study need to be discussed against the background of the uncertainty regarding uptake elasticity. As shown in the sensitivity analysis, uptake is an important determinant of both the sector-level impacts of the policy and public expenditure. Originally, Röhm and Dabbert (2003) developed their approach for grain intensities, while in this study the approach was adapted for intensities of meadows. However, as the choice of meadow intensity has more marked impacts on general farm management than grain intensities, the question arises whether the Röhm-Dabbert approach should be used for grassland activities at all and, if so, which elasticity factor should be employed. The above thoughts suggest opting for a lower sensitivity for grassland activities than for grains and rape. However, since no numbers are available that are supported by empirical results a medium sensitivity of 0.5 was opted for.

Thus the model showed plausible responses for the environmental effects in relation to the policy scenarios, even though the model responses are less elastic than in the *Greifensee* model (Zgraggen, 2005). Against the background of the susceptibility of the results to changes in uptake elasticity, the quantitative number of these effects should be regarded as estimations rather than exact values.

Abatement cost

As an indicator of the cost-effectiveness of the agri-environmental measures, abatement costs, were calculated on the basis of the above parameters. The cost-effectiveness of the agri-environmental policies was highly variable. Extenso payments induced only minimal environmental impacts at sector level. Hence, the cost-effectiveness of the policies was very low, with average abatement costs of 573 CHF/ha. Indeed, less intensive meadows induce negative environmental impacts due to their substitution effect with respect to extensive meadows. Abatement costs for this measure are not defined, especially because the implementation of less intensive meadows results in lower total public expenditure.

At the same time, the abatement costs of the payments for extensive meadows were low, at 9.3 CHF/ha for energy use, 0.7 CHF/ha for improving habitat quality and 6.6 CHF/ha for abating eutrophication. For the combined agri-environmental measures, the average abatement cost (unweighted mean over all environmental impacts) amounted to 10.1 CHF/ha.

Abatement costs were lower on organic farms than on conventional farms in terms of fossil energy use and eutrophication, while for habitat quality both farming systems showed similar abatement costs. Thus the policies improved the environmental categories at lower cost on organic farms than on conventional farms.

Compared to the figures on cost-effectiveness in the existing literature (Kränzlein, 2008; Schleef, 1999; Ziolkowska, 2008), the abatement costs calculated in the present study are rather high. There are several reasons for this gap. First, the LCA data on environmental impacts is somewhat underestimated when compared to the international literature. Second, as direct payment levels in Switzerland are high compared to the countries of origin of the studies cited, the public expenditure, *i.e.* the cost of the policy measures, is also high. Third, the level of cross-compliance in Switzerland is at a rather high level. Therefore, the potential for additionality of the analysed measures analysed is low. Finally, the costs were obtained by adding up the payments to the farmers plus the public transaction costs as compared to calculating the farm-level opportunity costs. While the farm-level opportunity costs do not include the cost of windfall profits, these are included in the calculations of the present study.

The lower abatement costs on organic farms compared to conventional farms suggest positive interactions between the policy measures and the organic farming system approach. Schader *et al.* (2008b) assumed that these positive interactions are caused by the greater willingness of organic farmers to take up agri-environmental measures. This greater willingness may be attributed either to a fit between the agri-environmental measures and organic farm management, for example due to stocking rates already lower than on conventional farms and/or a organic farmers' different attitude towards nature conservation; Or, this different attitude could be due to a greater generation of social capital by organic farmers through the uptake of agri-environmental measures as hypothesised by Burton *et al.* (2008) and analysed by Stotten (2008) in a case study for the Swiss lowlands. In the present study the level of positive interaction between OFASP and the combined AEM could be quantified. It showed that on organic farms the cost-effectiveness of the analysed agri-environmental measures is nearly ten times higher than on conventional farms. This results in about 10 % increase in cost-effectiveness of combined AEM for the total agricultural sector.

Comparison of organic farming with combined agri-environmental measures

The Tinbergen Rule was the starting point for the present analysis of the research question 'how cost-effective the organic farming is in providing environmental services?'.

It was found that although the Tinbergen Rule sets valid and largely useful paradigms for policy design, it is misinterpreted by both policy makers (Swiss Federal Council, 2009) and academics (Mann, 2005a; von Alvensleben, 1998). For instance, van Alvensleben (1998) put forward the argument that the Tinbergen Rule fundamentally contradicted the support for organic agriculture at the time. Van Alvensleben's thoughts sparked a debate on the economic efficiency of organic farming payments. Other agricultural economists (Stolze *et al.*, 2000) replied by arguing that savings in transaction costs might outweigh the costs of a non-targeted policy. In this study it was shown that the Tinbergen Rule does not contradict a multi-objective policy, e.g. organic farming support, as long as it is not used as a single instrument for addressing a set of different policies. As the latter is not the case, Mann's assertion that the Tinbergen Rule constitutes a primary principle for future policy reforms in Switzerland (Mann, 2005a) is flawed. A contradiction of the Tinbergen Rule and OFASP would only exist, if it were argued that the agri-environmental measures should be abolished for the benefit of OFASP and/or the ratio of cost and effectiveness of OFASP would not be competitive to a combination of agri-environmental measures. It is for this reason that the cost-effectiveness of organic farming support needs to be established in order to determine what role it should play in an efficient mix of agri-environmental policy instruments.

Thus in this study the figures for cost-effectiveness obtained for organic farming were compared to the cost-effectiveness of the combined measures. The approach of integrating both structural differences between the farming systems and the environmental indicators proved to be appropriate for addressing the research question, since the cost-effectiveness of organic farming and other agri-environmental policies is determined crucially by structural factors. However, these structural differences are often not taken into account, for instance in comparative life cycle assessments of single products.

In the FARMIS model analysis, which is based on empirical data in Switzerland, organic farming was shown to have a slightly higher average abatement cost, at 14.2 CHF/ha compared to the combination of agri-environmental measures, at 10.1 CHF/ha. However, extensive meadows showed the highest cost-effectiveness, with abatement costs at only

1.8 CHF/ha. The cost-effectiveness of less intensive meadows was not defined because the policy in fact resulted in negative environmental effects.

The sensitivity analysis revealed that the results are relatively stable, if the underlying assumptions are varied. Only variations in uptake elasticity revealed a significant influence on the ranking of the policy measures. If a more elastic uptake response was assumed, the model calculated a higher cost-effectiveness for the measures, while the assumption of more inelastic uptake led to lower cost-effectiveness. If the Röhm-Dabbert approach was not applied, *i.e.* assuming a standard elasticity of the policy measures, the combination of AEM would entail slightly higher abatement costs (15.5 CHF) than for organic farming.

Hagedorn *et al.* (2003) compared the cost-effectiveness of organic farming with a set of agri-environmental measures which was supposed to have the same environmental effects. Their study showed that organic farming does not lead to lower transaction costs, as farm-level costs are higher due to a 100 % rate of private inspections and certification rather than the usual inspection rate of 5 % for agri-environmental measures in the EU. If organic farming is conceptualised as an agri-environmental policy, comparable inspection procedures could be applied. Farmers cultivating their fields organically could be subdivided into those who a) market their products as certified organic and b) do not supply products for organic markets, as is practised in Sweden (Dabbert *et al.*, 2004). While the first group would require a private 100 % certification and inspection procedure, the latter group could be inspected using the same institutional framework as for the agri-environmental measures.

However, the empirical results of this study do not support the crucial role of transaction costs in evaluating the efficiency of organic farming, as preliminary studies do (Dabbert *et al.*, 2004; Hagedorn *et al.*, 2003; Stolze *et al.*, 2000). It highlights instead the importance of taking into account the total societal cost of the policy measures.

As shown above, the differences in abatement costs between organic farming and the combination of agri-environmental measures are small. However, the costs calculated are subject to considerable uncertainties due to several assumptions that had to be made in this study (e.g. regarding policy uptake, number and type of agri-environmental indicators). Nevertheless, the cost for both a) organic farming and b) agri-environmental measures remains within the same level of magnitude, even if underlying assumptions are varied in the context of a sensitivity analysis.

As Henrichsmeyer and Witzke (1994b, pp. 78) point out, 'balanced policy mixes', i.e. policy mixes which do not perform particularly poorly regarding specific policy goals are preferable to 'unbalanced policy mixes' since the policy indifference curves usually are convex shaped. Organic farming showed a relatively homogeneous performance in terms of environmental effectiveness for the analysed indicators and is, therefore, in this regard an adequate basis for option, particularly if the concept of strong sustainability rather than weak sustainability is favoured by the policy makers (Neumayer, 2003).

However, the system boundaries of this study do not allow of a final, definitive judgement on the cost-effectiveness of organic farming. Rather, the present study suggests that there are only small differences in cost-effectiveness between the option of organic farming support and the option of a combination of agri-environmental measures. Thus, from an economic point of view, the debate about the environmental efficiency of organic farming is of minor importance as the bulk of the societal costs of agricultural budgets is dedicated either to general direct payments or to the first-pillar measures of the CAP.

It should also be noted that, for the benefit of a sound empirical basis of the present analysis, only one among many other possible combinations of existing agri-environmental measures was compared to organic farming. A different combination of agri-environmental policies might result in more substantial differences with respect to the cost-effectiveness of organic farming. For example, environmental taxes could be implemented for addressing the problems of energy use and eutrophication, or environmental auctions could be an efficient instrument for targeting biodiversity goals, as discussed in qualitative terms in Chapter 2.

Nevertheless, this study cannot deliver generalisable results on the cost-effectiveness of organic farming with respect to environmental impacts, not least because environmental effectiveness and costs depend, of course, on specific geographic and political contexts. However, what the present study does show is that organic farming in Switzerland is able to deliver environmental services at a competitive cost compared to a combination of currently implemented agri-environmental policies.

Now, the question arises: What could be the reasons for the comparable cost-effectiveness of organic farming compared to specific agri-environmental measures, despite the fact that organic farming is a relatively inflexible package solution consisting of different measures? The answer to this question should be sought in the determinants of cost-effectiveness: a) policy uptake or – formulated more generally – differences in farm structure, b) environmental effects, and c) public expenditure or costs in general.

First, higher uptake levels of agri-environmental measures may be induced by farm-level restrictions regarding general farming intensity, particularly regarding fertiliser purchase and stocking density. This may promote the 'fit' of the agri-environmental measures with general farm management. Additionally, as discussed by Jurt (2003) and Stotten (2008), a different attitude of organic farmers towards nature conservation could be a significant factor.

Second, the system approach of organic farming, e.g. the combination of many different rules, may induce synergetic environmental effects additional to the effects of each single restriction. For instance, while both a ban on pesticides and the use of traditional, resistant fruit varieties may be inefficient as a single measure, the combination of both measures may perform well economically. Furthermore, such interconnections could exist not only between measures but also between environmental impact categories. For instance, a reduction in eutrophication is causally related to improvements in biodiversity.

Third, unlike single agri-environmental measures the systematic whole-farm approach of organic farming results in higher market values for agricultural produce. This should be considered if the welfare economic impacts of organic farming are being evaluated.

This suggests that financial support for organic farming can be as economically sound as support for agri-environmental measures. However, both the second and third aspect in particular has not yet been addressed by sufficient research projects to give precise answers based on empirical results.

Furthermore, the question regarding the means by which this support should be granted is a different one. Different combinations of measures could be more cost-effective than the current system of OFASP. In addressing this question, however, the support for organic farming becomes a policy goal in itself. According to Elliott *et al.* (2003), the main rationale for such a policy is the provision of environmental public goods and compensation for market failure. International evaluation studies of organic farming schemes (CRER, 2002) and organic action plans in Europe (Lampkin *et al.*, 2008) suggest a wide portfolio of measures, which could develop mutually synergetic effects with current support schemes both in Switzerland and elsewhere.

9 Conclusions

This chapter contains the main conclusions concerning the cost-effectiveness of organic farming in providing environmental services based on the findings of the previous chapters. Section 9.1 presents the contribution of this thesis to knowledge in the order of the research objectives and working hypotheses formulated in Chapters 1 and 5, respectively. Following this, the implications for a) agri-environmental policy (Section 9.2) and b) research on agri-environmental policy (Section 9.3) are formulated.

9.1 Contribution to knowledge

The present thesis contributes substantially to knowledge regarding the principal research aim: to compare the cost-effectiveness of organic farming with the cost-effectiveness of individual agri-environmental policies by developing and applying an economic modelling framework at sector level for the Swiss case.

The main research aim was broken down into the following research objectives:

1. To review current knowledge about economic evaluation and environmental impacts of organic farming at an international level as a basis for the development of an analytical framework and research hypotheses.

2. To design an analytical framework and economic model for analysing the cost-effectiveness of organic farming and other agri-environmental policy measures for the Swiss agricultural sector.

3. To assess the relative environmental impacts of organic farming with respect to fossil energy use, biodiversity and eutrophication with nitrogen and phosphorus.

4. To compare the cost-effectiveness of organic farming with the cost-effectiveness of agri-environmental measures.

The following section describes the contribution of this thesis to knowledge according to these research objectives.

Contribution to knowledge related to Objective 1

In an extensive literature review on agri-environmental policy evaluation and on the environmental impacts and costs of organic farming, knowledge from different scientific disciplines was woven together to form a comprehensive overview of the state of the art. Based on this literature review, a number of assumptions were formulated for a theoretical model in order to analyse the implications of the Tinbergen Rule for multi-objective policies. The model consisted of three policy objectives and four policy measures affecting the policy targets and giving rise to public expenditure. Using the model, it was possible to prove on a theoretical level that the Tinbergen Rule as a sole argument is not a sufficient reason to exclude organic agriculture support policies from a portfolio of agri-environmental policy instruments. The results of this model represent a contribution to knowledge additional to the state of the art and build the theoretical justification of the subsequent empirical model analysis.

Contribution to knowledge related to Objective 2

This thesis contributes to knowledge in methodological terms as it involved designing a conceptual model for assessing the cost-effectiveness of agri-environmental policies. Apart from public expenditure and environmental effectiveness, policy uptake and farm structure were identified as a major determinant of cost-effectiveness. In order to make the conceptual model workable on the basis of empirical data, the sector-representative farm-group model CH-FARMIS was used as a basis and expanded with three modules addressing the main determinants of cost-effectiveness. First, life cycle assessment data for energy use, biodiversity and eutrophication involving nitrogen and phosphorus were linked to CH-FARMIS in order to obtain environmental data for sector-representative farm groups. Second, the calculation of public expenditure was completed by including policy-related transaction costs in the model. Finally, an adaptation of the Röhm-Dabbert approach for modelling the uptake of agri-environmental policies using known activity levels in the base year was implemented for CH-FARMIS. The model thus developed is the first comprehensive model that both covers the main determinants of cost-effectiveness and enables analysis of the cost-effectiveness of organic farming.

Contribution to knowledge related to Objective 3

The contribution to knowledge related to Objective 3 is presented according to the working hypothesis formulated in Chapter 5. The first working hypothesis stipulated that:

> *H1 Generally, organic farms perform better with respect to the environmental impact categories energy use, habitat quality and eutrophication than conventional farms.*

The model analysis confirms Hypothesis 1. Organic farms have a better environmental performance per ha than conventional farms in all regions and for all farm types. Organic farms have an average energy use of 20.2 GJ/ha compared to conventional farms with 44.3 GJ/ha. Hence the relative improvement on organic farms as compared to conventional farms (disregarding issues concerning self-selection bias) is 54 %. These differences are predominantly attributable to the lower amounts of purchased fodder and lower animal husbandry-related energy use due to lower stocking rates. Furthermore, the ban on mineral fertilisers is an important factor for the lower energy use on organic farms.

Average habitat quality on organic farms is 25.7 % of a theoretical maximum value, while conventional farms achieve an average habitat quality of only 16.6 %. This equates to a 55 % better performance of organic farms. The major contributor to the higher average habitat quality is the larger share of grassland in total UAA and the larger share of extensive grassland in total grassland on organic farms compared to conventional farms.

Average eutrophication amounts to 59 kg N-eq/ha on organic farms, while conventional farms emit 91 kg N-eq/ha. Hence, organic farms have a 35 % better performance regarding this indicator. The lower eutrophication per ha is driven mainly by lower nitrate emissions due to smaller shares of arable land in total UAA and decreased ammonia emissions due to lower stocking rates.

> *H2 The relative differences in environmental impacts between conventional and organic suckler cow and dairy farms are smaller than on mixed farms, due to the higher proportion of grassland on these farm types and the smaller difference in environmental impacts on grassland between conventional and organic systems.*

Hypothesis 2 can be confirmed only for energy use, while it has to be rejected for habitat quality and eutrophication. Energy use per ha on organic mixed farms is about 50 % lower than on conventional mixed farms, whereas differences for suckler cow and dairy farms are only 30-36 %. Differences between farming systems regarding habitat quality are highest for suckler cow farms (86 %) and lowest on mixed farms (34 %), while dairy farms show medium differences between the farming systems, at 53 %. Differences in eutrophication are highest also on suckler cow farms (32 %) and lowest on mixed farms (13 %), whereas differences on dairy farms between both farming systems are in the medium range (25 %).

The higher differences in energy use per ha on mixed farms can be attributed to the greater differences in purchased fodder, stocking density and mineral fertiliser use between organic and conventional mixed farms than between both farming systems for the farm types suckler cow and dairy farms. Thus the share of arable land is a determining factor for the farm-type differences in energy use, but stocking density is much more influential for this environmental impact category.

The major determinant for the relative differences between organic and conventional farm types in habitat quality is the uptake level of ecological compensation measures. According to the model results, differences on arable land are relatively small, while differences on grassland are much more influential. In particular, the large differences in uptake of less intensive and extensive meadows (Table 33; page 174) gave rise to the great difference in habitat quality on organic suckler cow farms compared to conventional suckler cow farms. The ECA-uptake levels are also much higher on organic dairy farms than on conventional ones. By contrast, mixed farms of both farming systems have relatively similar uptake levels of ECA measures.

In contradiction to Hypothesis 2, differences in eutrophication between farming systems on mixed farms are the smallest of all the farm types analysed. Both the difference in nitrate- and ammonia eutrophication is smaller on mixed farms than on suckler cow and dairy farms. Only in relation to differences in phosphorus eutrophication was Hypothesis 2 confirmed. Cropping patterns are the most important driver for relative differences between organic and conventional farm types in eutrophication. Mixed farms have high shares of arable land, which on conventional farms is assumed to be fertilised primarily by mineral fertilisers, unlike grassland, which is fertilised predominantly by organic fertilisers. In contrast to conventional farms, organic farms use only organic fertilisers for both arable crops and grassland. Organic manure is the major cause for both ammonia emissions and nitrate leaching. These high emissions diminish or even outweigh (in case of ammonia emissions on mixed farms) the differences between the farming systems because of the lower stocking rate. By contrast, phosphorus emissions, being a function of the susceptibility of land to erosion, are more common on arable land, as the soil lies open during significant periods of time in the year. Due to the higher share of arable land on conventional mixed farms than on organic mixed farms, phosphorus erosion is higher on conventional mixed farms, while this difference is not as marked as on the other farm types.

H3 Relative differences in environmental performance between the farming systems are smaller in the mountain regions than in other regions.

Hypothesis 3, like Hypothesis 2, has to be rejected for habitat quality and eutrophication but is confirmed for energy use. Organic farms in the lowlands showed the highest reduction in energy use per ha (55 %) compared to their conventional counterparts. Differences in the hill regions are at a medium level (41 %) and in the mountain region these difference are smallest (33 %). In contrast to this, the effects of organic farming on habitat quality are highest in the mountain regions (61 %). In hill areas and in the lowlands, these effects are much smaller than in the mountain regions, at 29 % and 23 %. Similar to the results on habitat quality, the greatest differences between the farming systems in terms of eutrophication are calculated for hill and mountain regions (both 22 %), whereas the difference in lowlands is only 11 %.

It can be confirmed that the differences in energy use per ha between organic and conventional farming systems is smaller in mountain regions than in the lowlands and in hill regions. Energy use per ha is driven mostly by purchased concentrate fodder for animals. The fodder demand depends on the stocking density and type of animals. While ruminants have relatively low shares of concentrate fodder in their rations, pigs and poultry are fed on high shares of concentrate fodder. Consequently, as the shares of pig and poultry livestock are high on conventional lowland farms, energy use per ha is most different in the lowlands.

The greater differences in ECA uptake levels between organic farms and conventional farms in the mountain regions determine the differences in average habitat quality. The difference in uptake of extensive meadows between the farming systems is particularly significant.

The degree of eutrophication depends predominantly on stocking-rate differences and the share of arable land. As stocking-rate differences between the farming systems are almost equal in all regions, the difference in arable land compensates these differences to a larger extent, as the organic arable land is fertilised by organic fertiliser only. However, the way fertilisation was implemented in the model allows only a limited response to this hypothesis and Hypothesis 2, since the model builds on representative SALCA data which are differentiated by farming system and region but not by farm type. Hence, the data do not take account of farm-group variations of environmental impacts resulting from the model-endogenous fertiliser amounts, as discussed in Chapter 8.

Summarising the contribution to knowledge related to Objective 3, it can be said that the model analysis in this thesis was able for the first time to generate sector-representative figures on the environmental effects of organic farming on energy use, biodiversity and eutrophication involving nitrogen and phosphorus. Generally speaking, it was calculated that per ha fossil energy use is 54 % lower, habitat quality is 55 % higher, and eutrophication is 35 % lower on an average organic farm compared to an average conventional farm. Furthermore, this thesis revealed structural differences including, in particular, stocking density and agri-environmental policy uptake, as important driving factors for the environmental effects.

Contribution to knowledge related to Objective 4

The fourth working hypothesis anticipated that:

H4 Organic farming provides individual environmental services (reduction in energy use, improvement in habitat quality, reduction in eutrophication potential) at a higher cost than specialised agri-environmental measures.

This hypothesis is confirmed by the modelling analysis, as the environmental impact category habitat quality, on which the agri-environmental measures primarily focus, is improved by the combined AEM at lower cost than by organic farming. A 1 % habitat quality improvement is provided by organic farming at 12.5 CHF/ha, while the same environmental improvement is delivered via combined AEM at a cost of only 4.1 CHF/ha. Of the individual AEM, extensive meadows are the most efficient at 0.7 CHF/ha, while less intensive meadows as a single measure resulted in negative impacts on habitat quality. According to the model, extenso payments lead to very small improvements in habitat quality, which entail hypothetical provision costs of about 573 CHF/ha.

H5 Considering multiple environmental effects and public transaction costs of policies, the abatement costs of organic farming are comparable with or lower than other existing agri-environmental measures.

Hypothesis 5 is confirmed for the concrete comparison with selected Swiss agri-environmental policies. Considering the three environmental impact categories analysed in this thesis, the same level of average environmental services is delivered at comparable cost by both organic farming (14.2 CHF/ha) and the combination of agri-environmental payments (10.1 CHF/ha). However, it should be noted that the exact figures of cost-effectiveness at sector level are susceptible to the main assumptions of the modelling approach as discussed in Section 7.5 and Chapter 8. Most prominently, the selected environmental indicators and the assumptions regarding the ease of policy uptake influence the results. But, as shown in Chapter 2 by employing a theoretical model, this result does not contradict the Tinbergen Rule as such because it does not rule out multi-objective policies outright. Depending on how many policy goals are taken into account and depending on the cost-effectiveness of the measures, efficient agri-environmental policy may consist of either single agri-environmental measures only, targeted at one goal each, or a combination of single targeted agri-environmental measures complemented by multi-objective policies.

Therefore, against the background of both the results of the theoretical considerations and the empirical model analysis at sector level, organic farming can be regarded as competitive with current agri-environmental policies in delivering environmental services analysed in this study. However, it is important to emphasise that this result is valid only for the policy measures and environmental impact categories analysed.

H6 There are synergy effects between the system approach of organic farming and individual agri-environmental policy measures which result in a higher cost-effectiveness of the agri-environmental measures when applied on organic farms than when applied on conventional farms.

This hypothesis is in full confirmed by the model analysis. Clear synergy effects of the measures are identified, as they have stronger impacts on energy use and eutrophication on organic farms than on conventional farms. Moreover, additional public expenditure for the policies is lower on organic farms than on conventional farms. Consequently, the cost-effectiveness of the combined measures regarding a reduction in energy use and eutrophication is five to six times higher on organic farms than on conventional farms.

Effects on habitat quality are at the same level on both organic and conventional farms. However, since public expenditure for the combined AEM is lower on organic farms than on conventional farms, a three to four times higher cost-effectiveness with respect to habitat quality was calculated for organic farms.

Synergies between OFASP and the combined AEM could be quantified to a 10 % increase in average cost-effectiveness. Hence although the discussed methodological limitations bound the significance of the results, this thesis showed that cost-effectiveness of organic farming in Switzerland for providing the environmental services (reduction of fossil energy use, improvement of habitat quality, and reduction of eutrophication with nitrogen and phosphorus) is comparable with a combination of existing agri-environmental measures. This is the first representative study to analyse the cost-effectiveness of organic farming taking account of three environmental impact categories and public expenditure including transaction costs.

9.2 Implications for agri-environmental policy

The often-politicised question of whether or not organic agriculture can be an efficient means to address environmental problems provided the motivation for this thesis. Its results are therefore closely linked and highly relevant to current agri-environmental policy both in Switzerland and in the EU.

The budget-allocation model developed in Section 2.2.3 showed that the Tinbergen Rule is not applicable as a general argument for rejecting multi-objective policies, even if it is assumed that there are no interdependencies between different goals and different measures and even if it is assumed that transaction costs are zero. Tinbergen's conclusion regarding the inefficiency of multi-objective policies was confirmed, given the number of policy instruments is lower than the number of policy objectives. However, the inclusion of multi-objective policies within a policy mix that includes other policy measures can improve the efficiency of the policy mix as a whole. Thus organic farming can improve efficiency even if regarding each policy goal a respective individual targeted agri-environmental policy is more cost-effective. The optimal budgetary extent of the multi-objective policy within the policy mix depends on a) the specific context and is determined by the cost-effectiveness of the agri-environmental policies involved, b) the number and type of policy goals, and c) the initial environmental state in relation to the policy goal.

Consequently, although the Tinbergen Rule is generally a useful concept for policy design, the current reforms of Swiss Agricultural Policy (Swiss Federal Council, 2009) are based on a misinterpretation of the Tinbergen Rule, as the reform favours single-objective policies instead of multi-objective polices. Support for multi-objective policies such as organic farming does not, in principle, contradict the overall efficiency of a policy mix. Apart from the situation in Switzerland, the same debate takes place in many EU Member States and regions in the context of designing the agri-environmental measures contained in the Rural Development Plans. Thus there is no contradiction in principle to economic efficiency if organic farming is supported as one measure among other agri-environmental measures.

At the same time, the obligatory inclusion of support for organic farming in Rural Development Plans is not economically sound, at least if the rationale behind this inclusion is based on the provision of environmental services. As the high variability of the cost-effectiveness of agri-environmental measures and organic farming in different Swiss regions suggests, an economic analysis for identifying the optimal policy mix and the optimal payment rates would be beneficial.

If policy makers were to a) consider only the policy goals discussed in this study and b) choose only cost-effectiveness as a decision-making criterion for selecting the optimal mix of agri-environmental policies, then payments for extensive meadows should be expanded, while both extenso payments and payments for less intensive meadows should be abolished. However, it should be emphasised that the results of this thesis are not universally applicable, since the specific choice of impact categories differs from practical policy making, in which eco-toxicity, animal welfare and other environmental categories are also taken into account. Furthermore, extenso payments are focussed on arable land, which was not evaluated by a particular indicator. Moreover, farm incomes and productivity are central policy concerns. Therefore, definite policy recommendations regarding the choice of policy instruments would need more research focussing on the specific policy goals to be addressed.

Nevertheless, organic farming support proved to be a feasible complementary instrument for single agri-environmental policies due to its stable effectiveness across all regions and farm types and its comprehensive effect on many impact categories. The mutually enhancing interactions with the policy measure ‘payments for extensive meadows’ would be a further reason for a greater support for organic farming, as an expansion of organic farming would most likely also induce an increased extensification of meadows without higher payment levels per ha.

However, rising payments spent either on organic farming or individual policies will lead to windfall profits for an increasing proportion of farms. According to the economic literature, this will impair the environmental efficiency of the policy while improving transfer efficiency. Thus agri-environmental policies would gradually lose their character as voluntary policies, as welfare gains will be caused for farmers. Payment levels which are substantially higher than farm-level costs lose their character as compensation policies. Instead, they gradually acquire the character of a cross-compliance policy measure. The fact that all general direct payments are linked to cross-compliance, demonstrates that windfall profits are tacitly accepted as an outcome of this policy measure.

Changing the determination of payment levels of agri-environmental policies from a cost-oriented to a result-oriented rationale is theoretically more efficient. This approach would encourage farms which are not competitive on commodity markets to provide non-commodities instead. With the ‘Ordinance on Regional Promotion of Quality and Networking of Ecological Compensation Areas in Agriculture’ such a result-oriented approach was already introduced in 2001. An alternative area-wide measurement of actual environmental benefits of the policies would entail extraordinary policy-related transaction costs, which may outweigh the above mentioned efficiency gains.

Despite the strong focus of this thesis on Swiss agri-environmental policy, some implications can also be identified in relation to the CAP, as the European Commission, Member States and Regions are confronted with similar problems when designing, implementing and evaluating Rural Development Plans (especially second axis measures). The regionalisation of Rural Development Plans, with both specific portfolios of agri-environmental measures and payment levels, reflects the recognition that the problems tackled by these policies are regionally specific (Ahrens *et al.*, 2000; OECD, 2007d; Rudloff, 2002; Salhofer *et al.*, 2006). As shown by many studies, the regionalisation of payment levels improves the environmental efficiency of policy measures, as overcompensation is avoided. However, other authors stress that regional differences in payment levels are rather a function of political priorities than of accurate economic calculations (Stolze and Lampkin, 2009). Thus, transferring the approach developed here to EU Regions and Member States in order to analyse the cost-effectiveness of policy measures could be a useful means of improving the targeting of the CAP, which is a main objective of the CAP Health Check and the preceding Mid-Term Review. However, this would require a better monitoring of the uptake of agri-environmental measures within farm structure and FADN datasets. Dwyer *et al.* (2008) notice as one reason for the impossibility of a *'meaningful assessment of 2000-06 programmes and 2007-13 plans against quantified 'benchmarks''* that *'The paucity of extant data which links actual policy expenditures to clear and consistent outputs, results and impacts, in respect of even the most commonly-used instruments within the RD toolkit, across different countries and situations, is a major obstacle.'* (Dwyer *et al.*, 2008). Furthermore, a specific analysis of organic farming requires a minimum number of organic farms in FADN strata. The approach taken in this study would need to be adapted before it can be applied to other countries and before its results can be transferred to other circumstances.

Moreover, this study cannot provide the basis for advice regarding the choice of policy instruments for supporting organic farming, since the organic farming area support payments were not specifically evaluated: Instead, the total surplus direct payments to organic farms compared to conventional farms were considered. Other policy measures, e.g. conversion payments, investment support, compensation for inspection costs, input taxes on mineral fertilisers or pesticides, or tax exemptions for organic farms could be alternative, better-performing options. Furthermore, the results of this thesis suggest a regional or farm type-specific differentiation of payment levels, since both the costs and environmental effects of organic farming differed between the groups analysed. The approach developed could be used to analyse the potential for such a differentiation. Moreover, the strong interdependencies and mutual synergies shown to exist between organic farming on the one hand and single agri-environmental payments on the other indicate a huge potential for improving the cost-benefit ratio of agri-environmental policy.

When discussing policy implications it is crucial to emphasise the relative share of the policies analysed as a proportion of total public expenditure. The analysis showed that only very small amounts of the total budgets are changed. The significance of the cost-effectiveness of these policies may justifiably be questioned if they constitute a marginal share of the budget while general direct payments give rise to the greatest inefficiencies. This consideration is particularly important because the most obviously efficient way to improve the cost-effectiveness of the whole system would be to reduce the payment levels of those general direct payments which fundamentally contradict environmental policy goals. Most prominently, the direct payments linked to livestock units lead to negative environmental side-effects in all the impact categories analysed. A reduction in these payments would give rise to both environmental improvements and budget relief. If in turn the available budget was used for agri-environmental policies, large gains in efficiency would be the most likely consequence.

9.3 Implications for research on agri-environmental policy

Although this thesis contributed significantly to the knowledge on cost-effectiveness of organic farming in providing agri-environmental services, many open questions remain in this field of study. These research gaps simultaneously demonstrate starting points of possible new research projects, which are described in this section.

The results of this study imply that research on agri-environmental policies should increasingly employ a sector-level perspective in future. This will enable the analysis of the main determinants of cost-effectiveness, *i.e.* policy uptake, environmental effects and public expenditure, in an integrated way. At the same time, this approach suggests the importance of linking socio-economic and ecological research methods. Such integrated approaches are neither meant to substitute field work on environmental effects nor to replace analysis focussed on economic parameters. Rather, integrated approaches may supplement such studies and link complex results from both scientific fields in order to support decision making by policy makers by evaluating the performance of agri-environmental policies.

As the results of the sensitivity analysis have shown, further research on econometric estimation of uptake elasticity would be useful for reducing the uncertainty of the results. In the longer term, a full econometric estimation of the so called Q-matrix for all activities would be desirable. Moreover, if this approach was employed for *ex-ante* impact assessments of further policy reforms, its robustness would be improved further by taking into consideration structural change and farm-farm interactions on factor markets.

Organic agriculture was conceptualised here in a reductionist perception of agri-environmental policy evaluation. Yet, the research was focussed on only three of the multiple environmental effects that are ascribed to organic farming. These limitations were necessary in order to be able to address the research question within the limited time frame that was given for this PhD. However, for a comprehensive evaluation of organic farming, further environmental impacts, such as mitigation of greenhouse gases, eco-toxicity and animal welfare, need to be included. Given the current high policy significance of climate change, it would be particularly important and beneficial to include greenhouse gases in the model. With fossil energy use and N_2O emissions being already included, the work of this PhD builds a good foundation for including also CH_4 emissions.

Additionally, wider socio-economic impacts, for instance on food safety and agricultural markets, should be addressed, as organic farming also significantly affects commodity markets. This includes effects on other industry and service sectors resulting for example from a varying factor demand of the farms or from the wider impacts of organic foodstuffs, such as potential public health impacts. Therefore, further research should also address the linkages between organic agriculture and the rest of the economy. For instance, the argument that organic farming has lower transaction costs does not refer solely to policy-related transaction costs as covered in this thesis. Most notable are the reduced costs of labelling, including costs incurred by both retailers and consumers in searching for information, compared to a situation where many different environmental standards are employed. This implies a shift from supply-based models to integrated models covering both supply and demand for environmental services and other public goods.

Moreover, this study has demonstrated the impact of feedstuffs on energy use, and therefore also for resource depletion and climate change. Further research should address feedstuffs specifically and analyse the potential for improvements in environmental performance by changing livestock diets.

The extensions to the model that were established in this thesis make it possible to address many more questions with regard to agri-environmental policy in Switzerland. *Ex-ante* assessments of planned policy reforms would be an obvious application. The extensions made in this thesis widen the scope of possible questions which can be addressed using FARMIS. In particular, an impact assessment of the environmental effectiveness and cost-effectiveness of the policy reforms currently being planned can be carried out now. The ability to model specific farm types and farming systems is particularly helpful for the future generation of knowledge about organic farming. The modelling approach can be used, for instance, to identify the economic potential of conversion of different farm types and regions via an analysis of shadow prices.

In addition, the Röhm-Dabbert approach is a very useful tool for analysing the uptake of agri-environmental policies if different payment levels are given. Such an analysis could help to improve the targeting of agri-environmental policies, *i.e.* to identify the most efficient payment levels, and to estimate overcompensation effects and their impacts on environmental efficiency. In this regard, the economic and environmental impacts of a regional and farm group-specific variation of payment levels could be analysed.

As the Swiss FARMIS model is part of a European network of model versions in different Member States, the extensions added to it – although very Swiss specific in some respects – could feasibly be transferred to other Member States.

It is also important to reconsider the functional unit and system boundaries in LCA studies in different contexts. Commonly, the functional unit is production-related and gives rise to the positive performance of highly productive systems. At the same time, rebound effects as well as Jevon's paradox suggest that gains in environmental efficiency often lead to higher overall resource use. Such effects may provide the starting point for methodologically different solutions emerging from sector-level evaluations as opposed to evaluations at plot level, as shown here for payments for less intensive meadows.

In addition to drawing on pure neo-classical production economics, innovative approaches should be employed using ideas from ecological economics or other scientific disciplines. Such approaches could be used to analyse potential reasons for the competitive cost-effectiveness of organic farming, for instance economies of scope and linkages between measures and targets. This could also enable researchers to find improved ways to address environmental problems effectively and to identify trade-offs between effectiveness and efficiency. Furthermore, these approaches may help to define more precisely the role organic agriculture could play for potential policy solutions. This gives rise to the research question of where exactly the optimal level of support for organic farming in specific situations is, and on which factors this level of support depends. Moreover, the question arises as to whether organic farming support could be organised more efficiently than is the case with area-related direct payments, and how this might be achieved. Little empirical evidence on this topic exists to date. This point leads back, finally, to the question of conversion to organic agriculture, since the efficiency of different organic farming support policies depends crucially on the willingness of farmers to convert. As has been made clear above, farmers' decision to convert is influenced by manifold factors. The research agenda thus needs to be widened to include interdisciplinary projects, incorporating economic sociological, and psychological research approaches.

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Appendices

Annex A Details on Swiss agricultural policy

Table 67 Overview of general direct payment levels 1999-2007

Ordi-nance	Art-icle	Policy measure	Unit	2009	2008	2007	2006	2005	2004	2003	2002	2001	2000	1999
DZV	27	**General area payments**	ha											
DZV	27	General payment	ha	1,040	1,080	1,150	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,200
DZV	27	Additional payment for arable land and permanent crops	ha	620	450	450	400	400	400	400	400	400	400	0
DZV	36	**Hillside payments**	ha											
DZV	36	Gradient 18-35%	ha	370	370	370	370	370	370	370	370	370	370	370
DZV	36	Gradient >35%	ha	510	510	510	510	510	510	510	510	510	510	510
DZV	38	**Vineyard payment**	ha											
DZV	38	Gradient 30-50%	ha	1,500	1,500	1,500	1,500	1,500	1,500	1,500	1,500	1,500	1,500	1,500
DZV	38	Gradient 50%	ha	3,000	3,000	3,000	3,000	3,000	3,000	3,000	3,000	3,000	3,000	3,000
DZV	38	Terraces	ha	5,000	5,000	5,000	5,000	5,000	5,000	5,000	5,000	5,000	5,000	5,000
DZV	34	**Payment for animal husbandry under adverse conditions**	LU											
DZV	34	Valley region	LU	0	0	0	0	0	0	0	0	0	0	0
DZV	34	Hill region	LU	300	260	260	260	260	260	260	260	260	260	260
DZV	34	Mountain region 1	LU	480	440	440	440	440	440	440	440	440	440	440
DZV	34	Mountain region 2	LU	730	690	690	690	690	690	690	690	690	690	690
DZV	34	Mountain region 3	LU	970	930	930	930	930	930	930	930	930	930	930
DZV	34	Mountain region 4	LU	1,230	1,190	1,190	1,190	1,190	1,190	1,190	1,190	1,190	1,190	1,190
DZV	32	**Payment for grazing livestock**	LU											
DZV	32	Cattle, horses, goats, sheep	LU	690	860	900	900	900	900	900	900	900	900	900
DZV	32	Dairy cows	LU	450	200	200	0	0	0	0	0	0	0	0
DZV	32	Other grazing livestock	LU	520	400	400	400	400	400	400	400	400	400	400

DZV: Ordinance on Direct Payments for Agriculture (EVD, 2008)

Source: Payment levels of direct payments according to (EVD, 2008)

Table 68 Overview of ecological and ethological direct payments 1999-2007

Ordi-nance	Arti-cle	Policy measure	Unit	2009	2008	2007	2006	2005	2004	2003	2002	2001	2000	1999
DZV	58	**Organic farming**	ha											
DZV	58	Special crops	ha	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,000
DZV	58	Arable land	ha	800	800	800	800	800	800	800	800	800	800	600
DZV	58	Other UAA	ha	200	200	200	200	200	200	200	200	200	200	100
DZV	56	Extensive grain and rape production	ha	400	400	400	400	400	400	400	400	400	400	400
DZV	53	**Fallow land**	ha											
DZV	53	Rotational fallow	ha	2,800	3,000	3,000	3,000	3,000	3,000	3,000	3,000	3,000	3,000	3,000
DZV	53	Mixed fallow	ha	2,300	2,500	2,500	2,500	2,500	2,500	2,500	2,500	2,500	2,500	2,500
DZV	49	**Extensively-used meadows, Hedges, 'extensive meadows on wet sites'**	ha											
DZV	49	Valley region	ha	1,500	1,500	1,500	1,500	1,500	1,500	1,500	1,500	1,500	1,500	1,500
DZV	49	Hill region	ha	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,200	1,200
DZV	49	Mountain region 1 and 2	ha	700	700	700	700	700	700	700	700	700	700	700
DZV	49	Mountain region 3 and 4	ha	450	450	450	450	450	450	450	450	450	450	450
DZV	49	**Less intensively used meadows**	ha											
DZV	49	Valley and hill region	ha	300	650	650	650	650	650	650	650	650	650	650
DZV	49	Mountain region 1 and 2	ha	300	300	300	300	300	300	300	300	300	300	300
DZV	49	Mountain region 3 and 4	ha	300	300	300	300	300	300	300	300	300	300	300
DZV	49	High-stem fruit trees	Tree	15	15	15	15	15	15	15	15	15	15	15
DZV	62	**Payments for particular animal friendly housing (BTS)**	LU											
DZV	62	Cattle, Goats	LU	90	90	90	90	90	90	90	90	90	70	70
DZV	62	Pigs	LU	155	155	155	155	155	155	155	155	155	135	135
DZV	62	Poultry	LU	280	180	180	180	180	180	180	180	180	180	180
DZV	62	**Payments for regular access to outoor runs (RAUS)**	LU											
DZV	62	Cattle	LU	180	180	180	180	180	180	180	180	180	135	135
DZV	62	Other grazing livestock	LU	360	155	155	155	155	155	155	155	155	135	135
DZV	62	Pigs	LU	155	155	155	155	155	155	155	155	155	135	135
DZV	62	Poultry	LU	280	180	180	180	180	180	180	180	180	180	180

DZV: Ordinance on Direct Payments for Agriculture (EVD, 2008)

Source: Payment levels of direct payments according to (EVD, 2008)

Table 69 Overview of specific area payments and payments for alpine summer grazing 1999-2007

Ordinance	Article	Policy measure	Unit	2009	2008	2007	2006	2005	2004	2003	2002	2001	2000	1999
SBV	4	**Summer grazing**	ha											
SBV	4	Dairy cows, sheep and goats producing milk	ha	120	120	120	120	120	120	120	120	120	120	120
SBV	4	Other sheep	ha	300	300	300	300	300	300	300	300	300	300	300
SBV	4	Other grazing livestock	ha	260	260	260	260	260	260	260	260	260	260	260
ABBV	1	**Specific area payments**	ha											
ABBV	1	Oilseeds	ha	1,000	1,500	1,500	1,500	1,500	1,500	1,500	1,500	yield dependent	1,500	1,500
ABBV	1	Legumes	ha	1,000	1,500	1,500	1,500	1,500	1,500	1,500	1,260	1,260	1,260	1,260
ABBV	1	Sugar beets	ha	1,900	850	0	0	0	0	0	0	0	0	0
ABBV	1	Fibre plants other than hemp and flax	ha	2,000	2,000	2,000	2,000	2,000	2,000	0	0	0	2,000	2,000
ABBV	1	Oats, barley, triticale, emmer, einkorn	ha	0	0	0	0	0	0	0	0	400	770	770

SBV: Ordinance on alpine summer grazing (*Sömmerungsbeitragsverordnung, SöBV, SR 910.133*)
ABBV: Ordinance on contributions to the production of arable crops (*Ackerbaubeitragsverordnung, ABBV; 910.17*)

Source: Payment levels of direct payments according to (EVD, 2008)

Table 70 Overview of ecological compensation area measures

ECA-B	ECA-A
Mixed fallows (*Buntbrache*)	Extensive pastures (*Extensiv genutzte Weiden*)
Rotational fallows (*Rotationsbrache*)	Wood pasture (*Waldweiden (ohne bewaldete Fläche)*)
Extensive meadows (*Extensiv genutzte Wiesen (ohne Weiden)*)	Hedges without herbal strip (*Hecken-, Feld – und Ufergehölze (ohne Krautsaum)*)
Less intensive meadows (*Wenig intensiv genutzte Wiesen (ohne Weiden)*)	Water trenches and ponds (*Wassergräben, Tümpel, Teiche*)
Extensive meadows on wet sites (*Streue-flächen*)	Stone heeps and walls (*Ruderalflächen, Steinhau-fen und –wälle*)
Hedges with herbal strip (*Hecken-, Feld- und Ufergehölze (mit Krautsaum)*)	Dry masonry walls (*Trockenmauern*)
Buffer strip (*Ackerschonstreifen*)	Natural paths (*Unbefestigte, natürliche Wege*)
	Further ECA measures (*Weitere Ökoausgleichsflä-chen*)
	Vinyards with high species diversity (*Rebflächen mit hoher Artenvielfalt*)
	High-stem fruit trees (*Hochstamm- Feldobstbäume*)
	Native trees and alleyways (*Einheimische standort-gerechte Einzelbäume und Alleen*)

Source: Ordinance on Direct Payments for Agriculture (EVD, 2008)

Table 71 Swiss farm typology grid

No	Farm type	LU / UAA	OAA / UAA	Speciality crops	Cattle (LU)	Dairy cows (LU)	Suckler cows (LU)	Horses, sheep and goats (LU)	Pigs and poultry (LU)	Other rules
11	Arable crops	max. 1 ha	over 70 %	max 10 %						
12	Speciality crop	max. 1 ha		over 10 %						
21	Dairy cow		max. 25 %	max 10 %	over 75 %	over 25 %	max. 25 %			
22	Suckler cow		max. 25 %	max 10 %	over 75 %	max. 25 %	over 25 %			
23	Other cattle		max. 25 %	max 10 %	over 75 %					not 21 or 22
31	Horses/Sheep/Goats		max. 25 %	max 10 %				over 50 %		
41	Pig and poultry farms		max. 25 %	max 10 %					over 50 %	
51	Combined dairy /arable crops		over 40 %		over 75 %	over 25 %	max. 25 %			not 11-41
52	Combinded suckler cows				over 75 %	max. 25 %	over 25 %			not 11-41
53	Combinded pigs/poultry								over 25 %	not 11-41
54	Combined others									not 11-53

LU: Livestock units
UAA: Utilised agricultural area
OAA: Open arable land

Source: FAT (2000)

Annex B Detailed assumptions for the modelling analysis

Table 72 Maximum and minimum habitat quality scores

Indicator species group	Minimum score	Maximum score
All species	4.20	18.35
Amphibians (red list)	1.33	10.20
Grasshoppers (red list)	2.68	37.36
Beetles (red list)	3.41	28.00
Butterflies (red list)	2.28	41.43
Spiders (red list)	4.21	32.88
Arable weeds	5.59	41.25
Grassland weeds	1.56	24.21
Small mammals	1.53	12.75
Birds	4.00	45.00
Amphibians	1.22	13.58
Wild bees	1.91	30.39
Grasshoppers	3.00	37.36
Beetles	3.41	28.00
Butterflies	2.28	41.43
Spiders	4.21	32.88
Snails	2.00	12.71

Source: own calculations with SALCA-BD tool

Table 73 Description of codes used in the subsequent tables for energy use, biodiversity, model activities, policy uptake and farming systems

FARMIS-Code	Description	FARMIS-Code	Description
Energy-use components		EN_M_PPT	Energy use for crop protection
EN_TOTAL	Total energy use	EN_CARE_	Energy use for plant care
EN_ANIMA	Energy use for animal husbandry	EN_M_FER	Energy use for fertiliser spreading
EN_BUILD	Energy use for buildings	EN_ORGAN	Energy use for organic fertiliser storage
EN_FODDE	Energy use for purchased fodder	EN_MINER	Energy use for all mineral fertiliser
EN_TILLA	Energy use for tillage	EN_NITRO	Energy use for N-Fertiliser
EN_SEEDI	Energy use for seeding	EN_PHOSP	Energy use for P-Fertiliser
EN_PLAPR	Energy use for crop protection	EN_POTAS	Energy use for K-Fertiliser
EN_FERTI	Energy use for fertilisation	EN_M_HAR	Energy use for harvesting
EN_HARVE	Energy use for harvesting	EN_DRYIN	Energy use for drying
EN_OTHER	Energy use for other	EN_TRANS	Energy use for transport
EN_DEPOT	Energy use for fodder depots		
EN_STABL	Energy use for livestock housing (building)	*Biodiversity indicators*	
EN_FENCE	Energy use for fences		
EN_INSTA	Energy use in livestock housing systems	BI_H_AMP	Biodiversity score for amphibians as species with high ecological requirements
EN_MILKI	Energy use for milking		
EN_CONCE	Energy use for purchased concen-trates	BI_H_LOC	Biodiversity score for locusts as species with high ecological requirements
EN_ROUGH	Energy use for purchased roughage		
EN_SEEDS	Energy use for seeds	BI_H_CAR	Biodiversity score for carabids as species with high ecological requirements
EN_M_SEE	Energy use for seeding		
EN_INSEC	Energy use for insecticides	BI_H_BUT	Biodiversity score for butterflies as species with high ecological requirements
EN_FUNGI	Energy use for fungicides		
EN_HERBI	Energy use for herbicides	BI_H_SPI	Biodiversity score for spiders as species with high ecological requirements
EN_OTPPT	Energy use for other pesticides		

FARMIS-Code	Description
BI_G_TOT	Biodiversity score for general species
BI_G_ARA	Biodiversity score for arable land flora as general species
BI_G_GRA	Biodiversity score for grassland flora as general species
BI_G_SMA	Biodiversity score for small mammals as general species
BI_G_BIR	Biodiversity score for birds as general species
BI_G_AMP	Biodiversity score for amphibians as general species
BI_G_BEE	Biodiversity score for bees as general species
BI_G_LOC	Biodiversity score for locusts as general species
BI_G_CAR	Biodiversity score for carabids as general species
BI_G_BUT	Biodiversity score for butterflies as general species
BI_G_SPI	Biodiversity score for spiders as general species
BI_G_MOL	Biodiversity score for molluscs as general species
Model activities	
WHEAT___	Wheat (ha)
RYE_____	Rye (ha)
SPELT___	Spelt (ha)
OTHBRCER	Other bread cereals (ha)
BARLEY__	Barley (ha)
OATS____	Oats (ha)
TRITICAL	Triticale (ha)
OTHFOCER	Other fodder cereals (ha)
MAIZE___	Grain maize (ha)
FODMAIZE	Fodder maize or silage maize (ha)
POTATOES	Potatoes (ha)
SUGABEET	Sugar beet (ha)
FODROOTS	Fodder root crops (ha)
RAPE____	Rape (ha)
OTHOILS_	Other oilseed crops (ha)
SUNFLOWE	Sunflower (ha)
FIELBEAN	Field beans (ha)
FIELDPEA	Field peas (ha)
TOBACCO_	Tobacco (ha)
VEGETABL	Vegetables (ha)
OTHACROP	Other arable crops (ha)
MIXFALLO	Mixed fallow land (ha)
ROTFALLO	Rotational fallow land (ha)
BUFSTRIP	Buffer strips on arable land (ha)
GRASARAB	Short-term ley (ha)
GRASMEAD	Meadows (ha)
OTHEPAST	Pasture (ha)
EXTEPAST	Extensive pasture (ha)
ALPIMEAD	Alpine meadow (ha)
VINEYARD	Vineyards (ha)
FRUITS__	Fruits (ha)

FARMIS-Code	Description
BERRIES_	Berries (ha)
OTHPCROP	Other permanent crops (ha)
OTHAREA_	Other area (ha)
WOOD____	Wood (ha)
DAIRYCOW	Dairy cows (unit)
DAIRHEF3	Dairy breeding heifers 24-30 months (unit)
DAIRHEF2	Dairy breeding heifers 12-24 months (unit)
DAIRHEF1	Dairy breeding heifers 04-12 months (unit)
DAIRBUL3	Dairy breeding bulls 24-30 months (unit)
DAIRBUL2	Dairy breeding bulls 12-24 months (unit)
DAIRBUL1	Dairy breeding bulls 04-12 months (unit)
DAIRCALV	Dairy calves for breeding 01-04 month becoming DAIRBUL1 or DAIRHEF1 (unit)
SUCKLCOW	Suckler cows (unit)
SUCKBHEF	Suckler breeding heifers 12-24 months becoming a suckler cow (unit)
SUCKFHEF	Suckler fattening heifers >12 months (unit)
SUCKBBUL	Suckler breeding bulls >12 months (unit)
SUCKFBUL	Suckler fattening bulls >12 months (unit)
SUCKCALV	Suckler calves 1-12 month (unit)
FACATTLE	Fattening cattle (unit)
FACALVES	Fattening calves (unit)
HORSES__	Horses (unit)
SHEPMILK	Milk sheep (unit)
SHEPFATT	Fattening sheep (unit)
GOATS___	Goats (unit)
OROCLIVE	Other roughage consuming livestock (unit)
SOWS____	Sows for piglet production (unit)
PORK____	Pork fattening (unit)
LAYHENS_	Laying hens (unit)
BROILER_	Broiler (unit)
OPOULTRY	Other poultry (unit)
OANIMALS	Other animals (unit)
Policy uptake/Intensity level	
MAIN	Standard activity
INT	Intensive activity
LIN	Less intensive meadows
EXT	Extensive meadows
EXS	Extenso payments
Farming system	
CON	Conventional farming
ORG	Organic farming

Table 74 Assumptions for fossil energy use for mineral fertiliser

Fertiliser	Fossil energy use	Unit
Mineral nitrogen fertiliser	69.31	MJ-eq/kg
Mineral phosphate fertiliser	27.13	MJ-eq/kg
Mineral potassium fertiliser	13.34	MJ-eq/kg

Source: ecoinvent (2008), composition of fertiliser types derived from AGRIDEA (2008)

Table 75 Classification of crop production activities

Feedstuff	Conventional production (MJ/t)	Organic production (MJ/t)	Transport store to farm (MJ/tkm)	Average transport distance (store to farm)
Calf starter	9,550	8,435	3.06	10.00
Complete feed for breed non-suckling pigs	6,767	5,595	3.06	10.00
Complete feed for breed suckling pigs	6,767	5,595	3.06	10.00
Complete feed for fattening pigs	8,907	8,202	3.06	10.00
Complete feed for laying hens	9,550	8,435	3.06	10.00
Complete feed for rabbits	9,550	8,435	3.06	10.00
Concentrates for deers	9,550	8,435	3.06	10.00
Energy-balance feed for sheep	6,153	5,123	3.06	10.00
Fattening feed for cattle	8,408	7,495	3.06	10.00
Fattening feed for lambs	9,550	8,435	3.06	10.00
Fattening feed for poultry	9,550	8,435	3.06	10.00
Feed cereals	6,153	5,123	3.06	10.00
Feedstuff for pigkets	9,550	8,435	3.06	10.00
Gras silage	553	1,429	1.00	10.00
Grass	168	698	1.00	10.00
Hay with good quality	2,141	2,250	1.00	10.00
Hay with low quality	1,133	1,137	1.00	10.00
Maize silage	389	1,148	1.00	10.00
Milk supplement	78,340	76,930	3.06	10.00
MIlked powder for fattening calves	78,340	76,930	3.06	10.00
Mineral feed	2,816	2,816	3.06	10.00
Mineral feed for small ruminants	9,550	8,435	3.06	10.00
Performance feed for dairy cows	7,036	6,116	3.06	10.00
Performance feed for sheep or goats	7,036	6,116	3.06	10.00
Protein-balance feed for sheep	11,226	10,459	3.06	10.00
Skimmed milk	5,050	5,050	3.06	10.00
Soybean meal extract	11,226	10,459	3.06	10.00
Suckling milk	5,050	5,050	-	10.00
Sugarbeet chips	32,820	32,820	1.00	10.00

Source: Alig (2007), Nemecek (2007)

Table 76 Detailed assumptions on activity-specific fossil energy use per ha and year (MJ)

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Lowlands	RYE_____	ORG	EXS	10,220	4,053						527	345							855	142				3,237	869	193
Lowlands	RYE_____	CON	INT	18,633	3,728						456	345		2	374	6	202		1,319	26	5,041	1,506	805	3,304	1,244	276
Lowlands	RYE_____	CON	EXS	18,633	3,728						456	345		2	374	6	202		1,319	26	5,041	1,506	805	3,304	1,244	276
Lowlands	OATS____	ORG	EXS	9,104	3,890						443	345							552	92				2,970	645	168
Lowlands	OATS____	CON	INT	19,336	3,728						429	345		73	113		319		1,165		5,564	1,402	1,740	3,122	1,060	276
Lowlands	OATS____	CON	EXS	16,205	3,728						429	345		1	332		118		1,393	37	3,983	827	1,002	2,954	839	218
Lowlands	MAIZE___	ORG	MAIN	22,910	2,667						344	345				457		613	1,728	278				2,318	13,801	359
Lowlands	MAIZE___	CON	MAIN	34,502	2,505						431	345		13	348	457	286	306	1,752	93	4,961	1,714	2,077	2,318	16,468	428
Lowlands	TRITICAL	ORG	EXS	10,391	4,049						587	345							1,011	168				3,186	856	190
Lowlands	TRITICAL	CON	INT	21,055	3,728						612	345		42	388	30	385		1,469	19	6,720	1,597	849	3,296	1,289	286
Lowlands	TRITICAL	CON	EXS	19,432	3,728						612	345		2	316	3	185		1,535	30	6,014	1,370	647	3,212	1,173	260
Lowlands	PULSES__	ORG	MAIN	10,341	2,748						1,685	345						153	804	131				2,318	2,009	148
Lowlands	PULSES__	CON	MAIN	13,500	2,667						1,581	345	7	45	293		235		434	8		1,537	1,472	2,318	2,382	176
Lowlands	RAPE____	ORG	EXS	10,296	2,667						78	345						306	1,796	279				2,318	2,414	93
Lowlands	RAPE____	CON	INT	20,010	2,505						78	345	29		468	54	588		1,585	67	6,485	1,013	617	2,318	3,714	144
Lowlands	RAPE____	CON	EXS	16,202	2,505						78	345			532	44	319		2,061	145	4,209	321		2,318	3,202	124
Lowlands	SUNFLOWE	ORG	MAIN	13,452	2,505						53	345							776					2,318	7,309	145
Lowlands	SUNFLOWE	CON	MAIN	22,344	2,505						53	345		2	379	52	168		776		2,649	1,234	4,409	2,318	7,309	145
Lowlands	OTHOILS_	ORG	MAIN	9,806	2,667						884	345						306	201	32				2,318	2,923	130
Lowlands	OTHOILS_	CON	MAIN	13,606	2,505						831	345			667		202		668	46		1,458	1,375	2,318	3,055	135
Lowlands	POTATOES	ORG	MAIN	18,653	4,046						7,438	912		51			740		824	129				4,409		106
Lowlands	POTATOES	CON	MAIN	26,269	4,442						7,376	912	3	720	195		1,143		2,058	140	2,082	415	2,201	4,409		174
Lowlands	SUGABEET	ORG	MAIN	12,988	2,505						55	345						613	2,222	106				6,808		334
Lowlands	SUGABEET	CON	MAIN	21,829	2,505						55	345	21	32	703	17	740	613	2,222	106	3,097	1,480	2,752	6,808		334
Lowlands	VEGETABL	ORG	MAIN	20,319	3,780	7,059					99	345						6,149	731	1,966						190
Lowlands	VEGETABL	CON	MAIN	31,699	3,231	9,751					110	345	1,719	470	699		1,176	6,149	1,165		3,825	1,079	1,768			212

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Lowlands	FRUITS__	ORG	MAIN	10,045		7,059					99								731	1,966						190
Lowlands	FRUITS__	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Lowlands	VINEYARD	ORG	MAIN	10,045		7,059					99								731	1,966						190
Lowlands	VINEYARD	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Lowlands	TOBACCO_	ORG	MAIN	20,319	3,780	7,059					99	345						6,149	731	1,966						190
Lowlands	TOBACCO_	CON	MAIN	31,699	3,231	9,751					110	345	1,719	470	699		1,176	6,149	1,165		3,825	1,079	1,768			212
Lowlands	GRASARAB	ORG	MAIN	11,527	829		3,899		16		256	115							1,058	163		8		5,162		22
Lowlands	GRASARAB	CON	MAIN	14,400	829		4,332		16		256	115			66				1,352	178	1,321	530		5,380		25
Lowlands	FODMAIZE	ORG	MAIN	12,371	2,667						372	345						613	1,452	234				6,690		
Lowlands	FODMAIZE	CON	MAIN	19,754	2,505						465	345		14	230		168	306	2,168	160	3,434	1,609	1,200	7,151		
Lowlands	FODROOTS	ORG	MAIN	13,482	2,505						55	345						613	2,550	158				6,808		448
Lowlands	FODROOTS	CON	MAIN	19,003	2,505						55	345	33	5	506		487	613	2,550	158	1,241	1,019	2,230	6,808		448
Lowlands	DAIRYCOW	ORG	MAIN	8,178				3,668		4,511																
Lowlands	DAIRYCOW	CON	MAIN	8,178				3,668		4,511																
Lowlands	SUCKLCOW	ORG	MAIN	6,856				3,281		3,575																
Lowlands	SUCKLCOW	CON	MAIN	6,856				3,281		3,575																
Lowlands	PORK____	ORG	MAIN	522				346		176																
Lowlands	PORK____	CON	MAIN	2,231				305		1,926																
Lowlands	SOWS____	ORG	MAIN	1,382				915		467																
Lowlands	SOWS____	CON	MAIN	5,906				807		5,099																
Lowlands	LAYHENS_	ORG	MAIN	140				69		71																
Lowlands	LAYHENS_	CON	MAIN	140				69		71																
Lowlands	BROILER_	ORG	MAIN	114				56		58																
Lowlands	BROILER_	CON	MAIN	114				56		58																
Lowlands	OPOULTRY	ORG	MAIN	1,463				720		743																
Lowlands	OPOULTRY	CON	MAIN	1,463				720		743																
Lowlands	OANIMALS	ORG	MAIN	210				104		107																
Lowlands	OANIMALS	CON	MAIN	210				104		107																

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Lowlands	WHEAT___	ORG	EXS	10,562	4,045						648	345							1,167	194				3,135	842	187
Lowlands	WHEAT___	CON	INT	23,477	3,728						768	345		82	402	55	568		1,619	11	8,400	1,689	893	3,288	1,334	296
Lowlands	WHEAT___	CON	EXS	20,230	3,728						768	345		2	258		169		1,752	33	6,987	1,234	489	3,119	1,101	244
Lowlands	SPELT___	ORG	EXS	9,104	3,890						443	345							552	92				2,970	645	168
Lowlands	SPELT___	CON	INT	19,336	3,728						429	345		73	113		319		1,165		5,564	1,402	1,740	3,122	1,060	276
Lowlands	SPELT___	CON	EXS	16,205	3,728						429	345		1	332		118		1,393	37	3,983	827	1,002	2,954	839	218
Lowlands	BARLEY__	ORG	EXS	9,818	4,036						417	345							914	152				3,037	728	189
Lowlands	BARLEY__	CON	INT	20,894	3,728						394	345		84	417	71	451		1,751	34	6,178	1,613	1,093	3,212	1,209	314
Lowlands	BARLEY__	CON	EXS	18,753	3,728						394	345		1	373	6	200		1,700	26	5,570	1,318	860	3,026	957	249
Lowlands	OTHBRCER	ORG	MAIN	9,762	3,983						496	345							808	134				3,070	746	181
Lowlands	OTHBRCER	CON	MAIN	20,335	3,728						495	345		63	284	26	372		1,404	14	6,150	1,522	1,254	3,210	1,181	287
Lowlands	OTHBRCER	CON	EXS	18,005	3,728						495	345		2	334	3	161		1,511	32	5,113	1,142	832	3,071	996	241
Lowlands	OTHFOCER	ORG	EXS	9,771	3,992						482	345							825	137				3,065	743	182
Lowlands	OTHFOCER	CON	INT	20,428	3,728						478	345		66	306	34	385		1,462	18	6,154	1,538	1,227	3,210	1,186	292
Lowlands	OTHFOCER	CON	EXS	18,130	3,728						478	345		2	340	3	168		1,543	31	5,189	1,172	836	3,064	989	242
Lowlands	FIELBEAN	ORG	MAIN	10,188	2,667						1,345	345						306	804	131				2,318	2,115	156
Lowlands	FIELBEAN	CON	MAIN	13,084	2,667						1,273	345			101		168		388			1,549	1,736	2,318	2,364	175
Lowlands	FIELDPEA	ORG	MAIN	10,495	2,829						2,025	345							804	131				2,318	1,903	141
Lowlands	FIELDPEA	CON	MAIN	13,916	2,667						1,890	345	14	90	486		303		480	16		1,525	1,207	2,318	2,400	177
Lowlands	OTHACROP	ORG	MAIN	10,220	4,053						527	345							855	142				3,237	869	193
Lowlands	OTHACROP	CON	MAIN	18,633	3,728						456	345		2	374	6	202		1,319	26	5,041	1,506	805	3,304	1,244	276
Lowlands	GRASMEAD	ORG	INT	13,038		281	4,927		36										1,246	200		25		6,297		27
Lowlands	GRASMEAD	ORG	EXT	2,146			589																	1,541		16
Lowlands	GRASMEAD	ORG	LIN	8,340			3,670		24										768	118				3,745		15
Lowlands	GRASMEAD	CON	INT	17,716		331	5,796		36						97				1,776	223	1,946	799		6,681		31
Lowlands	GRASMEAD	CON	EXT	2,146			589																	1,541		16
Lowlands	GRASMEAD	CON	LIN	8,930			3,670		29						12				891	116	443	4		3,750		15
Lowlands	OTHEPAST	ORG	MAIN	1,259					241										704	120				194		

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Lowlands	OTHEPAST	CON	MAIN	5,160					241						80				1,529	102	2,950	28		230		
Lowlands	EXTEPAST	ORG	MAIN	1,259					241										704	120				194		
Lowlands	EXTEPAST	CON	MAIN	5,160					241						80				1,529	102	2,950	28		230		
Lowlands	ALPIMEAD	ORG	MAIN	1,303			29		229										669	114				261		1
Lowlands	ALPIMEAD	CON	MAIN	5,009			29		229						76				1,453	97	2,802	27		295		1
Lowlands	BERRIES_	ORG	MAIN	10,045		7,059					99								731	1,966						190
Lowlands	BERRIES_	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Lowlands	OTHPCROP	ORG	MAIN	10,045		7,059					99								731	1,966						190
Lowlands	OTHPCROP	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Lowlands	DAIRBUL3	ORG	MAIN	5,012				2,317		2,695																
Lowlands	DAIRBUL3	CON	MAIN	5,012				2,317		2,695																
Lowlands	DAIRBUL2	ORG	MAIN	3,341				1,544		1,797																
Lowlands	DAIRBUL2	CON	MAIN	3,341				1,544		1,797																
Lowlands	DAIRBUL1	ORG	MAIN	2,506				1,158		1,348																
Lowlands	DAIRBUL1	CON	MAIN	2,506				1,158		1,348																
Lowlands	DAIRCALV	ORG	MAIN	835				386		449																
Lowlands	DAIRCALV	CON	MAIN	835				386		449																
Lowlands	SUCKBHEF	ORG	MAIN	3,428				1,640		1,788																
Lowlands	SUCKBHEF	CON	MAIN	3,428				1,640		1,788																
Lowlands	SUCKFHEF	ORG	MAIN	3,428				1,640		1,788																
Lowlands	SUCKFHEF	CON	MAIN	3,428				1,640		1,788																
Lowlands	SUCKBBUL	ORG	MAIN	3,428				1,640		1,788																
Lowlands	SUCKBBUL	CON	MAIN	3,428				1,640		1,788																
Lowlands	SUCKFBUL	ORG	MAIN	3,428				1,640		1,788																
Lowlands	SUCKFBUL	CON	MAIN	3,428				1,640		1,788																
Lowlands	SUCKCALV	ORG	MAIN	1,457				697		760																
Lowlands	SUCKCALV	CON	MAIN	1,457				697		760																
Lowlands	FACATTLE	ORG	MAIN	2,571				1,230		1,341																

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Lowlands	FACATTLE	CON	MAIN	2,571				1,230		1,341																
Lowlands	FACALVES	ORG	MAIN	856				408		447																
Lowlands	FACALVES	CON	MAIN	856				408		447																
Lowlands	HORSES__	ORG	MAIN	587				289		298																
Lowlands	HORSES__	CON	MAIN	587				289		298																
Lowlands	SHEPMILK	ORG	MAIN	1,800				861		939																
Lowlands	SHEPMILK	CON	MAIN	1,800				861		939																
Lowlands	SHEPFATT	ORG	MAIN	1,800				861		939																
Lowlands	SHEPFATT	CON	MAIN	1,800				861		939																
Lowlands	GOATS___	ORG	MAIN	1,714				820		894																
Lowlands	GOATS___	CON	MAIN	1,714				820		894																
Lowlands	OROCLIVE	ORG	MAIN	587				289		298																
Lowlands	OROCLIVE	CON	MAIN	587				289		298																
Lowlands	DAIRYHEF3	ORG	MAIN	5,012				2,317		2,695																
Lowlands	DAIRYHEF3	CON	MAIN	5,012				2,317		2,695																
Lowlands	DAIRYHEF2	ORG	MAIN	3,341				1,544		1,797																
Lowlands	DAIRYHEF2	CON	MAIN	3,341				1,544		1,797																
Lowlands	DAIRYHEF1	ORG	MAIN	2,506				1,158		1,348																
Lowlands	DAIRYHEF1	CON	MAIN	2,506				1,158		1,348																
Lowlands	FCATCALV	ORG	MAIN	856				408		447																
Lowlands	FCATCALV	CON	MAIN	856				408		447																
Hills	RYE_____	ORG	EXS	10,393	4,053						527	345							905	150				3,291	920	204
Hills	RYE_____	CON	INT	16,265	3,728						456	345		2	349		202		1,868	113	3,671	746		3,289	1,225	271
Hills	RYE_____	CON	EXS	16,265	3,728						456	345		2	349		202		1,868	113	3,671	746		3,289	1,225	271
Hills	OATS____	ORG	EXS	9,148	3,890						443	345							567	94				2,981	656	171
Hills	OATS____	CON	INT	19,068	3,728						429	345		73	113		319		1,165		5,566	1,331	1,651	3,081	1,006	262
Hills	OATS____	CON	EXS	12,821	3,728						429	345		1	117		118		2,201	164	1,777			2,927	804	209
Hills	MAIZE___	ORG	MAIN	22,910	2,667						344	345				457		613	1,728	278				2,318	13,801	359

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Hills	MAIZE___	CON	MAIN	34,502	2,505						431	345		13	348	457	286	306	1,752	93	4,961	1,714	2,077	2,318	16,468	428
Hills	TRITICAL	ORG	EXS	10,489	4,048						588	345							1,039	173				3,217	885	196
Hills	TRITICAL	CON	INT	19,741	3,728						612	345		24	295	40	335		1,712	57	6,125	1,204	479	3,265	1,246	276
Hills	TRITICAL	CON	EXS	17,396	3,728						612	345		2	265		185		1,650	79	4,910	907	160	3,177	1,125	249
Hills	PULSES__	ORG	MAIN	10,341	2,748						1,685	345						153	804	131				2,318	2,009	148
Hills	PULSES__	CON	MAIN	13,500	2,667						1,581	345	7	45	293		235		434	8		1,537	1,472	2,318	2,382	176
Hills	RAPE____	ORG	EXS	10,296	2,667						78	345						306	1,796	279				2,318	2,414	93
Hills	RAPE____	CON	INT	20,010	2,505						78	345	29		468	54	588		1,585	67	6,485	1,013	617	2,318	3,714	144
Hills	RAPE____	CON	EXS	16,202	2,505						78	345			532	44	319		2,061	145	4,209	321		2,318	3,202	124
Hills	SUNFLOWE	ORG	MAIN	13,452	2,505						53	345							776					2,318	7,309	145
Hills	SUNFLOWE	CON	MAIN	22,344	2,505						53	345		2	379	52	168		776		2,649	1,234	4,409	2,318	7,309	145
Hills	OTHOILS_	ORG	MAIN	9,806	2,667						884	345						306	201	32				2,318	2,923	130
Hills	OTHOILS_	CON	MAIN	13,606	2,505						831	345			667		202		668	46		1,458	1,375	2,318	3,055	135
Hills	POTATOES	ORG	MAIN	18,375	4,046						7,438	912		53			286		966	151				4,409		115
Hills	POTATOES	CON	MAIN	23,677	4,442						7,376	912	3	720	195		1,143		2,476	201	601		1,055	4,409		145
Hills	SUGABEET	ORG	MAIN	12,988	2,505						55	345						613	2,222	106				6,808		334
Hills	SUGABEET	CON	MAIN	21,829	2,505						55	345	21	32	703	17	740	613	2,222	106	3,097	1,480	2,752	6,808		334
Hills	VEGETABL	ORG	MAIN	20,319	3,780	7,059					99	345						6,149	731	1,966						190
Hills	VEGETABL	CON	MAIN	31,699	3,231	9,751					110	345	1,719	470	699		1,176	6,149	1,165		3,825	1,079	1,768			212
Hills	FRUITS__	ORG	MAIN	10,045		7,059					99								731	1,966						190
Hills	FRUITS__	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Hills	VINEYARD	ORG	MAIN	10,045		7,059					99								731	1,966						190
Hills	VINEYARD	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Hills	TOBACCO_	ORG	MAIN	20,319	3,780	7,059					99	345						6,149	731	1,966						190
Hills	TOBACCO_	CON	MAIN	31,699	3,231	9,751					110	345	1,719	470	699		1,176	6,149	1,165		3,825	1,079	1,768			212
Hills	GRASARAB	ORG	MAIN	11,962	829		4,357		16		256	115							1,058	163		8		5,141		20
Hills	GRASARAB	CON	MAIN	12,643	829		3,736		16		256	115			66				1,152	144	1,074	370		4,869		18
Hills	FODMAIZE	ORG	MAIN	11,854	2,667						372	345						613	1,197	193				6,469		

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Hills	FODMAIZE	CON	MAIN	15,709	2,505						465	345		14	184		168	306	2,480	209	1,306	555	299	6,874		
Hills	FODROOTS	ORG	MAIN	13,482	2,505						55	345						613	2,550	158				6,808		448
Hills	FODROOTS	CON	MAIN	19,003	2,505						55	345	33	5	506		487	613	2,550	158	1,241	1,019	2,230	6,808		448
Hills	DAIRYCOW	ORG	MAIN	8,058				3,534		4,524																
Hills	DAIRYCOW	CON	MAIN	8,058				3,534		4,524																
Hills	SUCKLCOW	ORG	MAIN	6,691				3,098		3,593																
Hills	SUCKLCOW	CON	MAIN	6,691				3,098		3,593																
Hills	PORK____	ORG	MAIN	522				346		176																
Hills	PORK____	CON	MAIN	2,231				305		1,926																
Hills	SOWS____	ORG	MAIN	1,382				915		467																
Hills	SOWS____	CON	MAIN	5,906				807		5,099																
Hills	LAYHENS_	ORG	MAIN	140				69		71																
Hills	LAYHENS_	CON	MAIN	140				69		71																
Hills	BROILER_	ORG	MAIN	114				56		58																
Hills	BROILER_	CON	MAIN	114				56		58																
Hills	OPOULTRY	ORG	MAIN	1,463				720		743																
Hills	OPOULTRY	CON	MAIN	1,463				720		743																
Hills	OANIMALS	ORG	MAIN	210				104		107																
Hills	OANIMALS	CON	MAIN	210				104		107																
Hills	WHEAT___	ORG	EXS	10,585	4,044						648	345							1,174	195				3,142	849	188
Hills	WHEAT___	CON	INT	23,216	3,728						768	345		45	240	79	469		1,555	0	8,579	1,661	958	3,240	1,267	281
Hills	WHEAT___	CON	EXS	18,527	3,728						768	345		2	181		168		1,433	45	6,150	1,068	320	3,065	1,026	227
Hills	SPELT___	ORG	EXS	9,148	3,890						443	345							567	94				2,981	656	171
Hills	SPELT___	CON	INT	19,068	3,728						429	345		73	113		319		1,165		5,566	1,331	1,651	3,081	1,006	262
Hills	SPELT___	CON	EXS	12,821	3,728						429	345		1	117		118		2,201	164	1,777			2,927	804	209
Hills	BARLEY__	ORG	EXS	9,901	4,044						415	345							948	158				3,053	745	194
Hills	BARLEY__	CON	INT	19,373	3,728						396	345		62	350	89	366		1,303	22	5,765	1,249	1,102	3,162	1,139	296
Hills	BARLEY__	CON	EXS	15,443	3,728						396	345		1	331		195		1,894	117	3,805	491		2,992	910	237

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Hills	OTHBRCER	ORG	MAIN	9,835	3,984						495	345							832	138				3,090	765	185
Hills	OTHBRCER	CON	MAIN	19,398	3,728						496	345		51	233	34	335		1,411	27	5,829	1,264	1,073	3,170	1,129	274
Hills	OTHBRCER	CON	EXS	15,175	3,728						496	345		2	219		160		1,920	121	3,436	461	64	3,040	954	231
Hills	OTHFOCER	ORG	EXS	9,846	3,994						482	345							852	141				3,084	762	187
Hills	OTHFOCER	CON	INT	19,394	3,728						479	345		53	253	43	340		1,393	26	5,818	1,261	1,078	3,169	1,130	278
Hills	OTHFOCER	CON	EXS	15,220	3,728						479	345		2	238		166		1,915	120	3,498	466	53	3,032	946	232
Hills	FIELBEAN	ORG	MAIN	10,188	2,667						1,345	345						306	804	131				2,318	2,115	156
Hills	FIELBEAN	CON	MAIN	13,084	2,667						1,273	345			101		168		388			1,549	1,736	2,318	2,364	175
Hills	FIELDPEA	ORG	MAIN	10,495	2,829						2,025	345							804	131				2,318	1,903	141
Hills	FIELDPEA	CON	MAIN	13,916	2,667						1,890	345	14	90	486		303		480	16		1,525	1,207	2,318	2,400	177
Hills	OTHACROP	ORG	MAIN	10,393	4,053						527	345							905	150				3,291	920	204
Hills	OTHACROP	CON	MAIN	16,265	3,728						456	345		2	349		202		1,868	113	3,671	746		3,289	1,225	271
Hills	GRASMEAD	ORG	INT	11,533		345	4,739		36										1,057	170		22		5,144		21
Hills	GRASMEAD	ORG	EXT	2,146			589																	1,541		16
Hills	GRASMEAD	ORG	LIN	7,076		81	3,093		22										640	99				3,129		13
Hills	GRASMEAD	CON	INT	15,667		399	5,575		36						96				1,538	191	1,655	687		5,466		25
Hills	GRASMEAD	CON	EXT	2,146			589																	1,541		16
Hills	GRASMEAD	CON	LIN	7,546		82	3,093		28						11				720	97	363	3		3,135		13
Hills	OTHEPAST	ORG	MAIN	1,078		16			241										551	94				176		
Hills	OTHEPAST	CON	MAIN	4,172		27			241						76				1,083	84	2,420	23		219		
Hills	EXTEPAST	ORG	MAIN	1,078		16			241										551	94				176		
Hills	EXTEPAST	CON	MAIN	4,172		27			241						76				1,083	84	2,420	23		219		
Hills	ALPIMEAD	ORG	MAIN	1,132		16	29		229										523	89				244		1
Hills	ALPIMEAD	CON	MAIN	4,071		25	29		229						72				1,029	79	2,299	22		285		1
Hills	BERRIES_	ORG	MAIN	10,045		7,059					99								731	1,966						190
Hills	BERRIES_	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Hills	OTHPCROP	ORG	MAIN	10,045		7,059					99								731	1,966						190
Hills	OTHPCROP	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Hills	DAIRBUL3	ORG	MAIN	4,924				2,219		2,705																
Hills	DAIRBUL3	CON	MAIN	4,924				2,219		2,705																
Hills	DAIRBUL2	ORG	MAIN	3,283				1,479		1,803																
Hills	DAIRBUL2	CON	MAIN	3,283				1,479		1,803																
Hills	DAIRBUL1	ORG	MAIN	2,462				1,110		1,352																
Hills	DAIRBUL1	CON	MAIN	2,462				1,110		1,352																
Hills	DAIRCALV	ORG	MAIN	821				370		451																
Hills	DAIRCALV	CON	MAIN	821				370		451																
Hills	SUCKBHEF	ORG	MAIN	3,346				1,549		1,797																
Hills	SUCKBHEF	CON	MAIN	3,346				1,549		1,797																
Hills	SUCKFHEF	ORG	MAIN	3,346				1,549		1,797																
Hills	SUCKFHEF	CON	MAIN	3,346				1,549		1,797																
Hills	SUCKBBUL	ORG	MAIN	3,346				1,549		1,797																
Hills	SUCKBBUL	CON	MAIN	3,346				1,549		1,797																
Hills	SUCKFBUL	ORG	MAIN	3,346				1,549		1,797																
Hills	SUCKFBUL	CON	MAIN	3,346				1,549		1,797																
Hills	SUCKCALV	ORG	MAIN	1,422				658		764																
Hills	SUCKCALV	CON	MAIN	1,422				658		764																
Hills	FACATTLE	ORG	MAIN	2,509				1,162		1,347																
Hills	FACATTLE	CON	MAIN	2,509				1,162		1,347																
Hills	FACALVES	ORG	MAIN	845				396		448																
Hills	FACALVES	CON	MAIN	845				396		448																
Hills	HORSES__	ORG	MAIN	587				289		298																
Hills	HORSES__	CON	MAIN	587				289		298																
Hills	SHEPMILK	ORG	MAIN	1,800				861		939																
Hills	SHEPMILK	CON	MAIN	1,800				861		939																
Hills	SHEPFATT	ORG	MAIN	1,800				861		939																
Hills	SHEPFATT	CON	MAIN	1,800				861		939																

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Hills	GOATS___	ORG	MAIN	1,714				820		894																
Hills	GOATS___	CON	MAIN	1,714				820		894																
Hills	OROCLIVE	ORG	MAIN	587				289		298																
Hills	OROCLIVE	CON	MAIN	587				289		298																
Hills	DAIRYHEF3	ORG	MAIN	4,924				2,219		2,705																
Hills	DAIRYHEF3	CON	MAIN	4,924				2,219		2,705																
Hills	DAIRYHEF2	ORG	MAIN	3,283				1,479		1,803																
Hills	DAIRYHEF2	CON	MAIN	3,283				1,479		1,803																
Hills	DAIRYHEF1	ORG	MAIN	2,462				1,110		1,352																
Hills	DAIRYHEF1	CON	MAIN	2,462				1,110		1,352																
Hills	FCATCALV	ORG	MAIN	845				396		448																
Hills	FCATCALV	CON	MAIN	845				396		448																
Mountains	RYE_____	ORG	EXS	10,393	4,053						527	345							905	150				3,291	920	204
Mountains	RYE_____	CON	INT	16,265	3,728						456	345		2	349		202		1,868	113	3,671	746		3,289	1,225	271
Mountains	RYE_____	CON	EXS	16,265	3,728						456	345		2	349		202		1,868	113	3,671	746		3,289	1,225	271
Mountains	OATS____	ORG	EXS	9,120	3,890						443	345							557	93				2,974	649	169
Mountains	OATS____	CON	INT	16,503	3,728						429	345		73	113		319		1,165		4,155	1,024	1,271	2,905	774	201
Mountains	OATS____	CON	EXS	13,533	3,728						429	345		1	396		219		1,755	90	2,506	26	291	2,855	708	184
Mountains	MAIZE___	ORG	MAIN	22,910	2,667						344	345				457		613	1,728	278				2,318	13,801	359
Mountains	MAIZE___	CON	MAIN	34,502	2,505						431	345		13	348	457	286	306	1,752	93	4,961	1,714	2,077	2,318	16,468	428
Mountains	TRITICAL	ORG	EXS	10,294	4,016						601	345							935	155				3,196	857	190
Mountains	TRITICAL	CON	INT	19,487	3,728						614	345		25	289	40	333		1,714	57	5,978	1,164	461	3,247	1,220	270
Mountains	TRITICAL	CON	EXS	16,889	3,738						608	345		2	255		175		1,600	82	4,577	852	141	3,171	1,102	244
Mountains	PULSES__	ORG	MAIN	10,341	2,748						1,685	345						153	804	131				2,318	2,009	148
Mountains	PULSES__	CON	MAIN	13,500	2,667						1,581	345	7	45	293		235		434	8		1,537	1,472	2,318	2,382	176
Mountains	RAPE____	ORG	EXS	10,296	2,667						78	345						306	1,796	279				2,318	2,414	93
Mountains	RAPE____	CON	INT	20,010	2,505						78	345	29		468	54	588		1,585	67	6,485	1,013	617	2,318	3,714	144
Mountains	RAPE____	CON	EXS	16,202	2,505						78	345			532	44	319		2,061	145	4,209	321		2,318	3,202	124

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Mountains	SUNFLOWE	ORG	MAIN	13,452	2,505						53	345							776					2,318	7,309	145
Mountains	SUNFLOWE	CON	MAIN	22,344	2,505						53	345		2	379	52	168		776		2,649	1,234	4,409	2,318	7,309	145
Mountains	OTHOILS_	ORG	MAIN	9,806	2,667						884	345						306	201	32				2,318	2,923	130
Mountains	OTHOILS_	CON	MAIN	13,606	2,505						831	345			667		202		668	46		1,458	1,375	2,318	3,055	135
Mountains	POTATOES	ORG	MAIN	18,375	4,046						7,438	912		53			286		966	151				4,409		115
Mountains	POTATOES	CON	MAIN	23,677	4,442						7,376	912	3	720	195		1,143		2,476	201	601		1,055	4,409		145
Mountains	SUGABEET	ORG	MAIN	12,988	2,505						55	345						613	2,222	106				6,808		334
Mountains	SUGABEET	CON	MAIN	21,829	2,505						55	345	21	32	703	17	740	613	2,222	106	3,097	1,480	2,752	6,808		334
Mountains	VEGETABL	ORG	MAIN	20,319	3,780	7,059					99	345						6,149	731	1,966						190
Mountains	VEGETABL	CON	MAIN	31,699	3,231	9,751					110	345	1,719	470	699		1,176	6,149	1,165		3,825	1,079	1,768			212
Mountains	FRUITS__	ORG	MAIN	10,045		7,059					99								731	1,966						190
Mountains	FRUITS__	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Mountains	VINEYARD	ORG	MAIN	10,045		7,059					99								731	1,966						190
Mountains	VINEYARD	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Mountains	TOBACCO_	ORG	MAIN	20,319	3,780	7,059					99	345						6,149	731	1,966						190
Mountains	TOBACCO_	CON	MAIN	31,699	3,231	9,751					110	345	1,719	470	699		1,176	6,149	1,165		3,825	1,079	1,768			212
Mountains	GRASARAB	ORG	MAIN	12,831	829		5,273		16		256	115							1,058	163		8		5,098		17
Mountains	GRASARAB	CON	MAIN	13,398	829		4,522		16		256	115			66				1,152	144	1,074	370		4,841		14
Mountains	FODMAIZE	ORG	MAIN	11,854	2,667						372	345						613	1,197	193				6,469		
Mountains	FODMAIZE	CON	MAIN	15,709	2,505						465	345		14	184		168	306	2,480	209	1,306	555	299	6,874		
Mountains	FODROOTS	ORG	MAIN	13,482	2,505						55	345						613	2,550	158				6,808		448
Mountains	FODROOTS	CON	MAIN	19,003	2,505						55	345	33	5	506		487	613	2,550	158	1,241	1,019	2,230	6,808		448
Mountains	DAIRYCOW	ORG	MAIN	7,987				3,456		4,531																
Mountains	DAIRYCOW	CON	MAIN	7,987				3,456		4,531																
Mountains	SUCKLCOW	ORG	MAIN	6,481				2,866		3,615																
Mountains	SUCKLCOW	CON	MAIN	6,481				2,866		3,615																
Mountains	PORK____	ORG	MAIN	522				346		176																
Mountains	PORK____	CON	MAIN	2,231				305		1,926																

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Mountains	SOWS____	ORG	MAIN	1,382				915		467																
Mountains	SOWS____	CON	MAIN	5,906				807		5,099																
Mountains	LAYHENS_	ORG	MAIN	140				69		71																
Mountains	LAYHENS_	CON	MAIN	140				69		71																
Mountains	BROILER_	ORG	MAIN	114				56		58																
Mountains	BROILER_	CON	MAIN	114				56		58																
Mountains	OPOULTRY	ORG	MAIN	1,463				720		743																
Mountains	OPOULTRY	CON	MAIN	1,463				720		743																
Mountains	OANIMALS	ORG	MAIN	210				104		107																
Mountains	OANIMALS	CON	MAIN	210				104		107																
Mountains	WHEAT___	ORG	EXS	10,195	3,979						674	345							965	160				3,101	795	176
Mountains	WHEAT___	CON	INT	22,710	3,728						772	345		49	229	79	465		1,561	1	8,285	1,583	922	3,206	1,216	269
Mountains	WHEAT___	CON	EXS	17,514	3,747						760	345		2	160		149		1,331	50	5,483	958	281	3,052	979	217
Mountains	SPELT___	ORG	EXS	9,120	3,890						443	345							557	93				2,974	649	169
Mountains	SPELT___	CON	INT	16,503	3,728						429	345		73	113		319		1,165		4,155	1,024	1,271	2,905	774	201
Mountains	SPELT___	CON	EXS	13,533	3,728						429	345		1	396		219		1,755	90	2,506	26	291	2,855	708	184
Mountains	BARLEY__	ORG	EXS	9,288	3,923						437	345							641	107				2,991	670	174
Mountains	BARLEY__	CON	INT	17,376	3,728						418	345		69	191	29	335		1,210	7	4,646	1,090	1,206	2,984	888	231
Mountains	BARLEY__	CON	EXS	14,176	3,728						418	345		1	382		213		1,789	97	2,950	179	203	2,898	772	201
Mountains	OTHBRCER	ORG	MAIN	9,623	3,947						505	345							725	120				3,066	736	178
Mountains	OTHBRCER	CON	MAIN	17,872	3,728						501	345		53	199	22	328		1,394	24	4,982	1,093	934	3,058	975	235
Mountains	OTHBRCER	CON	EXS	15,004	3,732						499	345		2	337		200		1,700	88	3,423	387	213	2,990	878	211
Mountains	OTHFOCER	ORG	EXS	9,567	3,943						494	345							711	118				3,054	725	178
Mountains	OTHFOCER	CON	INT	17,789	3,728						487	345		56	198	23	329		1,363	21	4,926	1,093	979	3,046	961	234
Mountains	OTHFOCER	CON	EXS	14,866	3,732						485	345		2	344		202		1,715	90	3,344	352	212	2,975	861	210
Mountains	FIELBEAN	ORG	MAIN	10,188	2,667						1,345	345						306	804	131				2,318	2,115	156
Mountains	FIELBEAN	CON	MAIN	13,084	2,667						1,273	345			101		168		388			1,549	1,736	2,318	2,364	175
Mountains	FIELDPEA	ORG	MAIN	10,495	2,829						2,025	345							804	131				2,318	1,903	141

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Mountains	FIELDPEA	CON	MAIN	13,916	2,667						1,890	345	14	90	486		303		480	16		1,525	1,207	2,318	2,400	177
Mountains	OTHACROP	ORG	MAIN	10,393	4,053						527	345							905	150				3,291	920	204
Mountains	OTHACROP	CON	MAIN	16,265	3,728						456	345		2	349		202		1,868	113	3,671	746		3,289	1,225	271
Mountains	GRASMEAD	ORG	INT	9,240		312	4,212		36										746	120		16		3,786		12
Mountains	GRASMEAD	ORG	EXT	2,146			589																	1,541		16
Mountains	GRASMEAD	ORG	LIN	5,755		158	2,517		11										484	74				2,502		10
Mountains	GRASMEAD	CON	INT	12,419		359	4,956		36						92				1,190	137	1,121	502		4,012		15
Mountains	GRASMEAD	CON	EXT	2,146			589																	1,541		16
Mountains	GRASMEAD	CON	LIN	6,023		161	2,517		18						8				548	75	175	2		2,510		10
Mountains	OTHEPAST	ORG	MAIN	572		13			241										198	34				86		
Mountains	OTHEPAST	CON	MAIN	2,309		31			241						50				626	40	1,165	11		144		
Mountains	EXTEPAST	ORG	MAIN	572		13			241										198	34				86		
Mountains	EXTEPAST	CON	MAIN	2,309		31			241						50				626	40	1,165	11		144		
Mountains	ALPIMEAD	ORG	MAIN	651		12	29		229										188	32				159		1
Mountains	ALPIMEAD	CON	MAIN	2,301		30	29		229						48				595	38	1,107	11		214		1
Mountains	BERRIES_	ORG	MAIN	10,045		7,059					99								731	1,966						190
Mountains	BERRIES_	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Mountains	OTHPCROP	ORG	MAIN	10,045		7,059					99								731	1,966						190
Mountains	OTHPCROP	CON	MAIN	21,974		9,751					110		1,719	470	699		1,176		1,165		3,825	1,079	1,768			212
Mountains	DAIRBUL3	ORG	MAIN	4,866				2,155		2,711																
Mountains	DAIRBUL3	CON	MAIN	4,866				2,155		2,711																
Mountains	DAIRBUL2	ORG	MAIN	3,244				1,437		1,807																
Mountains	DAIRBUL2	CON	MAIN	3,244				1,437		1,807																
Mountains	DAIRBUL1	ORG	MAIN	2,433				1,078		1,355																
Mountains	DAIRBUL1	CON	MAIN	2,433				1,078		1,355																
Mountains	DAIRCALV	ORG	MAIN	811				359		452																
Mountains	DAIRCALV	CON	MAIN	811				359		452																
Mountains	SUCKBHEF	ORG	MAIN	3,240				1,433		1,808																

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Mountains	SUCKBHEF	CON	MAIN	3,240				1,433		1,808																
Mountains	SUCKFHEF	ORG	MAIN	3,240				1,433		1,808																
Mountains	SUCKFHEF	CON	MAIN	3,240				1,433		1,808																
Mountains	SUCKBBUL	ORG	MAIN	3,240				1,433		1,808																
Mountains	SUCKBBUL	CON	MAIN	3,240				1,433		1,808																
Mountains	SUCKFBUL	ORG	MAIN	3,240				1,433		1,808																
Mountains	SUCKFBUL	CON	MAIN	3,240				1,433		1,808																
Mountains	SUCKCALV	ORG	MAIN	1,377				609		768																
Mountains	SUCKCALV	CON	MAIN	1,377				609		768																
Mountains	FACATTLE	ORG	MAIN	2,430				1,075		1,356																
Mountains	FACATTLE	CON	MAIN	2,430				1,075		1,356																
Mountains	FACALVES	ORG	MAIN	834				385		449																
Mountains	FACALVES	CON	MAIN	834				385		449																
Mountains	HORSES__	ORG	MAIN	587				289		298																
Mountains	HORSES__	CON	MAIN	587				289		298																
Mountains	SHEPMILK	ORG	MAIN	1,800				861		939																
Mountains	SHEPMILK	CON	MAIN	1,800				861		939																
Mountains	SHEPFATT	ORG	MAIN	1,800				861		939																
Mountains	SHEPFATT	CON	MAIN	1,800				861		939																
Mountains	GOATS___	ORG	MAIN	1,714				820		894																
Mountains	GOATS___	CON	MAIN	1,714				820		894																
Mountains	OROCLIVE	ORG	MAIN	587				289		298																
Mountains	OROCLIVE	CON	MAIN	587				289		298																
Mountains	DAIRYHEF3	ORG	MAIN	4,866				2,155		2,711																
Mountains	DAIRYHEF3	CON	MAIN	4,866				2,155		2,711																
Mountains	DAIRYHEF2	ORG	MAIN	3,244				1,437		1,807																
Mountains	DAIRYHEF2	CON	MAIN	3,244				1,437		1,807																
Mountains	DAIRYHEF1	ORG	MAIN	2,433				1,078		1,355																

Region	Activity	Farming system	Policy uptake	EN_TOTAL	EN_TILLA	EN_OTHER	EN_DEPOT	EN_STABL	EN_FENCE	EN_INSTA	EN_SEEDS	EN_M_SEE	EN_INSEC	EN_FUNGI	EN_HERBI	EN_OTPPT	EN_M_PPT	EN_CARE_	EN_M_FER	EN_ORGAN	EN_NITRO	EN_PHOSP	EN_POTAS	EN_M_HAR	EN_DRYIN	EN_TRANS
Mountains	DAIRYHEF1	CON	MAIN	2,433				1,078		1,355																
Mountains	FCATCALV	ORG	MAIN	834				385		449																
Mountains	FCATCALV	CON	MAIN	834				385		449																

Source: own compilation based on Nemecek *et al.* (2005), ecoinvent (2009)

Table 77 Assumptions for habitat quality

Region	Activity	Farming system	Policy uptake	BI_H_AMP	BI_H_LOC	BI_H_CAR	BI_H_BUT	BI_H_SPI	BI_G_TOT	BI_G_ARA	BI_G_GRA	BI_G_SMA	BI_G_BIR	BI_G_AMP	BI_G_BEE	BI_G_LOC	BI_G_CAR	BI_G_BUT	BI_G_SPI	BI_G_MOL
Mountains	ALPIMEAD	CON	MAIN	9.75	32.46	19.44	33.55	18.60	18.35		20.70	11.14	24.16	11.21	23.94	34.66	20.88	33.55	20.86	9.22
Mountains	BARLEY__	CON	INT	1.36	7.72	10.78	7.88	9.55	6.86	13.22	2.71	3.78	7.37	2.01	3.62	7.88	11.25	7.88	9.55	2.31
Mountains	BARLEY__	CON	EXS	1.47	7.72	11.04	7.88	10.62	7.05	13.27	2.71	3.79	8.05	2.01	3.63	7.88	11.50	7.88	10.62	2.29
Mountains	EXTEPAST	CON	MAIN	5.69	18.63	10.24	19.38	10.71	11.42		12.69	10.72	16.72	7.61	15.81	20.69	11.24	19.55	12.51	5.26
Mountains	FIELBEAN	CON	MAIN	1.49	7.72	11.19	7.88	10.96	6.80	11.29		4.62	6.67	2.08	3.62	7.88	11.66	7.88	10.96	2.31
Mountains	FIELDPEA	CON	MAIN	1.40	7.72	10.30	7.88	10.16	6.54	11.00		4.62	6.18	2.01	3.34	7.88	10.89	7.88	10.16	2.29
Mountains	FODMAIZE	CON	MAIN	1.44	7.72	10.25	7.88	10.71	6.22	8.45		3.79	6.72	2.05	2.97	7.88	10.65	7.88	10.83	2.28
Mountains	FODROOTS	CON	MAIN	1.38	7.72	9.57	7.88	9.98	6.01	8.31		3.78	6.16	2.01	2.80	7.88	10.10	7.88	10.10	2.29
Mountains	GRASARAB	CON	MAIN	1.60	4.19	7.75	3.66	7.16	4.23		4.29	2.86	8.51	1.82	4.66	4.42	7.75	3.76	7.37	2.19
Mountains	GRASMEAD	CON	INT	1.67	5.38	7.09	5.34	7.18	4.98		3.65	7.36	8.09	1.77	6.24	5.67	7.28	5.52	7.41	5.14
Mountains	GRASMEAD	CON	EXT	9.75	31.50	22.44	31.75	20.70	17.96		19.44	11.23	21.39	8.83	22.14	32.50	23.33	31.75	22.30	11.64
Mountains	GRASMEAD	CON	LIN	4.60	17.53	14.04	15.81	13.38	11.11		12.03	11.07	12.99	4.58	16.63	18.01	14.46	16.77	14.19	5.82
Mountains	MAIZE___	CON	MAIN	1.36	7.72	9.77	7.88	10.33	6.10	8.44		3.79	6.41	2.05	2.84	7.88	10.27	7.88	10.44	2.30
Mountains	MIXFALLO	CON	MAIN	6.00	10.13	20.25	20.00	26.38	16.39	21.75		11.13	35.00	7.00	20.38	10.13	20.25	20.00	26.38	3.00
Mountains	OATS____	CON	INT	1.36	7.72	10.78	7.88	9.55	6.86	13.22	2.71	3.78	7.37	2.01	3.62	7.88	11.25	7.88	9.55	2.31
Mountains	OATS____	CON	EXS	1.47	7.72	11.04	7.88	10.62	7.05	13.27	2.71	3.79	8.05	2.01	3.63	7.88	11.50	7.88	10.62	2.29
Mountains	OTHBRCER	CON	MAIN	1.42		10.43		8.07	7.60	14.94		4.57	5.08	1.71	4.83		10.95		8.26	2.21

Region	Activity	Farming system	Policy uptake	BI_H_AMP	BI_H_LOC	BI_H_CAR	BI_H_BUT	BI_H_SPI	BI_G_TOT	BI_G_ARA	BI_G_GRA	BI_G_SMA	BI_G_BIR	BI_G_AMP	BI_G_BEE	BI_G_LOC	BI_G_CAR	BI_G_BUT	BI_G_SPI	BI_G_MOL
Mountains	OTHBRCER	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Mountains	OTHEPAST	CON	MAIN	5.69	18.63	10.24	19.38	10.71	11.42		12.69	10.72	16.72	7.61	15.81	20.69	11.24	19.55	12.51	5.26
Mountains	OTHFOCER	CON	INT	1.44		10.33		7.95	7.66	15.36		4.57	5.11	1.79	4.98		10.83		8.14	2.16
Mountains	OTHFOCER	CON	EXS	1.60		11.17		10.29	8.38	16.06		4.57	6.19	1.79	4.98		11.67		10.48	2.16
Mountains	POTATOES	CON	MAIN	1.27	7.72	10.41	7.88	8.87	6.09	8.42		4.62	5.84	1.99	3.33	7.88	11.06	7.88	8.99	2.31
Mountains	PULSES__	CON	MAIN	1.49	7.72	11.19	7.88	10.96	6.80	11.29		4.62	6.67	2.08	3.62	7.88	11.66	7.88	10.96	2.31
Mountains	RAPE____	CON	INT	1.48		7.87		9.05	7.20	14.51		4.58	5.54	1.54	4.13		8.39		9.05	2.17
Mountains	RAPE____	CON	EXS	1.67		9.68		10.40	7.97	14.85		4.58	6.76	1.62	5.19		9.93		10.40	2.17
Mountains	ROTFALLO	CON	MAIN	6.00	10.13	18.00	16.63	18.63	12.96	17.25		11.13	28.00	7.00	10.88	10.13	18.00	16.63	18.63	3.00
Mountains	RYE_____	CON	INT	1.44		10.33		7.95	7.66	15.36		4.57	5.11	1.79	4.98		10.83		8.14	2.16
Mountains	RYE_____	CON	EXS	1.60		11.17		10.29	8.38	16.06		4.57	6.19	1.79	4.98		11.67		10.48	2.16
Mountains	SPELT___	CON	INT	1.42		10.43		8.07	7.60	14.94		4.57	5.08	1.71	4.83		10.95		8.26	2.21
Mountains	SPELT___	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Mountains	SUGABEET	CON	MAIN	1.38	7.72	9.97	7.88	9.44	6.00	8.31		3.79	6.24	2.01	2.92	7.88	10.40	7.88	9.56	2.29
Mountains	SUNFLOWE	CON	MAIN	1.52	7.72	11.19	7.88	10.81	7.04	12.58		4.62	6.83	2.04	4.45	7.88	11.56	7.88	10.94	2.31
Mountains	TRITICAL	CON	INT	1.43		10.38		8.01	7.63	15.15		4.57	5.09	1.75	4.91		10.89		8.20	2.18
Mountains	TRITICAL	CON	EXS	1.60		11.17		10.29	8.37	16.04		4.57	6.18	1.79	4.98		11.67		10.48	2.16
Mountains	VEGETABL	CON	MAIN	1.29	7.72	10.85	7.88	9.18	6.17	8.34	2.71	4.60	6.16	1.95	3.28	7.88	11.51	7.88	9.31	2.30
Mountains	WHEAT___	CON	INT	1.42		10.43		8.07	7.60	14.94		4.57	5.08	1.71	4.83		10.95		8.26	2.21
Mountains	WHEAT___	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Hills	ALPIMEAD	CON	MAIN	9.75	32.46	19.44	33.55	18.60	18.35		20.70	11.14	24.16	11.21	23.94	34.66	20.88	33.55	20.86	9.22
Hills	BARLEY__	CON	INT	1.36	7.72	10.78	7.88	9.55	6.85	13.22	2.71	3.78	7.25	2.01	3.62	7.88	11.25	7.88	9.55	2.31
Hills	BARLEY__	CON	EXS	1.47	7.72	11.04	7.88	10.62	7.04	13.27	2.71	3.79	7.93	2.01	3.63	7.88	11.50	7.88	10.62	2.29
Hills	EXTEPAST	CON	MAIN	3.57	10.91	7.02	11.04	7.69	7.61		7.85	10.55	11.40	4.49	10.63	12.28	7.49	11.31	8.90	4.45
Hills	FIELBEAN	CON	MAIN	1.49	7.72	11.19	7.88	10.96	6.80	11.29		4.62	6.67	2.08	3.62	7.88	11.66	7.88	10.96	2.31
Hills	FIELDPEA	CON	MAIN	1.40	7.72	10.30	7.88	10.16	6.54	11.00		4.62	6.18	2.01	3.34	7.88	10.89	7.88	10.16	2.29
Hills	FODMAIZE	CON	MAIN	1.44	7.72	10.25	7.88	10.71	6.22	8.45		3.79	6.72	2.05	2.97	7.88	10.65	7.88	10.83	2.28
Hills	FODROOTS	CON	MAIN	1.38	7.72	9.57	7.88	9.98	6.01	8.31		3.78	6.16	2.01	2.80	7.88	10.10	7.88	10.10	2.29

Region	Activity	Farming system	Policy uptake	BI_H_AMP	BI_H_LOC	BI_H_CAR	BI_H_BUT	BI_H_SPI	BI_G_TOT	BI_G_ARA	BI_G_GRA	BI_G_SMA	BI_G_BIR	BI_G_AMP	BI_G_BEE	BI_G_LOC	BI_G_CAR	BI_G_BUT	BI_G_SPI	BI_G_MOL
Hills	GRASARAB	CON	MAIN	1.60	4.19	7.75	3.66	7.16	4.23		4.29	2.86	8.51	1.82	4.66	4.42	7.75	3.76	7.37	2.19
Hills	GRASMEAD	CON	INT	1.67	5.41	7.09	5.31	7.18	4.96		3.68	7.36	8.04	1.77	6.23	5.68	7.09	5.49	7.41	5.14
Hills	GRASMEAD	CON	EXT	9.75	31.50	22.44	31.75	20.70	17.96		19.44	11.23	21.39	8.83	22.14	32.50	23.33	31.75	22.30	11.64
Hills	GRASMEAD	CON	LIN	4.60	17.53	14.04	15.81	13.38	11.11		12.03	11.07	12.99	4.58	16.63	18.01	14.46	16.77	14.19	5.82
Hills	MAIZE___	CON	MAIN	1.36	7.72	9.77	7.88	10.33	6.10	8.44		3.79	6.41	2.05	2.84	7.88	10.27	7.88	10.44	2.30
Hills	MIXFALLO	CON	MAIN	6.00	10.13	20.25	20.00	26.38	16.39	21.75		11.13	35.00	7.00	20.38	10.13	20.25	20.00	26.38	3.00
Hills	OATS____	CON	INT	1.36	7.72	10.78	7.88	9.55	6.85	13.22	2.71	3.78	7.25	2.01	3.62	7.88	11.25	7.88	9.55	2.31
Hills	OATS____	CON	EXS	1.47	7.72	11.04	7.88	10.62	7.04	13.27	2.71	3.79	7.93	2.01	3.63	7.88	11.50	7.88	10.62	2.29
Hills	OTHBRCER	CON	MAIN	1.42		10.43		8.07	7.59	14.94		4.57	5.03	1.71	4.83		10.95		8.26	2.21
Hills	OTHBRCER	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Hills	OTHEPAST	CON	MAIN	3.57	10.91	7.02	11.04	7.69	7.61		7.85	10.55	11.40	4.49	10.63	12.28	7.49	11.31	8.90	4.45
Hills	OTHFOCER	CON	INT	1.44		10.33		7.95	7.66	15.36		4.57	5.11	1.79	4.98		10.83		8.14	2.16
Hills	OTHFOCER	CON	EXS	1.60		11.17		10.29	8.38	16.06		4.57	6.19	1.79	4.98		11.67		10.48	2.16
Hills	POTATOES	CON	MAIN	1.27	7.72	10.41	7.88	8.87	6.09	8.42		4.62	5.84	1.99	3.33	7.88	11.06	7.88	8.99	2.31
Hills	PULSES__	CON	MAIN	1.49	7.72	11.19	7.88	10.96	6.80	11.29		4.62	6.67	2.08	3.62	7.88	11.66	7.88	10.96	2.31
Hills	RAPE____	CON	INT	1.48		7.87		9.05	7.20	14.51		4.58	5.54	1.54	4.13		8.39		9.05	2.17
Hills	RAPE____	CON	EXS	1.67		9.68		10.40	7.97	14.85		4.58	6.76	1.62	5.19		9.93		10.40	2.17
Hills	ROTFALLO	CON	MAIN	6.00	10.13	18.00	16.63	18.63	12.96	17.25		11.13	28.00	7.00	10.88	10.13	18.00	16.63	18.63	3.00
Hills	RYE_____	CON	INT	1.44		10.33		7.95	7.66	15.36		4.57	5.11	1.79	4.98		10.83		8.14	2.16
Hills	RYE_____	CON	EXS	1.60		11.17		10.29	8.38	16.06		4.57	6.19	1.79	4.98		11.67		10.48	2.16
Hills	SPELT___	CON	INT	1.42		10.43		8.07	7.59	14.94		4.57	5.03	1.71	4.83		10.95		8.26	2.21
Hills	SPELT___	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Hills	SUGABEET	CON	MAIN	1.38	7.72	9.97	7.88	9.44	6.00	8.31		3.79	6.24	2.01	2.92	7.88	10.40	7.88	9.56	2.29
Hills	SUNFLOWE	CON	MAIN	1.52	7.72	11.19	7.88	10.81	7.04	12.58		4.62	6.83	2.04	4.45	7.88	11.56	7.88	10.94	2.31
Hills	TRITICAL	CON	INT	1.43		10.38		8.01	7.63	15.15		4.57	5.07	1.75	4.91		10.89		8.20	2.18
Hills	TRITICAL	CON	EXS	1.60		11.17		10.29	8.37	16.04		4.57	6.18	1.79	4.98		11.67		10.48	2.16
Hills	VEGETABL	CON	MAIN	1.29	7.72	10.85	7.88	9.18	6.17	8.34	2.71	4.60	6.16	1.95	3.28	7.88	11.51	7.88	9.31	2.30
Hills	WHEAT___	CON	INT	1.42		10.43		8.07	7.59	14.94		4.57	5.03	1.71	4.83		10.95		8.26	2.21

Region	Activity	Farming system	Policy uptake	BI_H_AMP	BI_H_LOC	BI_H_CAR	BI_H_BUT	BI_H_SPI	BI_G_TOT	BI_G_ARA	BI_G_GRA	BI_G_SMA	BI_G_BIR	BI_G_AMP	BI_G_BEE	BI_G_LOC	BI_G_CAR	BI_G_BUT	BI_G_SPI	BI_G_MOL
Hills	WHEAT___	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Lowlands	ALPIMEAD	CON	MAIN	9.75	32.46	19.44	33.55	18.60	18.35		20.70	11.14	24.16	11.21	23.94	34.66	20.88	33.55	20.86	9.22
Lowlands	BARLEY__	CON	INT	1.36	7.72	10.78	7.88	9.55	6.85	13.22	2.71	3.78	7.25	2.01	3.62	7.88	11.25	7.88	9.55	2.31
Lowlands	BARLEY__	CON	EXS	1.47	7.72	11.04	7.88	10.62	7.03	13.20	2.71	3.79	7.86	2.01	3.63	7.88	11.50	7.88	10.62	2.29
Lowlands	EXTEPAST	CON	MAIN	3.28	9.84	6.57	9.89	7.27	7.08		7.18	10.52	10.66	4.06	9.91	11.12	6.97	10.17	8.39	4.34
Lowlands	FIELBEAN	CON	MAIN	1.49	7.72	11.19	7.88	10.96	6.80	11.29		4.62	6.67	2.08	3.62	7.88	11.66	7.88	10.96	2.31
Lowlands	FIELDPEA	CON	MAIN	1.40	7.72	10.30	7.88	10.16	6.54	11.00		4.62	6.18	2.01	3.34	7.88	10.89	7.88	10.16	2.29
Lowlands	FODMAIZE	CON	MAIN	1.44	7.72	10.25	7.88	10.71	6.22	8.45		3.79	6.72	2.05	2.97	7.88	10.65	7.88	10.83	2.28
Lowlands	FODROOTS	CON	MAIN	1.38	7.72	9.57	7.88	9.98		8.31		3.78	6.16	2.01	2.80	7.88	10.10	7.88	10.10	2.29
Lowlands	GRASARAB	CON	MAIN	1.60	4.19	7.75	3.66	7.16	4.23		4.29	2.86	8.51	1.82	4.66	4.42	7.75	3.76	7.37	2.19
Lowlands	GRASMEAD	CON	INT	1.67	5.36	6.94	5.21	7.00	4.88		3.66	7.36	7.78	1.77	6.22	5.66	6.94	5.39	7.24	5.00
Lowlands	GRASMEAD	CON	EXT	9.75	31.50	22.44	31.75	20.70	17.96		19.44	11.23	21.39	8.83	22.14	32.50	23.33	31.75	22.30	11.64
Lowlands	GRASMEAD	CON	LIN	4.60	17.53	14.04	15.81	13.38	11.11		12.03	11.07	12.99	4.58	16.63	18.01	14.46	16.77	14.19	5.82
Lowlands	MAIZE___	CON	MAIN	1.36	7.72	9.77	7.88	10.28	6.10	8.45		3.79	6.43	2.05	2.84	7.88	10.27	7.88	10.40	2.28
Lowlands	MIXFALLO	CON	MAIN	6.00	10.13	20.25	20.00	26.38	16.39	21.75		11.13	35.00	7.00	20.38	10.13	20.25	20.00	26.38	3.00
Lowlands	OATS____	CON	INT	1.36	7.72	10.78	7.88	9.55	6.85	13.22	2.71	3.78	7.25	2.01	3.62	7.88	11.25	7.88	9.55	2.31
Lowlands	OATS____	CON	EXS	1.47	7.72	11.04	7.88	10.62	7.03	13.20	2.71	3.79	7.86	2.01	3.63	7.88	11.50	7.88	10.62	2.29
Lowlands	OTHBRCER	CON	MAIN	1.44		10.12		7.77	7.51	15.07		4.57	4.96	1.71	4.88		10.62		7.95	2.16
Lowlands	OTHBRCER	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Lowlands	OTHEPAST	CON	MAIN	3.28	9.84	6.57	9.89	7.27	7.08		7.18	10.52	10.66	4.06	9.91	11.12	6.97	10.17	8.39	4.34
Lowlands	OTHFOCER	CON	INT	1.44		10.12		7.77	7.50	14.93		4.57	5.11	1.71	4.88		10.62		7.95	2.16
Lowlands	OTHFOCER	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Lowlands	POTATOES	CON	MAIN	1.27	7.72	10.41	7.88	8.87	6.08	8.41		4.62	5.73	1.99	3.33	7.88	11.06	7.88	8.99	2.31
Lowlands	PULSES__	CON	MAIN	1.49	7.72	11.19	7.88	10.96	6.80	11.29		4.62	6.67	2.08	3.62	7.88	11.66	7.88	10.96	2.31
Lowlands	RAPE____	CON	INT	1.48		7.87		9.05	7.20	14.51		4.58	5.54	1.54	4.13		8.39		9.05	2.17
Lowlands	RAPE____	CON	EXS	1.67		9.68		10.40	7.97	14.85		4.58	6.76	1.62	5.19		9.93		10.40	2.17
Lowlands	ROTFALLO	CON	MAIN	6.00	10.13	18.00	16.63	18.63	12.96	17.25		11.13	28.00	7.00	10.88	10.13	18.00	16.63	18.63	3.00
Lowlands	RYE_____	CON	INT	1.44		10.12		7.77	7.50	14.93		4.57	5.11	1.71	4.88		10.62		7.95	2.16

Region	Activity	Farming system	Policy uptake	BI_H_AMP	BI_H_LOC	BI_H_CAR	BI_H_BUT	BI_H_SPI	BI_G_TOT	BI_G_ARA	BI_G_GRA	BI_G_SMA	BI_G_BIR	BI_G_AMP	BI_G_BEE	BI_G_LOC	BI_G_CAR	BI_G_BUT	BI_G_SPI	BI_G_MOL
Lowlands	RYE_____	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Lowlands	SPELT___	CON	INT	1.44		10.12		7.77	7.51	15.07		4.57	4.96	1.71	4.88		10.62		7.95	2.16
Lowlands	SPELT___	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Lowlands	SUGABEET	CON	MAIN	1.38	7.72	9.97	7.88	9.44	6.00	8.31		3.79	6.24	2.01	2.92	7.88	10.40	7.88	9.56	2.29
Lowlands	SUNFLOWE	CON	MAIN	1.52	7.72	11.19	7.88	10.81	7.04	12.58		4.62	6.83	2.04	4.45	7.88	11.56	7.88	10.94	2.31
Lowlands	TRITICAL	CON	INT	1.44		10.12		7.77	7.51	15.00		4.57	5.03	1.71	4.88		10.62		7.95	2.16
Lowlands	TRITICAL	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Lowlands	VEGETABL	CON	MAIN	1.29	7.72	10.85	7.88	9.18	6.17	8.34	2.71	4.60	6.16	1.95	3.28	7.88	11.51	7.88	9.31	2.30
Lowlands	WHEAT___	CON	INT	1.44		10.12		7.77	7.51	15.07		4.57	4.96	1.71	4.88		10.62		7.95	2.16
Lowlands	WHEAT___	CON	EXS	1.60		11.17		10.29	8.37	16.01		4.57	6.16	1.79	4.98		11.67		10.48	2.16
Mountains	ALPIMEAD	ORG	MAIN	9.75	32.46	19.44	33.55	18.60	18.35		20.70	11.14	24.16	11.21	23.94	34.66	20.88	33.55	20.86	9.22
Mountains	BARLEY__	ORG	EXS	1.74	9.98	11.44	9.41	11.10	7.84	14.47	3.17	3.77	10.31	2.32	3.62	9.98	11.92	9.41	11.10	2.39
Mountains	EXTEPAST	ORG	MAIN	7.30	24.49	12.83	25.93	13.03	14.42		16.46	10.85	21.12	9.97	19.82	27.11	14.27	26.00	15.42	5.98
Mountains	FIELBEAN	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Mountains	FIELDPEA	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Mountains	FODMAIZE	ORG	MAIN	1.71	9.98	10.48	9.41	10.99	6.90	9.18		3.78	8.79	2.33	2.94	9.98	10.90	9.41	11.12	2.39
Mountains	FODROOTS	ORG	MAIN	1.38	7.72	9.57	7.88	9.98	6.01	8.31		3.78	6.16	2.01	2.80	7.88	10.10	7.88	10.10	2.29
Mountains	GRASARAB	ORG	MAIN	1.33	4.62	8.23	4.08	7.25	4.43		4.18	2.75	9.72	1.67	4.72	4.79	8.23	4.08	7.54	2.25
Mountains	GRASMEAD	ORG	INT	1.80	5.83	7.92	6.48	7.85	5.48		4.19	7.32	9.31	1.75	6.38	6.05	8.17	6.48	8.42	5.46
Mountains	GRASMEAD	ORG	EXT	9.75	31.50	22.44	31.75	20.70	17.96		19.44	11.23	21.39	8.83	22.14	32.50	23.33	31.75	22.30	11.64
Mountains	GRASMEAD	ORG	LIN	4.60	17.53	14.04	15.81	13.38	11.11		12.03	11.07	12.99	4.58	16.63	18.01	14.46	16.77	14.19	5.82
Mountains	MAIZE___	ORG	MAIN	1.63	9.98	9.95	9.41	10.52	6.77	9.18		3.78	8.49	2.33	2.79	9.98	10.48	9.41	10.64	2.39
Mountains	MIXFALLO	ORG	MAIN	6.00	10.13	20.25	20.00	26.38	17.17	26.75		11.13	35.00	7.00	20.38	10.13	20.25	20.00	26.38	3.00
Mountains	OATS____	ORG	EXS	1.74	9.98	11.44	9.41	11.10	7.84	14.47	3.17	3.77	10.31	2.32	3.62	9.98	11.92	9.41	11.10	2.39
Mountains	OTHBRCER	ORG	MAIN	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Mountains	OTHEPAST	ORG	MAIN	7.30	24.49	12.83	25.93	13.03	14.42		16.46	10.85	21.12	9.97	19.82	27.11	14.27	26.00	15.42	5.98
Mountains	OTHFOCER	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Mountains	POTATOES	ORG	MAIN	1.63	9.98	11.85	9.41	9.88	7.03	9.18		4.61	8.53	2.30	3.61	9.98	12.37	9.41	10.00	2.41

Region	Activity	Farming system	Policy uptake	BI_H_AMP	BI_H_LOC	BI_H_CAR	BI_H_BUT	BI_H_SPI	BI_G_TOT	BI_G_ARA	BI_G_GRA	BI_G_SMA	BI_G_BIR	BI_G_AMP	BI_G_BEE	BI_G_LOC	BI_G_CAR	BI_G_BUT	BI_G_SPI	BI_G_MOL
Mountains	PULSES__	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Mountains	RAPE____	ORG	EXS	1.65		9.74		10.87	8.31	16.22		4.58	6.75	1.86	4.93		10.18		10.87	2.17
Mountains	ROTFALLO	ORG	MAIN	6.00	10.13	18.00	16.63	18.63	13.74	22.25		11.13	28.00	7.00	10.88	10.13	18.00	16.63	18.63	3.00
Mountains	RYE_____	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Mountains	SPELT___	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Mountains	SUGABEET	ORG	MAIN	1.38	7.72	9.97	7.88	9.44	6.00	8.31		3.79	6.24	2.01	2.92	7.88	10.40	7.88	9.56	2.29
Mountains	SUNFLOWE	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.56	12.36		4.61	9.02	2.39	3.62	9.98	11.92	9.41	11.33	2.39
Mountains	TRITICAL	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Mountains	VEGETABL	ORG	MAIN	1.74	9.98	12.42	9.41	11.31	7.29	9.29	3.17	4.61	9.08	2.30	3.60	9.98	12.95	9.41	11.44	2.40
Mountains	WHEAT___	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Hills	ALPIMEAD	ORG	MAIN	9.75	32.46	19.44	33.55	18.60	18.35		20.70	11.14	24.16	11.21	23.94	34.66	20.88	33.55	20.86	9.22
Hills	BARLEY__	ORG	EXS	1.74	9.98	11.44	9.41	11.10	7.84	14.47	3.17	3.77	10.20	2.32	3.62	9.98	11.92	9.41	11.10	2.39
Hills	EXTEPAST	ORG	MAIN	4.75	15.19	9.08	16.13	9.42	9.95		10.74	10.63	15.08	6.22	13.65	17.01	9.95	16.27	11.26	5.11
Hills	FIELBEAN	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Hills	FIELDPEA	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Hills	FODMAIZE	ORG	MAIN	1.71	9.98	10.48	9.41	10.99	6.90	9.18		3.78	8.79	2.33	2.94	9.98	10.90	9.41	11.12	2.39
Hills	FODROOTS	ORG	MAIN	1.38	7.72	9.57	7.88	9.98	6.01	8.31		3.78	6.16	2.01	2.80	7.88	10.10	7.88	10.10	2.29
Hills	GRASARAB	ORG	MAIN	1.33	4.62	8.23	4.08	7.25	4.43		4.18	2.75	9.72	1.67	4.72	4.79	8.23	4.08	7.54	2.25
Hills	GRASMEAD	ORG	INT	1.60	5.71	7.92	6.29	7.62	5.34		3.92	7.32	9.00	1.67	6.37	5.91	7.92	6.29	8.19	5.46
Hills	GRASMEAD	ORG	EXT	9.75	31.50	22.44	31.75	20.70	17.96		19.44	11.23	21.39	8.83	22.14	32.50	23.33	31.75	22.30	11.64
Hills	GRASMEAD	ORG	LIN	4.60	17.53	14.04	15.81	13.38	11.11		12.03	11.07	12.99	4.58	16.63	18.01	14.46	16.77	14.19	5.82
Hills	MAIZE___	ORG	MAIN	1.63	9.98	9.95	9.41	10.52	6.77	9.18		3.78	8.49	2.33	2.79	9.98	10.48	9.41	10.64	2.39
Hills	MIXFALLO	ORG	MAIN	6.00	10.13	20.25	20.00	26.38	17.17	26.75		11.13	35.00	7.00	20.38	10.13	20.25	20.00	26.38	3.00
Hills	OATS____	ORG	EXS	1.74	9.98	11.44	9.41	11.10	7.84	14.47	3.17	3.77	10.20	2.32	3.62	9.98	11.92	9.41	11.10	2.39
Hills	OTHBRCER	ORG	MAIN	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Hills	OTHEPAST	ORG	MAIN	4.75	15.19	9.08	16.13	9.42	9.95		10.74	10.63	15.08	6.22	13.65	17.01	9.95	16.27	11.26	5.11
Hills	OTHFOCER	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Hills	POTATOES	ORG	MAIN	1.63	9.98	11.85	9.41	9.88	7.03	9.18		4.61	8.53	2.30	3.61	9.98	12.37	9.41	10.00	2.41

Region	Activity	Farming system	Policy uptake	BI_H_AMP	BI_H_LOC	BI_H_CAR	BI_H_BUT	BI_H_SPI	BI_G_TOT	BI_G_ARA	BI_G_GRA	BI_G_SMA	BI_G_BIR	BI_G_AMP	BI_G_BEE	BI_G_LOC	BI_G_CAR	BI_G_BUT	BI_G_SPI	BI_G_MOL
Hills	PULSES__	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Hills	RAPE____	ORG	EXS	1.65		9.74		10.87	8.31	16.22		4.58	6.75	1.86	4.93		10.18		10.87	2.17
Hills	ROTFALLO	ORG	MAIN	6.00	10.13	18.00	16.63	18.63	13.74	22.25		11.13	28.00	7.00	10.88	10.13	18.00	16.63	18.63	3.00
Hills	RYE_____	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Hills	SPELT___	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Hills	SUGABEET	ORG	MAIN	1.38	7.72	9.97	7.88	9.44	6.00	8.31		3.79	6.24	2.01	2.92	7.88	10.40	7.88	9.56	2.29
Hills	SUNFLOWE	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.56	12.36		4.61	9.02	2.39	3.62	9.98	11.92	9.41	11.33	2.39
Hills	TRITICAL	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Hills	VEGETABL	ORG	MAIN	1.74	9.98	12.42	9.41	11.31	7.29	9.29	3.17	4.61	9.08	2.30	3.60	9.98	12.95	9.41	11.44	2.40
Hills	WHEAT___	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Lowlands	ALPIMEAD	ORG	MAIN	9.75	32.46	19.44	33.55	18.60	18.35		20.70	11.14	24.16	11.21	23.94	34.66	20.88	33.55	20.86	9.22
Lowlands	BARLEY__	ORG	EXS	1.74	9.98	11.44	9.41	11.10	7.84	14.47	3.17	3.77	10.20	2.32	3.62	9.98	11.92	9.41	11.10	2.39
Lowlands	EXTEPAST	ORG	MAIN	4.25	13.39	8.36	14.23	8.72	9.08		9.63	10.59	13.91	5.49	12.45	15.06	9.11	14.39	10.45	4.94
Lowlands	FIELBEAN	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Lowlands	FIELDPEA	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Lowlands	FODMAIZE	ORG	MAIN	1.71	9.98	10.48	9.41	10.99	6.90	9.18		3.78	8.79	2.33	2.94	9.98	10.90	9.41	11.12	2.39
Lowlands	FODROOTS	ORG	MAIN	1.38	7.72	9.57	7.88	9.98	6.01	8.31		3.78	6.16	2.01	2.80	7.88	10.10	7.88	10.10	2.29
Lowlands	GRASARAB	ORG	MAIN	1.33	4.62	8.23	4.08	7.25	4.43		4.18	2.75	9.72	1.67	4.72	4.79	8.23	4.08	7.54	2.25
Lowlands	GRASMEAD	ORG	INT	1.60	5.65	7.92	6.31	7.62	5.33		3.90	7.32	8.97	1.67	6.36	5.88	7.92	6.31	8.19	5.46
Lowlands	GRASMEAD	ORG	EXT	9.75	31.50	22.44	31.75	20.70	17.96		19.44	11.23	21.39	8.83	22.14	32.50	23.33	31.75	22.30	11.64
Lowlands	GRASMEAD	ORG	LIN	4.60	17.53	14.04	15.81	13.38	11.11		12.03	11.07	12.99	4.58	16.63	18.01	14.46	16.77	14.19	5.82
Lowlands	MAIZE___	ORG	MAIN	1.63	9.98	9.95	9.41	10.52	6.77	9.18		3.78	8.49	2.33	2.79	9.98	10.48	9.41	10.64	2.39
Lowlands	MIXFALLO	ORG	MAIN	6.00	10.13	20.25	20.00	26.38	17.17	26.75		11.13	35.00	7.00	20.38	10.13	20.25	20.00	26.38	3.00
Lowlands	OATS____	ORG	EXS	1.74	9.98	11.44	9.41	11.10	7.84	14.47	3.17	3.77	10.20	2.32	3.62	9.98	11.92	9.41	11.10	2.39
Lowlands	OTHBRCER	ORG	MAIN	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Lowlands	OTHEPAST	ORG	MAIN	4.25	13.39	8.36	14.23	8.72	9.08		9.63	10.59	13.91	5.49	12.45	15.06	9.11	14.39	10.45	4.94
Lowlands	OTHFOCER	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Lowlands	POTATOES	ORG	MAIN	1.63	9.98	11.65	9.41	9.79	6.99	9.18		4.61	8.40	2.30	3.61	9.98	12.17	9.41	9.92	2.41

Region	Activity	Farming system	Policy uptake	BI_H_AMP	BI_H_LOC	BI_H_CAR	BI_H_BUT	BI_H_SPI	BI_G_TOT	BI_G_ARA	BI_G_GRA	BI_G_SMA	BI_G_BIR	BI_G_AMP	BI_G_BEE	BI_G_LOC	BI_G_CAR	BI_G_BUT	BI_G_SPI	BI_G_MOL
Lowlands	PULSES__	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.52	12.17		4.61	9.02	2.36	3.62	9.98	11.92	9.41	11.19	2.39
Lowlands	RAPE____	ORG	EXS	1.65		9.74		10.87	8.31	16.22		4.58	6.75	1.86	4.93		10.18		10.87	2.17
Lowlands	ROTFALLO	ORG	MAIN	6.00	10.13	18.00	16.63	18.63	13.74	22.25		11.13	28.00	7.00	10.88	10.13	18.00	16.63	18.63	3.00
Lowlands	RYE_____	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Lowlands	SPELT___	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Lowlands	SUGABEET	ORG	MAIN	1.38	7.72	9.97	7.88	9.44	6.00	8.31		3.79	6.24	2.01	2.92	7.88	10.40	7.88	9.56	2.29
Lowlands	SUNFLOWE	ORG	MAIN	1.76	9.98	11.44	9.41	11.19	7.56	12.36		4.61	9.02	2.39	3.62	9.98	11.92	9.41	11.33	2.39
Lowlands	TRITICAL	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17
Lowlands	VEGETABL	ORG	MAIN	1.74	9.98	12.42	9.41	11.31	7.29	9.29	3.17	4.61	9.08	2.30	3.60	9.98	12.95	9.41	11.44	2.40
Lowlands	WHEAT___	ORG	EXS	1.61		11.32		10.47	8.67	17.26		4.58	6.43	1.79	4.83		11.87		10.68	2.17

Source: own compilation of data based on SALCA BD model results and Nemecek *et al.* (2005)

Table 78 Assumptions for N and P eutrophication per ha

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Lowlands	RYE_____	ORG	EXS	85.37	25.29	4.45	2.13	3.98
Lowlands	RYE_____	CON	INT	76.45	8.21	4.95	2.94	8.14
Lowlands	RYE_____	CON	EXS	78.74	7.31	4.83	2.80	7.29
Lowlands	OATS____	ORG	EXS	77.72	25.29	4.45	2.13	3.98
Lowlands	OATS____	CON	INT	82.71	8.21	4.95	2.94	8.14
Lowlands	OATS____	CON	EXS	91.08	7.31	4.83	2.80	7.29
Lowlands	MAIZE___	ORG	MAIN	49.55	37.87	6.47	2.63	4.13
Lowlands	MAIZE___	CON	MAIN	68.17	17.79	6.04	2.99	3.85
Lowlands	TRITICAL	ORG	EXS	89.92	29.31	4.74	2.18	4.04
Lowlands	TRITICAL	CON	INT	76.76	8.31	5.48	3.15	8.21
Lowlands	TRITICAL	CON	EXS	84.37	8.57	5.36	2.96	7.09
Lowlands	PULSES__	ORG	MAIN	28.06	15.49	6.17	2.17	3.88
Lowlands	PULSES__	CON	MAIN	14.12	1.98	5.15	2.44	5.53
Lowlands	RAPE____	ORG	EXS	14.62	32.38	4.86	1.72	3.44
Lowlands	RAPE____	CON	INT	29.26	14.85	5.77	2.64	5.26
Lowlands	RAPE____	CON	EXS	43.78	20.51	4.83	2.33	2.48
Lowlands	SUNFLOWE	ORG	MAIN	44.07	2.17	2.70	1.99	4.34
Lowlands	SUNFLOWE	CON	MAIN	44.07	2.17	2.70	1.99	4.34
Lowlands	OTHOILS_	ORG	MAIN	48.56	5.99	7.34	2.39	4.00
Lowlands	OTHOILS_	CON	MAIN	47.53	5.07	5.15	1.89	2.43
Lowlands	POTATOES	ORG	MAIN	92.51	20.63	4.60	3.09	2.64
Lowlands	POTATOES	CON	MAIN	98.51	22.61	6.17	3.91	3.42
Lowlands	SUGABEET	ORG	MAIN	9.83	16.61	4.85	3.23	2.59
Lowlands	SUGABEET	CON	MAIN	9.83	16.61	4.85	3.23	2.59
Lowlands	VEGETABL	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Lowlands	VEGETABL	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Lowlands	FRUITS__	ORG	MAIN	0.00	31.48	2.52	1.15	4.14
Lowlands	FRUITS__	CON	MAIN	0.00	45.82	3.32	1.37	5.13
Lowlands	VINEYARD	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Lowlands	VINEYARD	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Lowlands	TOBACCO_	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Lowlands	TOBACCO_	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Lowlands	GRASARAB	ORG	MAIN	60.01	61.94	4.45	2.40	5.98
Lowlands	GRASARAB	CON	MAIN	77.22	66.21	5.92	2.93	8.60
Lowlands	FODMAIZE	ORG	MAIN	14.12	38.74	4.45	2.88	3.88
Lowlands	FODMAIZE	CON	MAIN	24.38	23.67	4.81	3.35	9.08
Lowlands	FODROOTS	ORG	MAIN	10.16	19.13	3.95	3.06	3.13
Lowlands	FODROOTS	CON	MAIN	10.16	19.13	3.95	3.06	3.13
Lowlands	DAIRYCOW	ORG	MAIN		9.13			
Lowlands	DAIRYCOW	CON	MAIN		9.13			
Lowlands	SUCKLCOW	ORG	MAIN		1.57			
Lowlands	SUCKLCOW	CON	MAIN		1.57			
Lowlands	PORK____	ORG	MAIN		0.31			
Lowlands	PORK____	CON	MAIN		0.31			

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Lowlands	SOWS____	ORG	MAIN		2.14			
Lowlands	SOWS____	CON	MAIN		2.14			
Lowlands	LAYHENS_	ORG	MAIN		0.28			
Lowlands	LAYHENS_	CON	MAIN		0.28			
Lowlands	BROILER_	ORG	MAIN		0.17			
Lowlands	BROILER_	CON	MAIN		0.17			
Lowlands	OPOULTRY	ORG	MAIN		0.63			
Lowlands	OPOULTRY	CON	MAIN		0.63			
Lowlands	OANIMALS	ORG	MAIN		0.07			
Lowlands	OANIMALS	CON	MAIN		0.07			
Lowlands	WHEAT___	ORG	EXS	147.48	32.23	5.02	2.26	4.27
Lowlands	WHEAT___	CON	INT	259.16	8.43	5.95	3.35	8.19
Lowlands	WHEAT___	CON	EXS	133.48	10.11	5.82	3.09	6.78
Lowlands	SPELT___	ORG	EXS	77.72	21.91	3.12	1.95	2.46
Lowlands	SPELT___	CON	INT	82.71	7.14	4.11	2.97	5.75
Lowlands	SPELT___	CON	EXS	91.08	9.45	3.88	2.70	4.58
Lowlands	BARLEY__	ORG	EXS	89.29	26.91	4.55	2.10	3.79
Lowlands	BARLEY__	CON	INT	98.48	9.13	5.79	3.13	7.80
Lowlands	BARLEY__	CON	EXS	107.80	7.78	5.64	2.95	6.99
Lowlands	OTHBRCER	ORG	MAIN	70.77	25.19	4.09	2.07	3.36
Lowlands	OTHBRCER	CON	MAIN	71.52	7.53	4.93	3.03	7.14
Lowlands	OTHBRCER	CON	EXS	78.11	8.45	4.91	2.83	6.00
Lowlands	OTHFOCER	ORG	EXS	70.77	25.19	4.09	2.07	3.36
Lowlands	OTHFOCER	CON	INT	71.52	7.53	4.93	3.03	7.14
Lowlands	OTHFOCER	CON	EXS	78.11	8.45	4.91	2.83	6.00
Lowlands	FIELBEAN	ORG	MAIN	26.72	15.29	6.73	2.19	3.82
Lowlands	FIELBEAN	CON	MAIN	12.97	1.12	5.23	2.42	6.13
Lowlands	FIELDPEA	ORG	MAIN	29.39	15.70	5.62	2.15	3.94
Lowlands	FIELDPEA	CON	MAIN	15.27	2.83	5.08	2.46	4.93
Lowlands	OTHACROP	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Lowlands	OTHACROP	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Lowlands	GRASMEAD	ORG	INT	0.00	47.66	2.20	1.21	4.33
Lowlands	GRASMEAD	ORG	EXT	0.00	13.90	2.39	1.34	3.07
Lowlands	GRASMEAD	ORG	LIN	0.00	31.48	2.52	1.15	4.14
Lowlands	GRASMEAD	CON	INT	1.72	52.89	3.32	1.75	7.19
Lowlands	GRASMEAD	CON	EXT	0.00	13.90	2.39	1.34	3.07
Lowlands	GRASMEAD	CON	LIN	0.00	45.82	3.32	1.37	5.13
Lowlands	OTHEPAST	ORG	MAIN	17.76	41.47	4.90	0.68	4.09
Lowlands	OTHEPAST	CON	MAIN	21.60	31.25	6.86	1.30	4.12
Lowlands	EXTEPAST	ORG	MAIN	17.76	41.47	4.90	0.68	4.09
Lowlands	EXTEPAST	CON	MAIN	21.60	31.25	6.86	1.30	4.12
Lowlands	ALPIMEAD	ORG	MAIN	7.32	6.89	2.12	0.30	7.32
Lowlands	ALPIMEAD	CON	MAIN	7.32	6.89	2.12	0.30	7.32
Lowlands	BERRIES_	ORG	MAIN	0.00	31.48	2.52	1.15	4.14
Lowlands	BERRIES_	CON	MAIN	0.00	45.82	3.32	1.37	5.13

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Lowlands	OTHPCROP	ORG	MAIN	0.00	31.48	2.52	1.15	4.14
Lowlands	OTHPCROP	CON	MAIN	0.00	45.82	3.32	1.37	5.13
Lowlands	OTHAREA_	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Lowlands	OTHAREA_	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Lowlands	DAIRBUL3	ORG	MAIN		6.30			
Lowlands	DAIRBUL3	CON	MAIN		6.30			
Lowlands	DAIRBUL2	ORG	MAIN		4.50			
Lowlands	DAIRBUL2	CON	MAIN		4.50			
Lowlands	DAIRBUL1	ORG	MAIN		3.20			
Lowlands	DAIRBUL1	CON	MAIN		3.20			
Lowlands	DAIRCALV	ORG	MAIN		1.65			
Lowlands	DAIRCALV	CON	MAIN		1.65			
Lowlands	SUCKBHEF	ORG	MAIN		6.35			
Lowlands	SUCKBHEF	CON	MAIN		6.35			
Lowlands	SUCKFHEF	ORG	MAIN		6.35			
Lowlands	SUCKFHEF	CON	MAIN		6.35			
Lowlands	SUCKBBUL	ORG	MAIN		6.35			
Lowlands	SUCKBBUL	CON	MAIN		6.35			
Lowlands	SUCKFBUL	ORG	MAIN		6.35			
Lowlands	SUCKFBUL	CON	MAIN		6.35			
Lowlands	SUCKCALV	ORG	MAIN		6.35			
Lowlands	SUCKCALV	CON	MAIN		6.35			
Lowlands	FACATTLE	ORG	MAIN		4.81			
Lowlands	FACATTLE	CON	MAIN		4.81			
Lowlands	FACALVES	ORG	MAIN		1.57			
Lowlands	FACALVES	CON	MAIN		1.57			
Lowlands	HORSES__	ORG	MAIN		7.06			
Lowlands	HORSES__	CON	MAIN		7.06			
Lowlands	SHEPMILK	ORG	MAIN		2.23			
Lowlands	SHEPMILK	CON	MAIN		2.23			
Lowlands	SHEPFATT	ORG	MAIN		2.23			
Lowlands	SHEPFATT	CON	MAIN		2.23			
Lowlands	GOATS___	ORG	MAIN		2.28			
Lowlands	GOATS___	CON	MAIN		2.28			
Lowlands	OROCLIVE	ORG	MAIN		7.06			
Lowlands	OROCLIVE	CON	MAIN		7.06			
Lowlands	DAIRYHEF3	ORG	MAIN		6.30			
Lowlands	DAIRYHEF3	CON	MAIN		6.30			
Lowlands	DAIRYHEF2	ORG	MAIN		4.50			
Lowlands	DAIRYHEF2	CON	MAIN		4.50			
Lowlands	DAIRYHEF1	ORG	MAIN		3.20			
Lowlands	DAIRYHEF1	CON	MAIN		3.20			
Lowlands	FCATCALV	ORG	MAIN		2.67			
Lowlands	FCATCALV	CON	MAIN		2.67			
Hills	RYE_____	ORG	EXS	85.94	26.72	4.51	2.15	6.07

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Hills	RYE_____	CON	INT	85.20	13.06	5.83	2.90	8.86
Hills	RYE_____	CON	EXS	87.17	16.45	5.33	2.73	6.82
Hills	OATS____	ORG	EXS	72.42	26.72	4.51	2.15	6.07
Hills	OATS____	CON	INT	86.56	13.06	5.83	2.90	8.86
Hills	OATS____	CON	EXS	82.05	16.45	5.33	2.73	6.82
Hills	MAIZE___	ORG	MAIN	49.55	37.87	6.47	2.63	4.13
Hills	MAIZE___	CON	MAIN	68.17	17.79	6.04	2.99	3.85
Hills	TRITICAL	ORG	EXS	99.46	30.13	5.00	2.22	6.25
Hills	TRITICAL	CON	INT	90.26	10.17	5.75	3.11	9.44
Hills	TRITICAL	CON	EXS	95.04	13.41	5.62	2.85	7.79
Hills	PULSES__	ORG	MAIN	28.06	15.49	6.17	2.17	3.88
Hills	PULSES__	CON	MAIN	14.12	1.98	5.15	2.44	5.53
Hills	RAPE____	ORG	EXS	14.62	32.38	4.86	1.72	3.44
Hills	RAPE____	CON	INT	29.26	14.85	5.77	2.64	5.26
Hills	RAPE____	CON	EXS	43.78	20.51	4.83	2.33	2.48
Hills	SUNFLOWE	ORG	MAIN	44.07	2.17	2.70	1.99	4.34
Hills	SUNFLOWE	CON	MAIN	44.07	2.17	2.70	1.99	4.34
Hills	OTHOILS_	ORG	MAIN	48.56	5.99	7.34	2.39	4.00
Hills	OTHOILS_	CON	MAIN	47.53	5.07	5.15	1.89	2.43
Hills	POTATOES	ORG	MAIN	61.10	23.65	4.92	3.06	4.40
Hills	POTATOES	CON	MAIN	51.57	25.94	6.46	3.81	3.63
Hills	SUGABEET	ORG	MAIN	9.83	16.61	4.85	3.23	2.59
Hills	SUGABEET	CON	MAIN	9.83	16.61	4.85	3.23	2.59
Hills	VEGETABL	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Hills	VEGETABL	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Hills	FRUITS__	ORG	MAIN	0.00	29.10	2.17	1.00	7.76
Hills	FRUITS__	CON	MAIN	0.00	36.55	2.87	1.20	8.62
Hills	VINEYARD	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Hills	VINEYARD	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Hills	TOBACCO_	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Hills	TOBACCO_	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Hills	GRASARAB	ORG	MAIN	60.01	61.94	4.45	2.40	5.98
Hills	GRASARAB	CON	MAIN	77.22	66.21	5.92	2.93	8.60
Hills	FODMAIZE	ORG	MAIN	17.66	32.22	4.26	2.77	6.61
Hills	FODMAIZE	CON	MAIN	25.35	32.92	5.17	3.09	8.40
Hills	FODROOTS	ORG	MAIN	10.16	19.13	3.95	3.06	3.13
Hills	FODROOTS	CON	MAIN	10.16	19.13	3.95	3.06	3.13
Hills	DAIRYCOW	ORG	MAIN		9.13			
Hills	DAIRYCOW	CON	MAIN		9.13			
Hills	SUCKLCOW	ORG	MAIN		1.57			
Hills	SUCKLCOW	CON	MAIN		1.57			
Hills	PORK____	ORG	MAIN		0.31			
Hills	PORK____	CON	MAIN		0.31			
Hills	SOWS____	ORG	MAIN		2.14			
Hills	SOWS____	CON	MAIN		2.14			

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Hills	LAYHENS_	ORG	MAIN		0.28			
Hills	LAYHENS_	CON	MAIN		0.28			
Hills	BROILER_	ORG	MAIN		0.17			
Hills	BROILER_	CON	MAIN		0.17			
Hills	OPOULTRY	ORG	MAIN		0.63			
Hills	OPOULTRY	CON	MAIN		0.63			
Hills	OANIMALS	ORG	MAIN		0.07			
Hills	OANIMALS	CON	MAIN		0.07			
Hills	WHEAT___	ORG	EXS	127.25	32.48	5.52	2.31	6.56
Hills	WHEAT___	CON	INT	230.81	7.99	5.63	3.28	9.80
Hills	WHEAT___	CON	EXS	110.75	10.79	5.82	2.93	8.60
Hills	SPELT___	ORG	EXS	72.42	22.22	3.35	1.98	4.21
Hills	SPELT___	CON	INT	86.56	6.82	4.09	2.92	7.31
Hills	SPELT___	CON	EXS	82.05	27.86	4.34	2.42	3.53
Hills	BARLEY__	ORG	EXS	91.76	28.98	4.63	2.12	5.94
Hills	BARLEY__	CON	INT	92.54	13.33	6.20	3.00	8.86
Hills	BARLEY__	CON	EXS	98.80	17.07	5.62	2.72	6.22
Hills	OTHBRCER	ORG	MAIN	76.41	26.01	4.25	2.09	5.37
Hills	OTHBRCER	CON	MAIN	77.03	9.18	5.15	2.98	8.49
Hills	OTHBRCER	CON	EXS	80.11	18.54	5.17	2.66	5.76
Hills	OTHFOCER	ORG	EXS	76.41	26.01	4.25	2.09	5.37
Hills	OTHFOCER	CON	INT	77.03	9.18	5.15	2.98	8.49
Hills	OTHFOCER	CON	EXS	80.11	18.54	5.17	2.66	5.76
Hills	FIELBEAN	ORG	MAIN	26.72	15.29	6.73	2.19	3.82
Hills	FIELBEAN	CON	MAIN	12.97	1.12	5.23	2.42	6.13
Hills	FIELDPEA	ORG	MAIN	29.39	15.70	5.62	2.15	3.94
Hills	FIELDPEA	CON	MAIN	15.27	2.83	5.08	2.46	4.93
Hills	OTHACROP	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Hills	OTHACROP	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Hills	GRASMEAD	ORG	INT	0.00	38.93	1.90	1.06	7.93
Hills	GRASMEAD	ORG	EXT	0.00	9.71	1.64	0.95	6.66
Hills	GRASMEAD	ORG	LIN	0.00	29.10	2.17	1.00	7.76
Hills	GRASMEAD	CON	INT	0.00	43.27	2.82	1.53	10.38
Hills	GRASMEAD	CON	EXT	0.00	9.71	1.64	0.95	6.66
Hills	GRASMEAD	CON	LIN	0.00	36.55	2.87	1.20	8.62
Hills	OTHEPAST	ORG	MAIN	14.41	34.00	4.16	0.59	7.73
Hills	OTHEPAST	CON	MAIN	17.38	28.42	5.80	1.09	7.74
Hills	EXTEPAST	ORG	MAIN	14.41	34.00	4.16	0.59	7.73
Hills	EXTEPAST	CON	MAIN	17.38	28.42	5.80	1.09	7.74
Hills	ALPIMEAD	ORG	MAIN	7.32	6.89	2.12	0.30	7.32
Hills	ALPIMEAD	CON	MAIN	7.32	6.89	2.12	0.30	7.32
Hills	BERRIES_	ORG	MAIN	0.00	29.10	2.17	1.00	7.76
Hills	BERRIES_	CON	MAIN	0.00	36.55	2.87	1.20	8.62
Hills	OTHPCROP	ORG	MAIN	0.00	29.10	2.17	1.00	7.76
Hills	OTHPCROP	CON	MAIN	0.00	36.55	2.87	1.20	8.62

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Hills	OTHAREA_	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Hills	OTHAREA_	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Hills	DAIRBUL3	ORG	MAIN		6.30			
Hills	DAIRBUL3	CON	MAIN		6.30			
Hills	DAIRBUL2	ORG	MAIN		4.50			
Hills	DAIRBUL2	CON	MAIN		4.50			
Hills	DAIRBUL1	ORG	MAIN		3.20			
Hills	DAIRBUL1	CON	MAIN		3.20			
Hills	DAIRCALV	ORG	MAIN		1.65			
Hills	DAIRCALV	CON	MAIN		1.65			
Hills	SUCKBHEF	ORG	MAIN		6.35			
Hills	SUCKBHEF	CON	MAIN		6.35			
Hills	SUCKFHEF	ORG	MAIN		6.35			
Hills	SUCKFHEF	CON	MAIN		6.35			
Hills	SUCKBBUL	ORG	MAIN		6.35			
Hills	SUCKBBUL	CON	MAIN		6.35			
Hills	SUCKFBUL	ORG	MAIN		6.35			
Hills	SUCKFBUL	CON	MAIN		6.35			
Hills	SUCKCALV	ORG	MAIN		6.35			
Hills	SUCKCALV	CON	MAIN		6.35			
Hills	FACATTLE	ORG	MAIN		4.81			
Hills	FACATTLE	CON	MAIN		4.81			
Hills	FACALVES	ORG	MAIN		1.57			
Hills	FACALVES	CON	MAIN		1.57			
Hills	HORSES__	ORG	MAIN		7.06			
Hills	HORSES__	CON	MAIN		7.06			
Hills	SHEPMILK	ORG	MAIN		2.23			
Hills	SHEPMILK	CON	MAIN		2.23			
Hills	SHEPFATT	ORG	MAIN		2.23			
Hills	SHEPFATT	CON	MAIN		2.23			
Hills	GOATS___	ORG	MAIN		2.28			
Hills	GOATS___	CON	MAIN		2.28			
Hills	OROCLIVE	ORG	MAIN		7.06			
Hills	OROCLIVE	CON	MAIN		7.06			
Hills	DAIRYHEF3	ORG	MAIN		6.30			
Hills	DAIRYHEF3	CON	MAIN		6.30			
Hills	DAIRYHEF2	ORG	MAIN		4.50			
Hills	DAIRYHEF2	CON	MAIN		4.50			
Hills	DAIRYHEF1	ORG	MAIN		3.20			
Hills	DAIRYHEF1	CON	MAIN		3.20			
Hills	FCATCALV	ORG	MAIN		2.67			
Hills	FCATCALV	CON	MAIN		2.67			
Mountains	RYE_____	ORG	EXS	85.94	26.72	4.51	2.15	6.07
Mountains	RYE_____	CON	INT	85.20	13.06	5.83	2.90	8.86
Mountains	RYE_____	CON	EXS	87.17	16.45	5.33	2.73	6.82

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Mountains	OATS____	ORG	EXS	67.51	26.72	4.51	2.15	6.07
Mountains	OATS____	CON	INT	99.52	13.06	5.83	2.90	8.86
Mountains	OATS____	CON	EXS	81.71	16.45	5.33	2.73	6.82
Mountains	MAIZE___	ORG	MAIN	49.55	37.87	6.47	2.63	4.13
Mountains	MAIZE___	CON	MAIN	68.17	17.79	6.04	2.99	3.85
Mountains	TRITICAL	ORG	EXS	63.44	16.93	4.31	2.56	5.91
Mountains	TRITICAL	CON	INT	106.12	10.13	6.12	3.14	11.73
Mountains	TRITICAL	CON	EXS	108.34	13.26	5.92	2.87	10.12
Mountains	PULSES__	ORG	MAIN	28.06	15.49	6.17	2.17	3.88
Mountains	PULSES__	CON	MAIN	14.12	1.98	5.15	2.44	5.53
Mountains	RAPE____	ORG	EXS	14.62	32.38	4.86	1.72	3.44
Mountains	RAPE____	CON	INT	29.26	14.85	5.77	2.64	5.26
Mountains	RAPE____	CON	EXS	43.78	20.51	4.83	2.33	2.48
Mountains	SUNFLOWE	ORG	MAIN	44.07	2.17	2.70	1.99	4.34
Mountains	SUNFLOWE	CON	MAIN	44.07	2.17	2.70	1.99	4.34
Mountains	OTHOILS_	ORG	MAIN	48.56	5.99	7.34	2.39	4.00
Mountains	OTHOILS_	CON	MAIN	47.53	5.07	5.15	1.89	2.43
Mountains	POTATOES	ORG	MAIN	61.10	23.65	4.92	3.06	4.40
Mountains	POTATOES	CON	MAIN	51.57	25.94	6.46	3.81	3.63
Mountains	SUGABEET	ORG	MAIN	9.83	16.61	4.85	3.23	2.59
Mountains	SUGABEET	CON	MAIN	9.83	16.61	4.85	3.23	2.59
Mountains	VEGETABL	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Mountains	VEGETABL	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Mountains	FRUITS__	ORG	MAIN	0.00	16.96	1.61	0.76	7.68
Mountains	FRUITS__	CON	MAIN	0.00	26.50	2.12	0.90	8.31
Mountains	VINEYARD	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Mountains	VINEYARD	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Mountains	TOBACCO_	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Mountains	TOBACCO_	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Mountains	GRASARAB	ORG	MAIN	60.01	61.94	4.45	2.40	5.98
Mountains	GRASARAB	CON	MAIN	77.22	66.21	5.92	2.93	8.60
Mountains	FODMAIZE	ORG	MAIN	17.66	32.22	4.26	2.77	6.61
Mountains	FODMAIZE	CON	MAIN	25.35	32.92	5.17	3.09	8.40
Mountains	FODROOTS	ORG	MAIN	10.16	19.13	3.95	3.06	3.13
Mountains	FODROOTS	CON	MAIN	10.16	19.13	3.95	3.06	3.13
Mountains	DAIRYCOW	ORG	MAIN		9.13			
Mountains	DAIRYCOW	CON	MAIN		9.13			
Mountains	SUCKLCOW	ORG	MAIN		1.57			
Mountains	SUCKLCOW	CON	MAIN		1.57			
Mountains	PORK____	ORG	MAIN		0.31			
Mountains	PORK____	CON	MAIN		0.31			
Mountains	SOWS____	ORG	MAIN		2.14			
Mountains	SOWS____	CON	MAIN		2.14			
Mountains	LAYHENS_	ORG	MAIN		0.28			
Mountains	LAYHENS_	CON	MAIN		0.28			

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Mountains	BROILER_	ORG	MAIN		0.17			
Mountains	BROILER_	CON	MAIN		0.17			
Mountains	OPOULTRY	ORG	MAIN		0.63			
Mountains	OPOULTRY	CON	MAIN		0.63			
Mountains	OANIMALS	ORG	MAIN		0.07			
Mountains	OANIMALS	CON	MAIN		0.07			
Mountains	WHEAT___	ORG	EXS	127.25	9.84	4.89	2.94	7.17
Mountains	WHEAT___	CON	INT	131.69	9.55	6.18	3.26	13.34
Mountains	WHEAT___	CON	EXS	124.04	11.45	6.16	2.89	12.36
Mountains	SPELT___	ORG	EXS	67.51	22.22	3.35	1.98	4.21
Mountains	SPELT___	CON	INT	99.52	6.82	4.09	2.92	7.31
Mountains	SPELT___	CON	EXS	81.71	27.86	4.34	2.42	3.53
Mountains	BARLEY__	ORG	EXS	48.59	20.64	4.65	2.39	8.80
Mountains	BARLEY__	CON	INT	70.08	6.70	4.09	2.66	9.87
Mountains	BARLEY__	CON	EXS	67.65	13.98	4.78	2.54	6.97
Mountains	OTHBRCER	ORG	MAIN	60.74	20.81	4.26	2.34	6.31
Mountains	OTHBRCER	CON	MAIN	82.47	8.93	5.17	2.95	10.00
Mountains	OTHBRCER	CON	EXS	84.17	16.78	5.26	2.67	7.48
Mountains	OTHFOCER	ORG	EXS	60.74	20.81	4.26	2.34	6.31
Mountains	OTHFOCER	CON	INT	82.47	8.93	5.17	2.95	10.00
Mountains	OTHFOCER	CON	EXS	84.17	16.78	5.26	2.67	7.48
Mountains	FIELBEAN	ORG	MAIN	26.72	15.29	6.73	2.19	3.82
Mountains	FIELBEAN	CON	MAIN	12.97	1.12	5.23	2.42	6.13
Mountains	FIELDPEA	ORG	MAIN	29.39	15.70	5.62	2.15	3.94
Mountains	FIELDPEA	CON	MAIN	15.27	2.83	5.08	2.46	4.93
Mountains	OTHACROP	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Mountains	OTHACROP	CON	MAIN	80.33	5.94	6.14	5.59	5.73
Mountains	GRASMEAD	ORG	INT	0.00	24.65	1.40	0.80	7.80
Mountains	GRASMEAD	ORG	EXT	0.00	9.71	1.64	0.95	6.66
Mountains	GRASMEAD	ORG	LIN	0.00	16.96	1.61	0.76	7.68
Mountains	GRASMEAD	CON	INT	0.00	27.99	2.08	1.17	9.60
Mountains	GRASMEAD	CON	EXT	0.00	9.71	1.64	0.95	6.66
Mountains	GRASMEAD	CON	LIN	0.00	26.50	2.12	0.90	8.31
Mountains	OTHEPAST	ORG	MAIN	10.91	22.96	3.16	0.46	7.64
Mountains	OTHEPAST	CON	MAIN	12.95	22.64	4.30	0.86	7.65
Mountains	EXTEPAST	ORG	MAIN	10.91	22.96	3.16	0.46	7.64
Mountains	EXTEPAST	CON	MAIN	12.95	22.64	4.30	0.86	7.65
Mountains	ALPIMEAD	ORG	MAIN	7.32	6.89	2.12	0.30	7.32
Mountains	ALPIMEAD	CON	MAIN	7.32	6.89	2.12	0.30	7.32
Mountains	BERRIES_	ORG	MAIN	0.00	16.96	1.61	0.76	7.68
Mountains	BERRIES_	CON	MAIN	0.00	26.50	2.12	0.90	8.31
Mountains	OTHPCROP	ORG	MAIN	0.00	16.96	1.61	0.76	7.68
Mountains	OTHPCROP	CON	MAIN	0.00	26.50	2.12	0.90	8.31
Mountains	OTHAREA_	ORG	MAIN	65.85	16.16	5.12	4.48	3.48
Mountains	OTHAREA_	CON	MAIN	80.33	5.94	6.14	5.59	5.73

Region	Activity	Farming system	Policy uptake	NO_3 (kg N-eq)	NH_3 (kg N-eq)	N_20 (kg N-eq)	Other nitrogen (kg N-eq)	Phosphorus (total kg PO_4-eq)
Mountains	DAIRBUL3	ORG	MAIN		6.30			
Mountains	DAIRBUL3	CON	MAIN		6.30			
Mountains	DAIRBUL2	ORG	MAIN		4.50			
Mountains	DAIRBUL2	CON	MAIN		4.50			
Mountains	DAIRBUL1	ORG	MAIN		3.20			
Mountains	DAIRBUL1	CON	MAIN		3.20			
Mountains	DAIRCALV	ORG	MAIN		1.65			
Mountains	DAIRCALV	CON	MAIN		1.65			
Mountains	SUCKBHEF	ORG	MAIN		6.35			
Mountains	SUCKBHEF	CON	MAIN		6.35			
Mountains	SUCKFHEF	ORG	MAIN		6.35			
Mountains	SUCKFHEF	CON	MAIN		6.35			
Mountains	SUCKBBUL	ORG	MAIN		6.35			
Mountains	SUCKBBUL	CON	MAIN		6.35			
Mountains	SUCKFBUL	ORG	MAIN		6.35			
Mountains	SUCKFBUL	CON	MAIN		6.35			
Mountains	SUCKCALV	ORG	MAIN		6.35			
Mountains	SUCKCALV	CON	MAIN		6.35			
Mountains	FACATTLE	ORG	MAIN		4.81			
Mountains	FACATTLE	CON	MAIN		4.81			
Mountains	FACALVES	ORG	MAIN		1.57			
Mountains	FACALVES	CON	MAIN		1.57			
Mountains	HORSES__	ORG	MAIN		7.06			
Mountains	HORSES__	CON	MAIN		7.06			
Mountains	SHEPMILK	ORG	MAIN		2.23			
Mountains	SHEPMILK	CON	MAIN		2.23			
Mountains	SHEPFATT	ORG	MAIN		2.23			
Mountains	SHEPFATT	CON	MAIN		2.23			
Mountains	GOATS___	ORG	MAIN		2.28			
Mountains	GOATS___	CON	MAIN		2.28			
Mountains	OROCLIVE	ORG	MAIN		7.06			
Mountains	OROCLIVE	CON	MAIN		7.06			
Mountains	DAIRYHEF3	ORG	MAIN		6.30			
Mountains	DAIRYHEF3	CON	MAIN		6.30			
Mountains	DAIRYHEF2	ORG	MAIN		4.50			
Mountains	DAIRYHEF2	CON	MAIN		4.50			
Mountains	DAIRYHEF1	ORG	MAIN		3.20			
Mountains	DAIRYHEF1	CON	MAIN		3.20			
Mountains	FCATCALV	ORG	MAIN		2.67			
Mountains	FCATCALV	CON	MAIN		2.67			

Source: own compilation based on Nemecek *et al.* (2005) and unpublished SALCA model data

Table 79 List of products

Crop products	Livestock products	Feeding stuffs
Wheat	Milk in tons	Hay with good quality
Rye	Beef from culled cows	Hay with low quality
Spelt	Miscellaneous revenues	Gras silage
Other cereals	Dairy calve 01 month old	Sugarbeet chips
Barley	Dairy calve 04 month old	Soybean meal extract
Oats	Dairy heifer 12 month old	Performance feed for dairy cows
Triticale	Dairy heifer 24 month old	Mineral feed
Grain maize	Dairy heifer 30 month old for replacement	Calf starter
Fodder maize or silage maize in	Dairy bull 12 month ol	MIlked powder for fattening calves
Potatoes	Dairy bull 24 month old	Fattening feed for cattle
Sugar beet	Calve from suckling cow 01 month old	Milk supplement
Rape	Young suckler cow for breeding fattening or slaugthering 12 month old	Skimmed milk
Vegetables	Heifer 24 month old for suckler cow replacement	Skimmed milk supplement
Fodder root cropsin	Beef from fattening suckler cows	Performance feed for sheep or goats
Tobacco	Natua beef	Energy-balance feed for sheep
Field beans	Beef from fattening cattle	Protein-balance feed for sheep
Field peas	Veal from fattening calves	Mineral feed for small ruminants
Sunflower	Milk from sheeps	Fattening feed for lambs
Other oilsseed crops	Beef from fatting sheep	Futtergetreide
Green fodder	Wool from sheep	Concentrates for deers
Vineyards	Young sheep for replacement	Complete feed for fattening pigs
Fruits	Milk from goats	Complete feed for breed suckling pigs
Revenues from growing berries	Beef from fatting goats	Complete feed for breed non-suckling pigs
Revenues from other permanent crops	Young goat for replacement	Feedstuff for pigkets
Other crop products	Revenues from keeping other roughage consuming livetsock	Complete feed for laying hens
Revenues from managing wood	Sows meat in ton slaughter weight	Fattening feed for poultry
	Piglets	Complete feed for rabbits
	Pork meat in ton slaughter weight	
	Eggs from laying hens in ton	
	Young laying hens	
	broiler meat in ton slaughter weight	
	Young broiler for fattening	
	Revenues from keeping horses	
	Revenues from keeping other poultries	
	Revenues from keeping other livetsock	

Source: Sanders (2007), adapted

Annex C Details on results of the model analysis

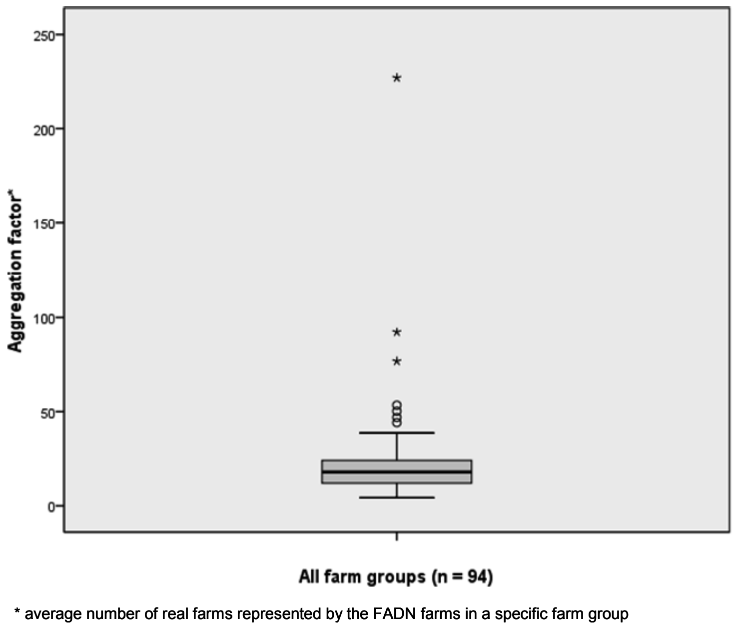

Figure 36 Distribution of UAA on organic and conventional farms per region and farm type

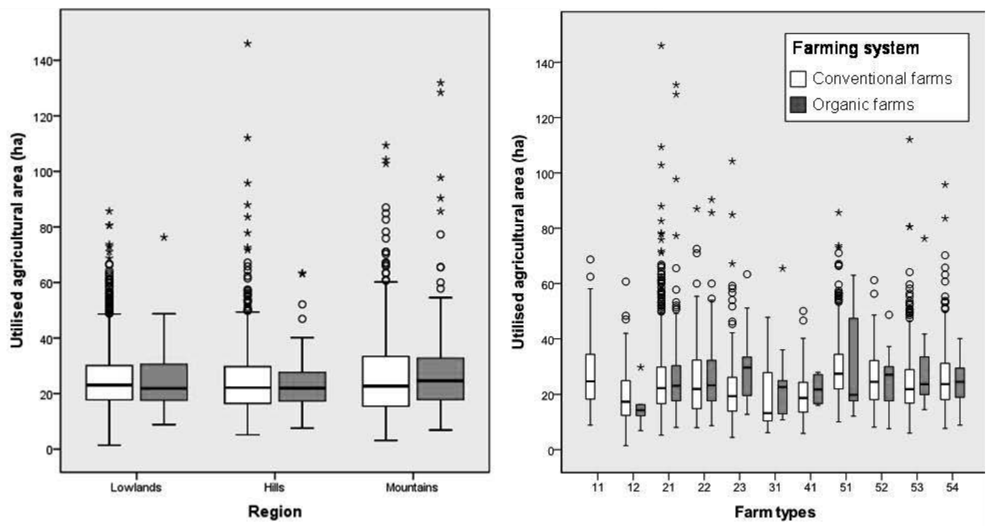

Figure 37 Differences in fossil energy use per ha between organic and conventional farms per region

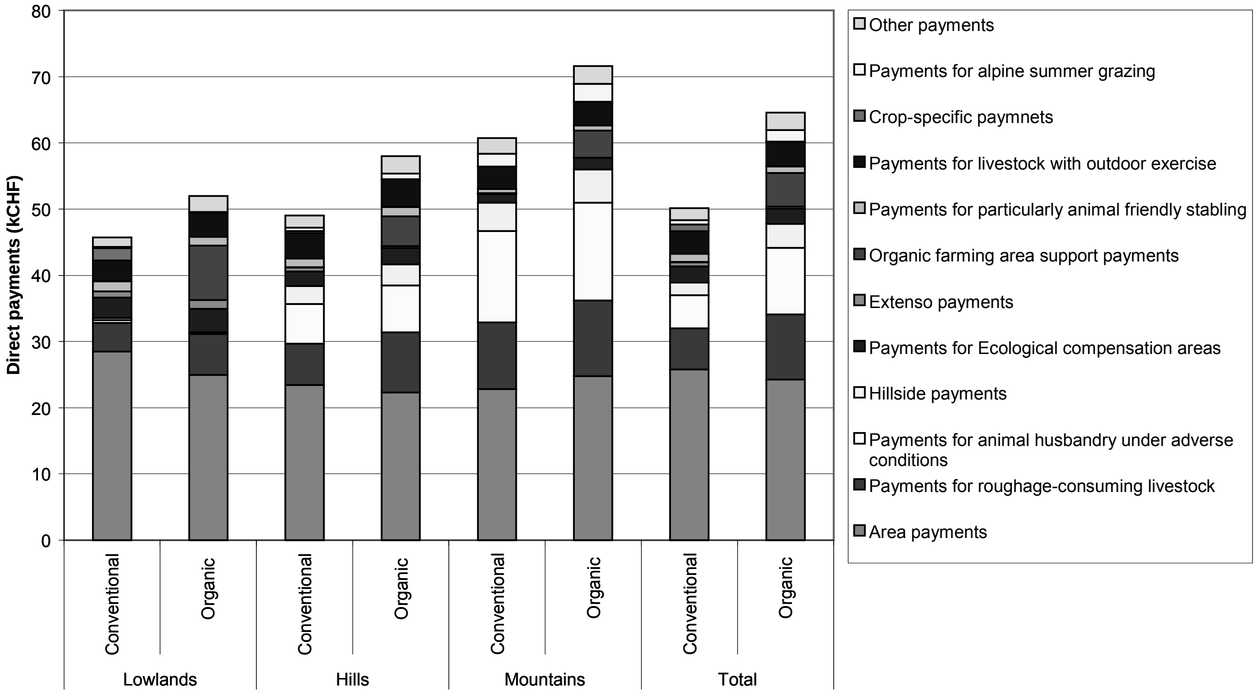

Figure 38 Composition of total direct payments received by representative organic and conventional farms in different regions

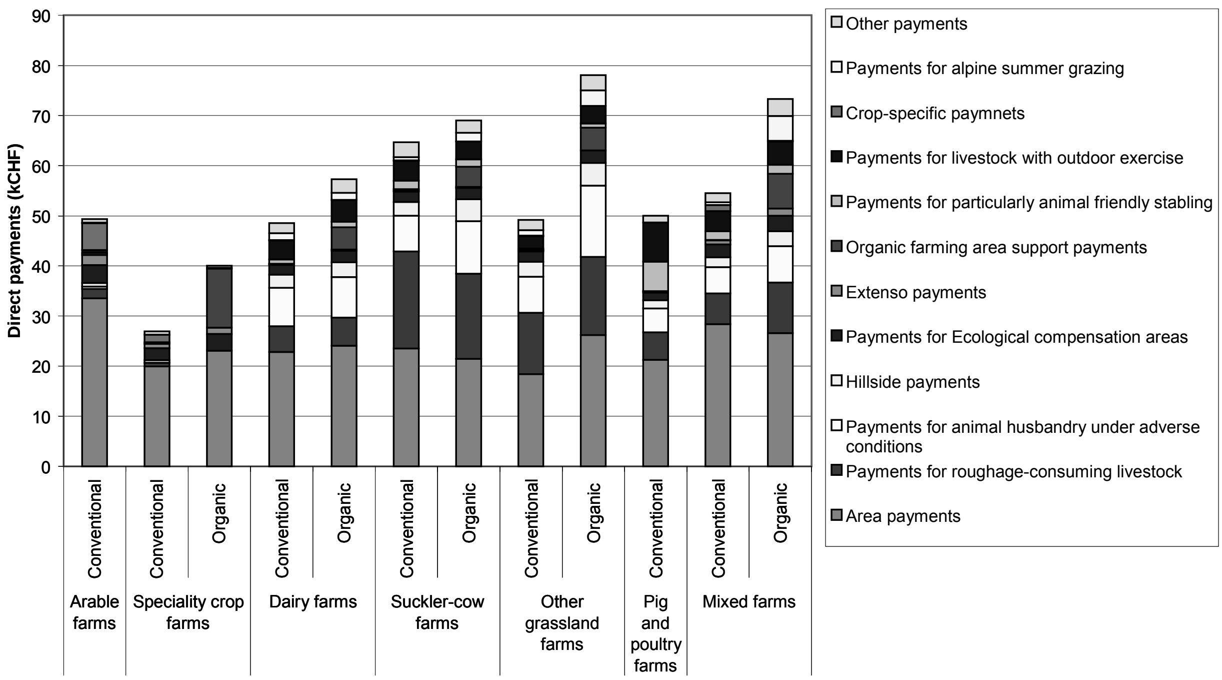

Figure 39 Composition of total direct payments received by representative organic and conventional farm types

The question whether it is economically efficient to pursue environmental policy targets using area support payments for organic farming is both highly policy relevant and methodologically difficult to answer. Using a sector-representative economic model and life cycle assessment data, this Ph.D. thesis calculates the impacts of organic farming on fossil energy use, biodiversity and nitrogen and phosphorus eutrophication for Switzerland. These environmental impacts are related to public expenditure for organic farms and compared to targeted agri-environmental instruments in Switzerland.

The thesis concludes that supporting organic farming via direct payments can be an efficient means for achieving environmental targets.

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