A Modelling Approach for Evaluating Agri-Environmental Policies at Sector Level

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This paper presents a new approach to evaluate the cost effectiveness of agri-environmental policies at sector level. Policy uptake, cumulative environmental effects and public expenditure are identified as the main determinants of cost-effectiveness. On the basis of the sector-consistent, comparative-static, farm group model FARMIS, the determinants of policy cost-effectiveness at sector level are addressed. Firstly, intensity levels for the FARMIS activities are defined in order to model uptake of agri-environmental policies with FARMIS, secondly, life-cycle assessment data is attached to these intensity levels to determine environmental effects of the policies and thirdly, public expenditure is calculated under consideration of transaction costs. This paper concludes delineating the strengths and limitations of the approach.

Keywords: positive mathematical programming, life-cycle assessment, organic farming, uptake rate, environmental indicators, economic efficiency

JEL classification: Q12, Q18, Q57

1. Background

Swiss agricultural policy has been following a progressive ecological agenda since the introduction of direct payments in 1993. Full cross-compliance was already introduced in 1998 and additional ecological services were stimulated by targeted agri-environmental payments in-

Christian Schader et al.: A Modelling Approach for Evaluating Agri-Environmental Policies at Sector Level. Yearbook of Socioeconomics in Agriculture 2008, 93-132

cluding payments for organic management. Against the background of a limited public budget, the considerations of cost-effectiveness play a fundamental role for a further development of the direct payment system.

In this context, both single plot measures like "extensive grassland" and whole farm measures, such as "organic farming", need to be addressed. Particularly organic farming is of interest because in Switzerland as in most other European countries, organic farms receive additional support payments for providing public goods, especially of environmental nature (e.g. Stolze et al. 2000). As this support has led to higher conversion rates (Lampkin and Stolze 2006), the question of cost-effectiveness of the organic area payments is increasingly relevant.

For instance, agricultural economists have two distinct views on the cost-effectiveness of organic farming support payments: One the one hand, von Alvensleben (1998) and Mann (2005a) argue that the organic area payments are not cost-effective, as the policy objectives could be achieved more efficiently by flexible combinations of various agri-environmental measures. The rationale behind this argument was introduced by Tinbergen, who theorised that an efficient policy requires as many specific instruments as there are specific objectives (Tinbergen 1956). On the other hand, the Tinbergen Rule may not be fully applicable in this case due to interactions between policies, conflicting objectives and a limited determinability of different kinds of objectives. Furthermore, the multi-purpose character of organic agriculture could increase its cost-effectiveness due to potentially lower transaction costs as compared to targeted agri-environmental measures (Dabbert *et al.* 2004).

The cost-effectiveness of organic farming has not been evaluated in a consistent quantitative approach at sector level so far. This paper introduces an approach to address this question conceptually by deriving the main determinants of cost-effectiveness at sector level and practically by adapting a sector-consistent farm group model. Specific objectives of this paper are:

- to develop a conceptual approach for deriving the cost-effectiveness of agri-environmental measures at agricultural sector level
- to illustrate the use of programming models against the background of data constraints in *ex-ante* evaluations at sector level

- to describe how the conceptual approach can be implemented for evaluating the cost-effectiveness of Swiss organic farming payments, using a mathematical programming model
- to discuss the strengths and limitations of this approach

2. Modelling cost effectiveness of agri-environmental policies at sector level

This section describes the major determinants of cost-effectiveness at agricultural sector level as environmental effects, uptake and public expenditure. In both the *ex-ante* and the ex-post case, there are substantial data constraints for deriving cost-effectiveness. Therefore, we argue that programming models are useful yet imperfect tools for the evaluation of agri-environmental policies and overcome common gaps of observed data, particularly in *ex-ante* evaluations.

2.1 Conceptual derivation of cost-effectiveness of agri-environmental policies at sector level

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-accentuating cost-benefit analysis. In contrast to cost-benefit-analysis, for cost-effectiveness analysis the effects do not have to be expressed in monetary terms (Drummond 2005).

From a policy-maker perspective, cost-effectiveness is an essential parameter for decision-making, since resources are scare and public money needs to be allocated as efficiently as possible (Pearce 2004). From this perception, cost-effectiveness of a policy relates the public expenditure to the impacts achieved by the policy. In the context of agri-environmental direct payments, the degree to which a policy achieves objectives, determines its effectiveness (Marggraf 2003). Cost is commonly conceived as the payments to the beneficiaries (farmers), opportunity and technical costs as well as the associated transaction costs at farm level and for public administration (Mann 2003).

Contrary to evaluations at plot or single farm level, a sector-level evaluation necessarily requires to consider the uptake or adoption of the policy by farmers, as the uptake of a policy determines how relevant the effects derived by the policy at plot or farm level are at sector level (Osterburg 2004; Mann 2005b). For instance, a policy which leads to significant improvements of biodiversity may not be relevant at sector level if only few farmers decided to adopt the policy on their land.

Therefore, cost-effectiveness of an agri-environmental policy at sector level can be understood as a function of its uptake, its cumulative environmental effect and the cumulative policy-relevant costs.

2.1.1 Uptake of agri-environmental policies

The uptake of agri-environmental measures has been studied many times in both the EU and Switzerland (e.g. Dupraz *et al.* 2004; Mann 2005b). On the one hand, according to surveys of reasons for farmers' adoption of agri-environmental schemes, numerous factors, e.g. the age and education of farmers, influence the uptake decision (Vanslembrouck et al. 2002). Burton explains low uptake rates of agri-environmental programmes with small gains in social capital of farmers (Burton et al. 2008). Often farmers take up agri-environmental policies to generate a perceivable improvement for the environment, while they are convinced that their uptake decision does not depend on economic considerations at all (Jurt 2003). Particularly the uptake of those measures which have a fundamental impact on the farm organisation, e.g. conversion to organic farming, is driven by various economic and non-economic factors, e.g. contact to neighbouring farms and the farmer's environmental motivation (Bichler et al. 2005). Padel (2001) also examines the relevance of adoption theory to understand the rate at which organic farming may be adopted and the goals (financial and non-financial) and type of farmers (pioneers, mainstream early and late adopters) that will be 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.

On the other hand, economic theory says 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 of farmers is supported by empirical evidence, as farmers' uptake rates tend to be higher if opportunity and technical cost of adoption is low. For example, uptake rates of agri-environmental programmes are higher in mountain areas where only an extensive form of production is possible. Furthermore, the less technical costs for farmers occur, the higher is their likelihood to participate in an agri-environmental programme (Mann 2005b).

2.1.2 Environmental effects of agri-environmental policies

The most frequently studied issue about agri-environmental policies is their effectiveness in achieving policy objectives, i.e. minimisation of negative environmental impacts of agriculture (e.g. Stolze *et al.* 2000; Bengtsson *et al.* 2005; Nemecek *et al.* 2005).

In Switzerland, extensive life-cycle assessments of agricultural activities (Swiss Agricultural Life-Cycle Assessments (SALCA)) have been carried out (Nemecek et al. 2005). SALCA data has been calculated for most relevant impacts of agricultural activities representative for Swiss agriculture. Data for the farming activities is differentiated by farming system (integrated and organic farming) and region (valley, hill and mountain region). Furthermore, the environmental impacts of the most important agri-environmental measures are incorporated and most of the relevant impact categories have been analysed.

However, like most of the literature, the effects are studied at field or farm level. Only few studies conceptually combine the effects of the policies on a local level with the achieved uptake, which necessarily has to be done in order to determine the sector level effects of policies (Julius *et al.* 2003; Schmidt and Osterburg 2005; Pufahl 2008).

The basic issue for the upscaling from field or farm level to sector level is whether a linear relation between uptake rates and effects can be assumed. The potential reasons for non-linearity, i.e. decreasing, increasing or variable marginal effects at sector level can be of different nature:

- **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 there is no or al-most no change in management necessary (Henning and Michalek 2008)

- **Regional differences, and differences between farm types:** a measure has a larger impact if it is implemented on a specialised cash crop farm than on an already extensively managed mixed farm (Pufahl 2008).
- **1st Gossen 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 also for non-commodities. 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 et al. 2007).
- **Minimum ecological requirements:** contrary to the 1st Gossen Law, there might also be cases where marginal utility increases with higher uptake. Sometimes a minimum of landscape complexity must be achieved 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 Fig. 1A whereas the marginal environmental effect at sector level ($\frac{\partial E}{\partial U}$ [c1]) may be constant, increasing, variable or decreasing. The run 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, as will be shown below. However, using econometric models the curves can be estimated, provided that individual farm data on the environmental impacts is available (Frondel and Schmidt 2005).

2.1.3 Public expenditure for agri-environmental policies

As Mann (2003) pointed out, there can be different interpretations of costs of policy measures. While some authors understand costs of policy measures as the cumulative payments to the farmers (Wilhelm

1999), Mann (2003) distinguishes between costs at farm level and costs at state level. Farm level costs comprise production cost, opportunity cost, farm-level transaction cost. State-level costs are composed of the payments to the beneficiaries and public level transaction costs (occurring at federal, cantonal and municipality level). Additional tariff revenues due to higher imports have to be deducted from these state-level costs (Mann 2003).

Taking the perspective of a policy maker rather than a farm-level perspective, the costs for public authorities for implementing the policy and achieving environmental effects constitute public expenditure. While the principal share of public expenditure consists of the payments to the beneficiaries, which are meant to compensate the farm-level costs, there is a highly variable share of public transaction costs. Transaction costs occur at different levels: At national level, the overall disbursement of the payments, reporting and supervision of the cantons are the main administrative tasks. At municipality and cantonal level, managing the payments, gathering monitoring and control data and verification of eligibility criteria are major parts of the transaction costs. Farm-level transaction costs, which according to most authors are the main part of total transaction costs, involve filling in forms by the farmer and additional workload due to farm inspections (Tiemann *et al.* 2005; Buchli and Flury 2006). Many studies showed that transaction costs at different levels and for different policies can add up to a significant share of total public expenditure (Vatn 2002; McCann *et al.* 2005). As a special case of an agri-environmental measure, organic farming incurs additional transaction costs for private inspection and certification (Tiemann et al. 2005), although this is often used to reduce transaction costs of public administration by reducing the requirement for agri-environment scheme inspections.

The level of transaction costs depends on the institutional environment, 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 are: asset specificity, uncertainty and frequency of transaction (Williamson 1989); due to these factors the share of policy-related transaction costs 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, for the differences in transaction costs between policies may influence the policy-makers choice (Vatn *et al.* 2002; Rørstad 2007).

Although some authors stress the role of transaction costs as "quality assurance costs" (e.g. Buchli and Flury 2006), there is a general agreement that for efficient policy the share of transaction costs should be kept as small as possible (Jacobsen 2002; Vatn 2002).

As farm-level transaction costs just like opportunity and technical cost are meant to be compensated by direct payments, they should not be added on top of the public transaction costs and the payments to the beneficiaries to public expenditure in the cost-effectiveness evaluation framework. Nevertheless farm-level transaction cost is a relevant parameter to be analysed in an evaluation of cost-effectiveness of policy measures that is useful to be determined separately (Tiemann et al. 2005).

As demonstrated in Fig. 1B, a linear relation between uptake rate 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 from the uptake rate (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. Note, that the transaction costs at farm level (TC_{FARM}) are not a cost component of public expenditure, since per definition these are already remunerated within the payment cost (PC).

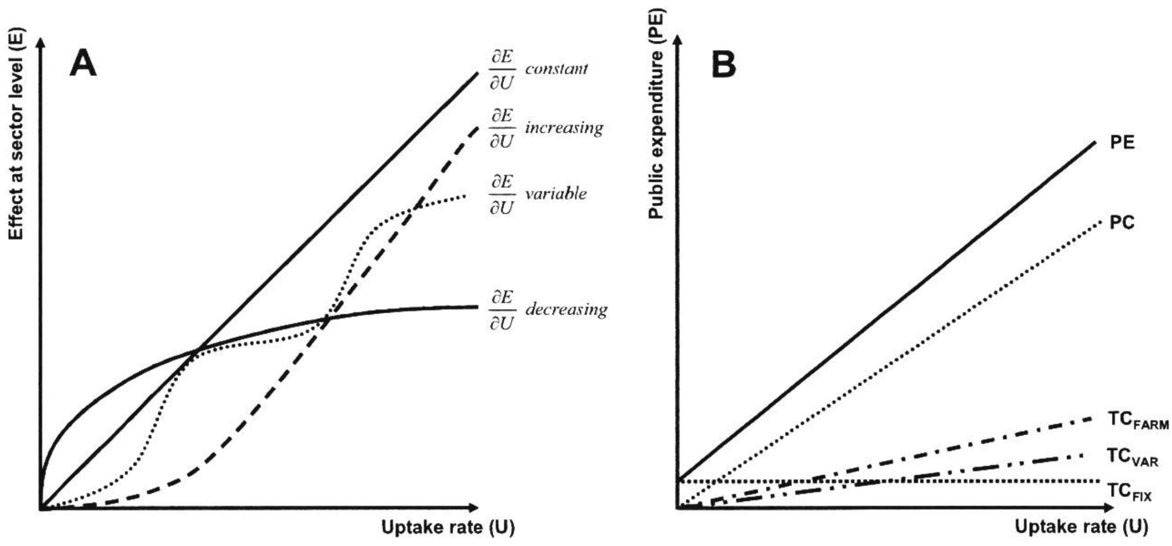

Fig. 1: Environmental effects (A) and public expenditure components (B) of an agri-environmental policy in relation to its uptake rate (own representation).

2.1.4 Linking the parameters for an integrated analysis of cost effectiveness

By drawing together the identified determinants of cost-effectiveness, Fig. 2 shows graphically how different payment levels of a hypothetical agri-environmental policy influence cost-effectiveness at sector level.

The **north-eastern quadrant** of Fig. 2 presents the relation between payment levels for a policy measure and policy uptake[1]. The curve is s-shaped because very small payment levels will not lead to a significant uptake by farmers as long as at least the farm level costs (opportunity, technical and transaction costs) are covered. The more the payment level increases, the higher will be the uptake, with more farms adopting the policy. When a certain uptake level is achieved, it is likely that the farms remaining outside have not entered due to very high opportunity costs or other factors, and much higher payment levels will be required

[1] We assume an agri-envrionmental measure with uniform payment rates, rather than regionally differentiated payment rates or an auction-based policy.

to encourage them to enter. Therefore, the course of the uptake curve is assumed to flatten in the end.

As illustrated above, there is a linear relation between uptake and costs. Thus, the uptake-public expenditure curve, shown in the **south-eastern quadrant of Fig. 2**, runs according to the course of the uptake-payment level curve. The fixed share of transaction costs make the curve not start in the graph's origin.

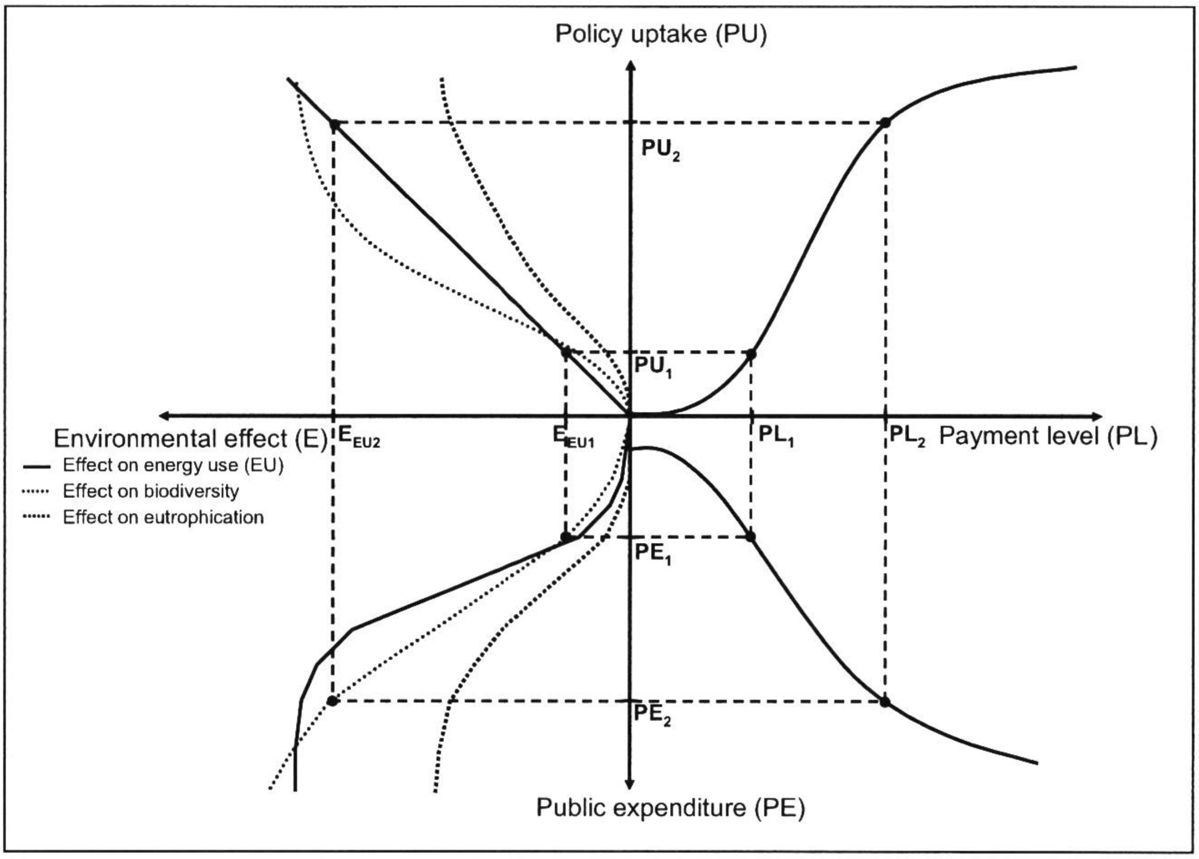

Fig. 2: [c2]*Graphical derivation of the cost effectiveness at sector level for single policy measures in dependence of the payment level (hypothetical example) (own representation).*

The **north-western quadrant of Fig. 2** shows the relation between uptake rate and environmental effect for energy use, biodiversity and eutrophication, as three exemplary environmental impact indicators. Fig. 1A demonstrates that there might be linear relations as well as non-linear relations such as described for biodiversity. For illustration purposes in this paper, we assume a linear effect-uptake relation for energy use, because energy savings are less dependent on local conditions than other environmental effects, not considering the abovementioned reasons for non-linearity. For both eutrophication and biodiversity we assume several reasons as described above.

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 Fig. 2, payment levels lower than PL_1 only cause minimal effects and the additional effects of payment levels beyond PL_2 lead to disproportionately high costs. This effect is even stronger for biodiversity or eutrophication due to the non-linear uptake-effect curve.

2.2 Data constraints for determining cost-effectiveness in ex-post and *ex-ante* evaluations

On the basis of the previous section, we assume that the quality of a cost-effectiveness evaluation of agri-environmental policy largely depends on the availability of data on its main determinants: uptake rate, effects and public expenses.

In *ex-post* evaluations, data on uptake and public expenditure is available until the time the evaluation is carried out. Principally, also data on environmental effects is available, if the respective environmental indicators are monitored regularly (European Commission 2006a). However, the question to which degree the policy under evaluation influenced the environmental indicators, i.e. the additionality of a policy, remains uncertain because other developments could also have influenced the indicator (Pearce 2004).

The core of this question lies in what is described the "fundamental evaluation problem" (e.g. Frondel and Schmidt 2005), which constitutes that we cannot observe the counterfactual situation, i.e. how farms would have developed without taking up the policy (or if the policy were not available). This implies that "the effect of the treatment on the treated" is uncertain (Henning and Michalek 2008). The fundamental evaluation problem can be addressed by both experimental and observational approaches, though experimental approaches are generally not applicable at sector level due to the infeasibility of generating a sufficient sample size. Observational approaches range from before-after comparisons over cross-section, difference-in-difference to matching estimators (Frondel and Schmidt 2005; Caliendo and Hujer 2006).

In *ex-ante* evaluations, i.e. forecasts of potential effects of new or adapted policies, we face the fundamental problem that observational data is not available for future periods. Therefore, there is an even stronger uncertainty in ex-ante evaluation as described for the fundamental evaluation problem (European Commission 2006b).

To clarify the data constraints for evaluations at sector level, a hypothetical example is given. Suppose, an agri-environmental policy introduced at time t_0 may influence an indicator I at short term (t_1) and long term (t_2) (Fig. 3). Conducting a simple 'before-after-comparison', i.e. comparing I (t_0) with the indicator under a certain policy $I_{WP}(t_1)$ or $I_{WP}(t_2)$, does not reflect exactly the real additionality of the policy in question, because other changes might have occurred during that period and influenced the data to an unknown extent (Pearce 2004).

In order to derive the additionality, i.e. the extra effect of a particular policy measure or scheme, a 'with-without-comparison' has to be carried out (Osterburg 2004). A 'with-without-comparison' would involve comparing the indicator with the policy ($I_{WP}(t_1)$) with the indicator without the policy $I_{WOP}(t_1)$ for deriving the short-term effects or $I_{WP}(t_2)$ with $I_{WOP}(t_2)$ for deriving the long-term effects. However, if the policy has been implemented already (*ex-post* case), the situation without the policy (WOP) is unknown. The fact that the subject of evaluation, in our case a group of farms, cannot be observed both under a policy and without a policy at the same time constitutes the "fundamental evaluation problem".

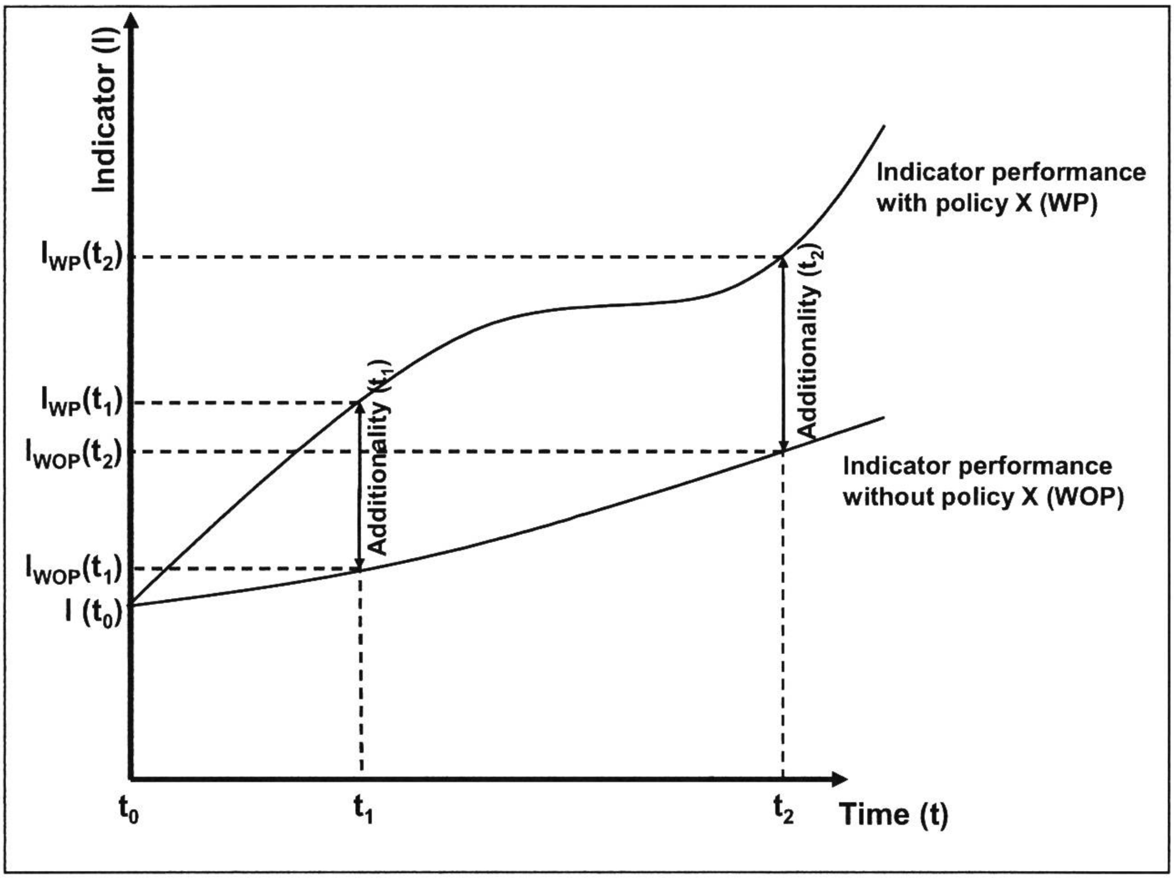

Fig. 3: [c3]*Graphical illustration of the short- and long-term additionally of a policy measure (own representation).*

Either in *ex-post* evaluations only observational on short term (t_1) impacts are available, or, in *ex-ante* evaluations there is no observed data of future years.

Economic models can be used to bridge the data gap, since they allow forecasting responses of the farm sector or simulating reactions to different policy settings, such as the reference situation (WOP). Hence, if empirical data is not available, modelled data can be taken as a substitute, under consideration of its underlying assumptions (Kleinewefers and Jans 1983).

2.3 Current European efforts to model cost-effectiveness determinants at sector level

As Britz & Heckelei (2008) illustrated recently, 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 for the assessment of agri-environmental policies (Mittenzwei et al. 2007). Therefore, according to Britz and Heckelei (2008), there is only exceptional coverage of environmental indicators as a basis for modelling the effectiveness of agri-environmental policies in partial equilibrium models.

The use of programming models for impact assessment of agri-environmental policies at sector level is more common (Britz and Heckelei 2008). There are several approaches addressing uptake, environmental effects or public expenditure, however, integrated approaches covering all aspects are scarce.

2.3.1 Coverage of environmental effects in programming models

Modelling environmental effects at an aggregate level, whether for the agricultural sector, for regions or for different farm types. is a common use for mathematical programming models. In total, 12 European programming models which integrated environmental indicators were found, of which 7 were Positive Mathematical Programming (PMP) models and 5 Linear Programming (LP) models. Furthermore, within the 6th Framework Programme of the EU, several efforts have been started to link models of different classes together in order to be able to address environmental concerns at an aggregate level. Among these approaches are 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 (Kraxner 2006; Jansson *et al.* 2007; 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, dynamisation, and site specificity), as shown for relevant European PMP approaches in Table 1. There are also relevant LP approaches, covering environmental indicators with AROPAj (De Cara et al. 2004) and MODAM (Sattler et al. 2006) among them.

Concerning the geographical scope, all reviewed programming models except CAPRI work at national level. The calibration is done according to supply elasticities for the activities in all models, while 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. Besides 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, while both CAPRI and SILAS currently implement a dynamisation, i.e. a yearly calculation of the reactions 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 the reviewed models in different analytical contexts, using different approaches. For instance, nutrient balances can be modelled either by using completely normative data or according to fertiliser purchase data from FADN, e.g. 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 & Heckelei (2008)). However, only RAUMS and SILAS cover the indicator of pesticide risk or eco-toxicity, whereas in SILAS the indicator is not yet operable. The most problematic aspect about eco-toxicity as an indicator within agricultural sector models is the high variability combined with a high degree of uncertainty.

Even rarer is the coverage of biodiversity indicators within agricultural-sector models. According to Britz & Heckelei (2008) the coverage of biodiversity requires site specificity in the economic model. However, at sector scale, possibilities for site-specific modelling are rather limited. Only RAUMIS (NUTS 3 level) and CAPRI (NUTS 2 level) consider soil types as site-specific information. RAUMIS covers crop diversity, as a habitat diversity indicator, whereas species diversity has not been implemented in an aggregate programming model so far. An 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. 2006), but currently only preliminary results are available. Other authors, however, state that biodiversity impacts can well be covered by economic models, even at a larger scale (Mattison and Norris, 2005). There is certainly a trade-off between ecological relevance and analytical tractability (Eppink and van den Bergh 2007). Overview of the reviewed European PMP models and their characteristics

Table 1: Overview of the reviewed European PMP models and their characteristics

Model	Main publi cation referred to	Geographic scope	Calibration	Regional repre-sentation	Farm type repre-sen-tation	Static/dynamic	Site specificity	Coverage of environmental indicators
CAPRI	Helming (2003)	EU-level, NUTS 1, NUTS 2	Econometric for plant activities supply elasticity for animal activities	YES	indirect represen-tation	Static (dyna-misation in pro-gress)	NO (but soil types considered in results calculation)	N, P, K balances, Ammonia output, Green-house gas emissions Water balances
DRAM	Helming (2005)	The Nether-lands	Supply elasticity	YES	NO	Static	NO	Ammonia emissions, Nitrogen surplus
FARMIS	Bertels-meier, 2004	Selected EU member states and Switzerland	Supply elasticity Intensities based on Röhm-Dabbert-Approach	YES	YES	Static	NO	Currently in develop-ment for CH-FARMIS: Energy use, Eutrophica-tion with N and P Biodiversity (CH)
PASMA	Schmid & Sinabell 2006	Austria	Röhm-Dabbert-Approach, linear approxima-tion	YES	NO	Static	NO	Fertiliser balances
PRO-MAPA.G	Júdez et al. (2001)	Spain	Optional econometric calibration	YES	YES	Static	NO	Nutrient balances
RAUMIS	Julius et al. 2003	Germany, differentia-tion up to NUTS 3 level	Supply elasticity	YES	NO	Static	YES (differentia-tion accord-ing to soil type classifi-cation)	Nutrient balances, NH3 emissions, Pesticide risk, Crop diversity
SILAS	Mack et al. 2007	Switzerland	Supply elasticity	YES	NO	Static (dyna-misation in pro-gress)	NO	Energy use, Eutrophica-tion, Green-house gas potential, Eco-toxicity

2.3.2 Coverage of public expenditure in programming models

Due to their nature as a policy information tool, a necessary common feature of aggregate programming models is coverage of public expenditure for 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 expenditures are 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 allocation of 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.

2.3.3 Coverage of the uptake decision in programming models

Modelling the decision of farms to take up agri-environmental programmes is perhaps the biggest challenge (Britz and Heckelei 2008). Agri-environmental policies are basically implemented in programming models by defining a separate activity for each policy measure. For example, the grassland extensification measure can be implemented by defining the activity "extensive grassland". In LP approaches, if run without calibration restrictions, the problem of overspecialisation can occur, i.e. the 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 model's reactions are not econometrically estimated and therefore to a certain extent arbitrary.

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, a different supply elasticity can be attached to each of them. In other words, the slope of the marginal costs 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 the farm structure.

However, just like the standard PMP approach, the weakness of the Röhm-Dabbert approach remains its arbitrariness (Britz and Heckelei 2008). Thus, the level of exchangeabilty is defined externally and not necessarily on the basis of econometric estimations (Heckelei 2002). Nevertheless, the approach performs more satisfyingly than standard LP or PMP approaches and an alternative approach is currently not available (Gocht 2005; Kanellopoulos et al. 2007). As far as we know, only PASMA adopted the Röhm-Dabbert approach for concrete policy analysis (Schmid *et al.* 2007), while within the EU Integrated Project SEAMLESS, several CAPRI calibration procedures have been tested, the Röhm-Dabbert approach among them (Kanellopoulos *et al.* 2007).

Independent of the exact calibration, the different policy measures can be modelled more easily than others. For example, extensification of grassland or grains can be modelled easily, because the uptake decision is rather straightforward. Organic farming, on the other hand, is much more difficult to model (Schmid *et al.* 2007). A key requirement for modelling whole farm agri-environmental policies, such as organic farming area support payments, is a farm-level representation, as the conversion decision is made for the whole farm. While economic factors like conversion costs and expected changes in farm income influence the conversion behaviour of farmers, several non-economic factors, which cannot be included in the objective function of a programming model, play an important role (Padel 2001; Jurt 2003). Some authors suggest addressing the conversion decision using dynamic models, e.g. based on New Investment Theory (Musshoff and Hirschauer 2004; Odenig *et al.* 2004) or using the qualitative concept of path dependency (Latacz-Lohmann *et al.* 2001). Due to these multiple decision factors an econometric estimation of conversion promises to deliver more realistic estimations of conversion rates than programming models. However, only very few authors have been trying to combine econometric estimations with aggregate programing models so far (Hollenberg 2001).

In essence, we could not identify a modelling approach that addressed the topic of cost-effectiveness of agri-environmental programmes in a consistent way, considering environmental effects, public expenditure (incl. transaction costs) and the uptake decision. However, many approaches exist that could address specific issues such as different environmental effects or the decision to take up agri-environmental schemes.

In view of the determinants of cost-effectiveness and the evaluation problems, we consider programming models to be most suitable to represent agri-environmental policies. If endogenous price reactions are a relevant model feature, programming models need to be linked to partial or general equilibrium models which also consider the demand for agricultural products (Britz and Heckelei 2008).

From the ecological indicator side, however, it seems to be most promising to link programming models either to bio-physical models or reliable environmental indicator data. In view of the availability of standardised SALCA data (Nemecek et al. 2005), the linkage to these data promises interesting results.

3. A modelling approach for Swiss agri-environmental policy evaluation

In this section, we show how the theoretical approach to evaluating the cost-effectiveness of agri-environmental policies at sector level, (outlined in section 2.1) can be implemented practically using a non-linear mathematical programming model. Firstly, the sector-level model FARMIS, used as the basis of this approach, is described. Secondly, we show how the model addresses the determining factors of cost-effectiveness (uptake, environmental effects and public expenditure).

This approach for evaluation is based on the comparative-static farm group model FARMIS, which has been used for policy analysis in Germany since 1998 and 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 with a representation of the agricultural sector based on differentiation by 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. By default, a differentiation is made between different farm types, geographic regions and farming systems. For instance, a farm group of integrated dairy farms in mountain regions could be generated and optimised separately. This differentiation complies with the Swiss standard regional and farm type classification system.

CH-FARMIS is primarily based on farm accountancy data (*Zentrale Auswertung*, equivalent to the European Farm Accountancy Data Network (FADN)) and distinguishes between 30 plant production activities and 15 animal production activities. Positive mathematical programming (PMP) facilitates exact reproduction of the base year situation, and solves the LP-problem of over-specialisation (Howitt 1995). CH-FARMIS is calibrated using supply elasticities as described in Bertelsmeier (2004). The relatively large share of organic farms in Switzerland and the ample coverage of organic farms in the Swiss FADN sample allow the generation and separate analysis of organic farm groups.

In order to evaluate the cost-effectiveness of specific designs of direct payment measures, CH-FARMIS is being extended to include different intensity levels, life-cycle assessment (LCA) data and public expenditure data in order to derive the main determinants of cost-effectiveness at sector level: uptake, cumulative environmental effects and total policy-related public expenditure. These extensions are outlined in the following sections.

3.1 Determination of uptake of agri-environmental measures

Empirical evidence, as shown in chapter 2.3.3, suggests that economic models, based on the assumption of rational behaviour of farmers, are a feasible means to estimate aggregate uptake rates, although for individual farmers non-economic decision factors may play a role, especially in the context of organic farming.

Accordingly, uptake of agri-environmental policies is modelled by defining separate sub-activities reflecting the uptake choices of farmers. Besides the support payments for organic farming, two types of grassland extensification payments as well as the extensification of grains and rape (*Extenso-Beiträge*) are implemented as intensity levels in the model. For the activity “wheat”, for instance, three optional intensity levels are defined: intensive production[2], extensive production according to defined extensification restrictions, and organic production. Since each activity level has defined input/output coefficients, the optimisation process simultaneously considers the different activity intensity levels.

This method of modelling uptake of agri-environmental policies has been used beforehand in PMP models (Schmid and Sinabell 2007) and is the basis of the Röhm-Dabbert approach (Röhm and Dabbert 2003). The Röhm-Dabbert approach involves also more realistic model behaviour by defining the intensity levels as "similar activities", i.e. activities which have similar requirements in terms of machinery and labour input. Without the definition of similar activities, all activities can be exchanged with a similar supply elasticity[3]. However, in reality farmers may be able to easily switch between different intensity levels without replacing their whole machinery or other farm processes. Switching e.g. from wheat production to grassland, requires many changes on the farm, which go along with massive costs for the farms that are considered in the model as 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, there are now two types of quadratic hidden cost parameters (ω) in the extended objective function (1). This implies that hidden costs are divided into a share which depends on the level of the intensity (with ω_{n1} as slope coefficient), and one which is dependent on the level of the other intensities of a particular activity (with ω_{n2} as slope coefficient).

$$\max Z_n = \sum_j \sum_k p_{njk} Y_{njk} - \sum_i \sum_k c_{nik} X_{nik} + \sum_i \sum_k dp_{nik} PX_{nik} - \sum_u r_{nu} U_{nu} - \quad \text{[c4]}$$

$$\sum_v r_{nv} V_{nv} - \sum_l r_{nl} LAND_{nl} - \sum_i \sum_k \delta_{nik} X_{nik} - 0.5 \sum_i \sum_k \omega_{ni1} X_{nik}^2 - 0.5 \sum_i \sum_w \omega_{ni2} X_{niw}^2 \quad \forall n \qquad (1)$$

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

where:

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[2] In Switzerland, more than 95 % of the farms cultivate their land at least according to cross-compliance rules, which require minimum ecological standards regarding nutrient balance, livestock density, and rotation. An agricultural production not fulfilling additional standards is henceforth called 'intensive'.

[3] In this case the supply elasticity is not an own-price elasticity but dependent on both product prices (p) and direct payment rates (dp).

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 ≠ 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)

So, farm-level costs and their compensation via direct payments determine the uptake decision of the model farms. Opportunity cost, technical costs and transaction cost are the three main components of farm-level costs of implementing agri-environmental policies (Mann 2003). Opportunity cost is considered in programming models in the same way as for the ordinary activities, by occupying the scarce farm resources land and labour. Technical costs are included directly within the objective function terms (activity-specific costs (c) and variable costs (r)). Transaction costs are not modelled explicitly but taken into account as hidden cost for the uptake decision (δ, $\omega_{n1,}$ ω_{n2}).

Since data on the uptake of agri-environmental policy is included in the Swiss FADN, FARMIS can be calibrated exactly according to the uptake rates in the base year. All components of the objective function have to be modelled farm-group specifically because farm groups are optimised and reported separately.

The decision of farmers to take up organic farming area support payments, i.e. conversion to organic agriculture, is not modelled explicitly, since the conversion decision is a complex mix of different factors. This makes the adoption of organic agriculture difficult to represent adequately in economic sector models (see also Schmid and Sinabell, 2007) Therefore, the uptake rates of organic farming area support payments have to be considered on the basis of assumptions.

3.2 Determination of environmental effects at sector level

Having defined intensity levels for the FARMIS activities, the environmental effects associated with the uptake of the policies need to be determined. The "driving-force-pressure-state-impact-response" (DPSIR) model (Smeets and Weterings 1999) and the indicator selected within the IRENA-operation (EEA 2005) are used as a framework for the determination of the environmental effects with FARMIS. Driving force indicators covered within FARMIS relate to input use (mineral fertiliser consumption, expenditure for pesticides and energy use), land use (land use change) and trends (intensification / extensification, specialisation / differentiation). As pressures and benefits, the gross nitrogen and phosphorus balance of farm groups is included. Comprised Farm sector responses include "area under agri-environmental support", "area under organic farming" and "organic farm incomes". Finally, state and impact indicators are "eutrophication potential" and "species biodiversity".

With the exception of energy use, all indicators of the domains "driving forces", "pressures and benefits" and "responses" can be directly derived from FARMIS using the Röhm-Dabbert approach (Röhm and Dabbert 2003). Energy use, eutrophication and species biodiversity were determined by linking data from the Swiss Agricultural Life Cycle Assessments (SALCA) data by *Agroscope Reckenholz-Tänikon (ART)* and from the ecoinvent Database (Frischknecht *et al.* 2007).

SALCA data has been calculated for most relevant impacts of agricultural activities that are typical for Swiss agriculture. Data for the farming activities is differentiated by farming system (integrated and organic farming), region (valley, hill and mountain region) and therefore compatible with the classical FARMIS farm groupings. Furthermore, the environmental impacts of the most important agri-environmental measures are covered. Of the possible impact categories, direct and indirect energy use, nitrogen and phosphorus eutrophication and species biodiversity are integrated as three impact indicators for each activity and management intensity in CH-FARMIS.

There are both direct, i.e. on-farm use of primary energy, and indirect energy use components, i.e. inputs for agricultural production, which themselves require the input of primary energy for their production in agriculture. For the modelling of **energy use**, we base our analysis on ecoinvent and SALCA data (Nemecek *et al.* 2005). Additional data is gathered for activities that were not explicitly covered by SALCA or ecoinvent. Both direct (i.e. fuel, gas, electricity) and indirect energy use (i.e. seeds, plant protection, fertiliser, feedstuffs, machines, buildings) are modelled. While most of the energy use components are linked to FARMIS via the model activities, indirect energy use of imported feedstuffs is calculated via the FARMIS-endogenous feed balances of each farm group.

Within CH-FARMIS there is a normative link to the SALCA **eutrophication** data. As the basis of the SALCA eutrophication data, nitrogen and phosphorus models calculate eutrophication potential in dependence of key factors like season and types of application (Prasuhn, 2006; Richner *et al.* 2006). Simultaneously, CH-FARMIS calculates nutrient balances, independent of seasonal differences of application, according to the fertiliser purchase of farm groups, based on FADN data. The model allows a comparison between the results of the eutrophication potential and the nutrient balance. These two parallel procedures of an input and an impact indicator for nutrient enrichment allows mutual comparison and verification of the results of both procedures.

Besides eutrophication effects, **biodiversity** effects belong to the most studied environmental impacts of agriculture (e.g. Bengtsson *et al.* 2005). As there is the general relations of management practices and intensity of agricultural practices (Faucheux and Noël 1995), there is a principal possibility to take into account biodiversity impact within aggregated economic models without referring to site specific characteristics (Mattison and Norris 2005).

The SALCA biodiversity indicators express the habitat quality for 11 groups of species. Groups with high ecological requirements (i.e. amphibians, locusts, butterflies, spiders and carabid beetles) obtain a special emphasis in the biodiversity model. Further, groups of indicator species are flora on arable land, flora on grassland, birds, small mammals, molluscs, butterflies, bees and locusts. The value for total biodiversity expresses a weighed mean of all groups, with weightings according to their specific importance in the food chain of a habitat, as proposed by Jeanneret et al. (2006). The biodiversity model considers the most important species-specific impacts of agricultural crop cultivation practices. This allows for a detailed coverage of the impacts of agricultural policies on species level at macro-scale.

Against the background of potential non-linear relations between uptake rates and environmental effects at sector levels as described in section 2.1.2, sensitivity analyses have to be conducted assuming non-linear curve progressions due to e.g. deadweight effects or decreasing marginal utility.

3.3 Determination of public expenditure

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 summing up the payments to the beneficiaries (PC) (2). Furthermore, variable as well as fixed transaction costs at cantonal and national level are added (TC_{VAR} and TC_{FIX}), while farm-level transaction cost is not considered, as it is meant to be compensated by the direct payments already. Except for the fixed transaction costs, all components are calculated for each farm group separately to allow for a comparative analysis between farm groups. This is especially necessary, because the organic farming area support payments are not covered by the Röhm-Dabbert approach in FARMIS but by configuring separate farm groups.

The total transaction cost (TC_{TOTAL}) of a policy is estimated (3). As illustrated in section 2.1.3, assessing the total transaction cost as a separate indicator is relevant for policy analysis, because policies with lower farm-level transaction cost, eventually do 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 farm-group specifically to be later able to report specifically per farm group and agri-environmental policy. The additional consideration of total transaction costs is of particular interest with regard to the organic farming area support payments.

$$PE_{TOTAL} = \sum_{n} \sum_{i} \sum_{k} (PC_{nik} + TC_{VAR\,nik}) + TC_{FIX} \tag{2}$$

$$TC_{TOTAL} = \sum_{n} \sum_{i} \sum_{k} (TC_{FARM_{nik}} + TC_{VAR\,nik}) + TC_{FIX} \tag{3}$$

where:

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

The payments to the beneficiaries are obtained from FADN and public expenditure statistics. Transaction cost data is derived from recent Swiss and international studies (Mann 2003; Buchli and Flury 2006), data gaps are bridged by polling a set of experts. While Buchli and Flury (2006) calculated transaction costs for common and ecological direct payments, only roughly differentiating between different agri-environmental measures, Mann (2003) focussed on agri-environmental payments, and calculated separate values for single measures.

4. Conclusions

Static programming models are a well-suited option for assessing the cost effectiveness of agricultural policies at sector level (Britz and Heckelei 2008). The comparison of European models showed that currently, static PMP models are the most widespread sector modelling approach integrating environmental concerns (Helming 2003; 2006a;

Mack *et al.* 2007). Coverage and the data origin of the environmental indicators are supposed to vary among the existing approaches. While environmental indicators are used in many existing models, agri-environmental policies, as part of pillar 2 of the EU CAP, are rarely addressed.

Against the background of the literature discussed in chapter 2, the approach suggested in this paper is a consistent quantitative way to model cost-effectiveness of agri-environmental policies at sector level under consideration of their environmental effects, public expenditure and uptake rates. Particular **strengths and limitations** of the approach as compared to existing models are listed below.

While most comparable approaches particularly address the issue of environmental effects, our approach draws special attention to the cost side and the uptake questions and their interdependencies. Unlike other existing approaches transaction costs at different levels are explicitly considered in the model. Therefore, the cost-effectiveness of agri-environmental policy is covered by addressing environmental effects, public expenditure and uptake as shown in section 2.1.4.

Due to the characteristic of FARMIS, to group the farms flexibly, e.g. according to farm type, region and farming system is a major advance of this approach, farm-group specific cost-effectiveness of policies can be modelled. The ability to separately analyse organic farms is an essential requirement to analyse the cost-effectiveness of payments made to organic farms. Further classification criteria, e.g. according to the uptake of agri-environmental measures or livestock density or farm size, are also useful in the analysis of specific policies.

Not only evaluations of single agri-environmental policy measures but also comprehensive policy scenarios and the interactions and consistency between policies can be assessed. Furthermore, comparative analysis of the economic and environmental performance of different farm types and farming systems lies within the scope of this approach.

With the PMP calibration, hidden costs can be considered within the objective function, this solves the standard-LP problem of over-specialisation, and allows to exactly calibrate the model in the base year without calibration constraints.

The most important (in terms of area covered) Swiss agri-environmental payments are covered (extensive meadows, less intensive meadows, Extenso-payments for grains and rape and organic farming area support payments). Particularly the cost-effectiveness assessment of organic farming support payments is a major advantage compared to other models.

A comprehensive coverage of DPSIR indicators is implemented, for assessing environmental consequences broadly. Particularly, eutrophication potential and species biodiversity have rarely be implemented using an environmental dataset of comparable quality.

Taking the perspective of policy makers, the approach is tailor-made to contribute to the current process of reorientation of the Swiss direct payment system.

On the other hand some important **limitations** go along with this approach. The flexible grouping of the farms is limited by FADN data constraints. Farm groups cannot be split up above a certain extent, since a farm group has to consist of a minimum number of farms to achieve a satisfying degree of representativeness for the agricultural sector.

Since farms cannot switch between groups, structural change has to be taken into account indirectly, either by introducing restrictions for accessible land, and/or by conducting sensitivity analyses. For the same reason the impact of farms converting to organic farming can only be assessed using assumptions for the uptake rate. However, an indication that scenarios may induce conversion or re-conversion is, if land is traded between organic and conventional farm groups.

In relation to PMP, calibration lacks foundation in economic theory, because the shadow prices in the base and target years are assumed to be equal. Furthermore, the Röhm-Dabbert-Approach is criticised for its arbitrarily assigned supply elasticity coefficients (Britz and Heckelei, 2008). As interrelations between model activities are disregarded, econometric specification of the model is seen as superior to PMP (Heckelei 2002).

In terms of significance of the environmental effects calculated at sector level, the assumption of a linear relation between uptake and environmental effects is daring. Uncertainty about the course of the uptake-effect curves (e.g. due to self-selection bias) has to be addressed by sensitivity analyses. In particular, the eutrophication and biodiversity indicators have to be interpreted with care, as non-linearity of these effects is potentially strong. However, the only way to get over this limitation was to introduce environmental data into the FADN at a sufficient sample size and data quality.

Furthermore, the model only works with average environmental assessments, rather than on a single farm basis. For instance, an average value for energy use for extensively managed wheat is used, although empirical farm-level analyses show a high variation. As environmental impacts often depend on site specific characteristics, such as slope, soil type or landscape complexity, aggregating the environmental impacts, forming a broad average over many farms can only provide limited information, especially regarding complex environmental indicators such as biodiversity. Nevertheless, average values for different habitats can be useful to indicate the environmental performance of the agricultural sector or specific farm groups.

Lastly, the approach does not allow for efficiency calculations in a macro-economic sense, as market responses and impacts on other sectors are not considered.

Even so, the results of a model as described in this paper can contribute substantially to the knowledge on cost-effectiveness of agri-environmental policies, particularly organic farming area support policies, at sector level.

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