Counterfactual approach for assessing agri-environmental policy: The case of the Finnish water protection policy

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Abstract – *This paper applies counterfactual approach to assess the impacts of agri-environmental programs. We focus on* ex-post *policy evaluation in a case where the control group is* de facto *non-existent and treatment group covers the whole population. We employ a theoretical framework based on profit maximization and the interlinkages between the behavior of agents and the response of environmental systems to the economic decisions. Optimization in the absence of policies produces the control case, while optimization under the policies fits to the treatment case. We apply our model to assess the performance of the Finnish Agri-Environmental Programme to reduce agricultural nutrient runoff to the Baltic Sea. We demonstrate that the Finnish Agri-Environmental Programme does not achieve its goals, because it fails to anticipate farmers' responses to incentives created by the Common Agricultural Policy and the Agri-Environmental Programme itself. The social cost-benefit analysis of the Programme shows negative net benefits.*

Keywords: *Environmental policy evaluation, Counterfactual analysis, Nutrient runoff, the Baltic Sea*

Approche contrefactuelle pour évaluer les politiques agro-environnementales : le cas de la politique de protection de l'eau finlandaise

Résumé – Le présent document utilise l'approche contrefactuelle pour évaluer les impacts des programmes agro-environnementaux. Nous mettons l'accent sur l'évaluation *ex-post* des politiques dans le cas où le groupe de contrôle est *de facto* inexistant et le groupe de traitement couvre l'ensemble de la population. Nous utilisons un cadre théorique basé sur la maximisation du profit et sur les liens entre le comportement des agents et la réponse des systèmes environnementaux aux décisions économiques. L'optimisation en l'absence de politiques correspond au cas du contrôle, tandis que les politiques d'optimisation correspond au cas du traitement. Nous appliquons notre modèle pour évaluer les performances du Programme agro-environnemental finlandais visant à réduire le ruissellement des nutriments agricoles dans la mer Baltique. Nous démontrons que le Programme agro-environnemental finlandais n'atteint pas ses objectifs, car il ne parvient pas à anticiper les réponses des agriculteurs aux incitations créées par la Politique Agricole Commune et le Programme Agri-environnemental lui-même. L'analyse coûts-avantages du programme montre des avantages nets négatifs.

Mots-clés : évaluation de la politique environnementale, l'analyse contrefactuelle, le ruissellement des éléments nutritifs, la mer Baltique

JEL Classification: Q18, Q28 and Q58

1. Introduction

Common agricultural policy (CAP) of the EU includes a possibility for voluntary environmental protection programs in agriculture. The programs provide an incentive payment for compensating the compliance costs and farmers' private transaction costs. In many countries these voluntary programs entail significant monetary transfers from tax payers to farmers. This monetary transfer is economically justified provided that the provision of public goods and reduction of negative externalities match the payments, that is, provided that these programs are environmentally effective with regard to their goals and cost-efficient in the allocation of government outlays.

It is important to assess the success of these voluntary agri-environmental programs. As it is well-known, it is a challenging task to assess any program, save a case where environmental effects are deeply involved. One method for assessing environmental policies is to contrast them with an alternative no-policy case and thereby reveal the impacts of policies, that is, to resort counterfactual analysis. *Counterfactual* analysis belongs to the basic tools for policy evaluation in economics. Counterfactual analysis answers the question: what if…? A comparison of the counterfactual with the actual case sheds light to the critical factors explaining the impacts of policy. Statistical and experiments policy evaluation require typically data on the treatment group (group subject to policy in question) and the control group (group not subject to policy) and that the policy instrument is well-defined. The statistical and econometric techniques for counterfactual analysis are well developed (see Heckman and Vytlacil, 2007 and Heckman, 2010).[1]

The *ex-post* analysis of agri-environmental policy is complicated by the fact that the data is seldom available on all relevant variables at farm level for this kind of analysis. Moreover, sometimes the whole population may be subject to the policy under scrutiny (for instance, roughly 95% of the Finnish farmers participate in the Finnish Agri-environmental Programme). Statistical analyses of policies suit well for cases that entail a partial participation, so that both the treatment and control groups can be well identified. In the absence of an evident control group, counterfactual analysis must search for other approaches than statistical techniques. In this paper, we suggest for the case where the control group is absent, an approach that creates the control group in an hypothetical way, based on the use of behavioural assumptions, that is the farmers' profit maximization hypothesis. The farmers maximize profits both in the presence and the absence of policy. Hence, we suggest that if the control group is absent, it can be created by allowing the farmers to maximize

[1] A recent example of a counterfactual econometric analysis of macroeconomic development is developed by Chang-Jin *et al*. (2008). For a general formalization of counterfactuals and critical discussion on counterfactual, see, for instance, Galles and Pearl (1998) and Dawid (2000), respectively, and the references of these articles. Kluve (2004) provides an example of the philosophical treatment of the subject.

profits without the policy in question to yield the control group. The choices over the use of inputs and land allocation between crops can be linked to nutrient runoff. Other relevant environmental process functions that predict environmental outcomes in the presence and absence of the policy can then be contrasted to the goals. They measure the outcomes of agri-environmental policies.[2]

We develop a theoretical framework and derive the counterfactuals for empirical analysis to examine the *ex-post* success of agri-environmental policies. We are not aware of this type of counterfactual analysis in the context of agri-environmental policies. We apply our theoretical frame to agricultural water protection policy which aims at reducing nutrient runoff from arable lands to the Baltic Sea. In Finland, like in all Baltic Sea countries, agriculture is the main source of nitrogen and phosphorus loads (60% of the anthropogenic phosphorus and 52% of nitrogen loads) to the Sea (Helcom, 2010). Given that nutrient loads from point sources have been reduced considerably, pressure to reduce loads from agriculture is high and agri-environmental program tries to cope with this challenge. The first Agri-Environmental Programme (AEP) ran for the period 1995-1999. Although the program addressed many environmental issues (water quality, air quality, biodiversity and landscape), improving surface water quality has been and still is the highest priority—also in the subsequent programmes (2000-2006, 2007-2013) that have only slightly fine-tuned the original programme. The overall goal of the programme is to reduce 30% of nitrogen and phosphorus loads relative to the 1994 pre-program nutrient loading level.

Over the three periods, the Agri-environmental Programmes provide a fairly rigid water quality improvement package, which, however, can be divided into two schemes: the *General Protection Scheme* that targeted all farmers, whereas the *Supplementary Protection Scheme* includes more specialised and environmentally effective measures targeted only a limited number of farmers. In the General Protection Scheme, the basic mandatory measures are environmental planning and monitoring, upper limits for fertiliser use, plant protection, buffer strips (3 meters wide), and wintertime plant cover, biodiversity maintenance and landscape management. Measures, in the Supplementary Protection Scheme, include organic production and conversion to organic production, establishment and management of buffer zones (15 meters wide), sedimentation ponds and wetlands, promotion of biodiversity, management of traditional biotopes, management and development of landscapes, extensification of production, controlled

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[2] For evaluation of environmental effects of agri-environmental policy based on the participating and non-participating groups, see Primdahl *et al*. (2003) and for the assessment of agri-environmental policy see Hanley and Oglethorpe (1999) and Hanley *et al*. (1999). For a study on the impacts on biodiversity based on analysis of reported studies, see Kleijn and Sutherland (2003). Pufahl and Weiss (2009) analyse the impact of AEP on input use, land allocation and use of agrichemicals.

subsurface drainage and lime filter drainage, proper management of manure, liming of acid sulphate soil and raising of native breeds.

Finnish authorities expected that agri-environmental program could achieve about 30-40% decrease of phosphorus and nitrogen runoff by 2010 (MMM, 2000). Even more ambitious 50% nutrient runoff reduction targets by 2005 were set for agriculture in the Government resolution as regards surface water protection targets (YM, 2000). Despite the ambitious goals and all efforts and support payments, something has gone wrong, however. No actual reduction in measured/monitored nutrient runoff can be observed and especially nitrogen runoff is reported to have increased.[3] This raises many kinds of questions. Are the selected abatement measures to reduce nutrient loading inappropriate? Is there something wrong in the self-selection of farmers in the program? Or do factors that are external to agri-environmental policy, such as market parameters and other agricultural policies explain the no-progress? Criticism has been presented towards both farmers (for noncomplying behaviour) and natural scientists (for suggesting inefficient instruments). Interestingly, though, Finland is not alone in its agricultural water protection problems. Similar no-progress can be found, for instance, in Sweden (and in many other countries).

Clearly, a theory-guided analysis of the water protection policy in agriculture is needed to evaluate the impacts of the water protection policy and to find an explanation for the observed no-progress in actual nutrient loads. The number of possibilities impacting the outcomes of the program is large and contains both program-based factors and changes in external factors. To examine the performance of agri-environmental policy, we develop theoretically and employ empirically two basic counterfactuals. They answer the following two questions: what would have happened to nutrient loads from agriculture 1. if land allocation would have remained the same as in year 1994 for which the program was designed, and 2. if no voluntary agri-environmental programme would have been implemented at all in 1995 but the rest of the CAP would be in force. While the first counterfactual measures the effectiveness of the means included in the Programme, the second counterfactual describes the preventive impact of the Programme relative to market solution with no agri-environmental policy package. We modify these two counterfactuals in many ways to examine the potential impacts of many kinds of external factors, especially those raised by the CAP.

[3] The environmental effectiveness of Finnish agri-environmental programme has been analyzed in following research projects: MYTVAS 1 (1995-1999), MYTVAS 2 (2000-06) and MYTVAS 3 (2007-2013). MYTVAS stands for "Follow-up of the effectiveness of the agri-environmental programme". While MYTVAS 1 reported some positive changes in cultivation practices and nutrient runoff, MYTVAS 2 and MYTVAS 3 have reported more mixed evidence. Phosphorus runoff is slightly decreased but nitrogen runoff has increased. Moreover, soil phosphorus content in the main agricultural area in South-Western Finland has increased.

The rest of the paper is structured as follows. Section 2 provides our theoretical framework and section 3 the empirical framework. Section 4 presents the data and section 5 is devoted to the empirical analysis. In section 6, we provide a brief cost-benefit analysis of the Finnish Agri-environmental Programme and concluding section 7 ends the paper.

2. The framework: a counterfactual approach

Consider an introduction of a new agri-environmental policy, A, starting a period t = T. This ends the previous policy regime, which can be no policy (free market solution), or some other policy denoted by B. Now, let and denote the vector of instruments of no policy intervention and policies A and B, respectively. For no policy intervention, this instrument set is naturally equal to zero, $x_0 = \{0\}$. To keep the presentation simple, policy instruments under policy A are a fertilizer application constraint ($\bar{l}^A$), a buffer strip requirement ($\bar{m}$), the CAP area payment (a) and the environmental support payment per hectare (b), that is, $x^A = \{\bar{l}^A, \bar{m}, a, b\}$. Furthermore, the previous policy regime B is assumed to consist of direct price support (s) and a requirement for large set-aside areas (E), that is, $x^{\mathrm{B}} = \{s, E\}$.

The farmers optimize their agricultural profits subject to exogenous variables and the policy instruments under each policy regime. Denote the conventional response function of crop i as $y = \mathrm{f}_i(l_{\mathrm{i}})$ with $f_i'(l_i) > 0$ but $f_i''(l_i) < 0$. Let p_{i} be the price of the crop and c the price of fertilizer input, and L_{i} be the amount of land allocated to each crop under the three policies. Market parameters alone or together with the instruments under policy regimes A and B determine the optimal use of inputs and land allocation (including entry and exit of land in agricultural sector). We next develop the farmer's choices under each policy regime.

Under no-policy intervention (denoted by superscript 0), the profits of a given parcel allocated to crop i are given by $\pi_i^0 = p_i f_i(l_i) - c$, and profits from the land area allocated to crop i by $r_i = \pi_i^0 L_i^0$. The optimal solution entails $l_i^0(p_i, c)$ and $L_i^0(p_i, c, p_{-i})$, where $p_{\text{-i}}$ refers to prices of the other crops and $\sum_{i=1}^{n} L_i^0(p_i, p_{-i}, c) = L^0$, where L^0 refers to overall land in cultivation in the no-policy regime.

Under the new policy regime A, the *per* parcel profits are given by $\pi_i^A = (1-\bar{m})[p_i f_i(l_i) - cl_i] + a + b$ subject to $l_i^A \leq \bar{l}_i$. At the optimal solution output price and fertilizer cost no longer impact fertilizer intensity, because the fertilizer application constraint is binding ($l_i^A = \bar{l}_i$); also the buffer strip is at mandated level $m^A = \bar{m}^A$. Using these two mandatory figures, the overall amount of fertilizer applied to each hectare is $l_i^A = (1-\bar{m})\,\bar{l}_i$. Land allocation, however, continues to depend on the relative profitability of each crop, and thereby it depends on prices, so that $L_i^A\,(p_i, c, p_{-i}, a, b)$, where *a* refers to CAP compensation payment and *b* refers to area-based environmental support payment. The overall profits are then given by $r_i^A = \pi_i^A L_i^A$ with $\sum_{i=1}^{n} L_i^A\,(p_i, p_{-i}, c, a, b) = L^A$, where L^{A} refers to overall land in cultivation in the policy regime A.

Under the previous policy B the *per* parcel and total profits for land in cultivation are given by $\pi_i^B = (p_i + s)\, f_i\,(l_i) - c$ and $r_i^B = \pi_i^B L_i^B$, respectively. The optimal fertilizer intensity is a function of crop price, fertilizer price and price support, $l_i^B\,(p_i, s, c)$ and land allocation between crops depends on relative profitability as follows: $L_i^B = L_i^B\,(p_i, p_{-i}, c, s)$. For the total amount of land in cultivation, it holds that $\sum_{i=1}^{n} L_i^B + E = L^B$, where L^{B} refers to overall amount of land in cultivation in the policy regime B (recall E is the mandatory fallow area).

The environmental quality is a function of the input use, the amount of cultivated arable land, and its allocation between the crops. Let function G represent the way the use of inputs in agriculture transform to environmental quality, nutrient runoff in our case. Then, drawing on the above discussion environmental quality can be expressed for our three cases as a function of respective optimal choices:

$$G^0 = G^0\left(\sum_{i=1}^{n} l_i^0\,(p_i, c)\, L_i^0\,(p_i, p_{-i}, c)\right) \tag{1}$$

$$G^A = G^A\left(\sum_{i=1}^{n} l_i^A L_i^A\,(p_i, p_{-i}, c, a, b) + \varepsilon\bar{e}\right) \tag{2}$$

$$G^B = G^B\left(\sum_{i=1}^{n} l_i^B\,(p_i, s, c)\, L_i^B\,(p_i, p_{-i}, c, s) + \varepsilon\bar{E}\right) \tag{3}$$

where β denotes runoff from fallow land ($\bar{e}$ under CAP and $\bar{E}$ under the old regime). Recall, our aim is to assess the performance of the new agri-environmental policy A. Let $\bar{G}^A$ be the announced environmental target of the new policy; in our case reduction in nutrient loads, while the observed environmental quality under this policy is G^{A}. Naturally, the difference between the goal and the actually measured nutrient runoff, $G^A - \bar{G}^A$, can be any sign and is due to multiple reasons. The challenge of the counterfactual analysis is to explain this difference.

We can now use the above analysis to formulate our two counterfactuals, which recall, were the following. What would have happened to G^{A} if the land allocation between crops and green set-aside would not have changed from policy regime B? Second, what would have happened to G^{A} if no voluntary agri-environmental policy would have taken place when Finland joined the EU? Economic mechanisms present in (2) and (3) readily suggest how to formalise these counterfactuals (*CF*):

$$CF_1 = G^{CF_1}\left(\sum_{i=1}^{n} l_i^A L_i^B\left(p_i, p_{-i}, c, s\right) + \varepsilon E\right) \tag{4}$$

$$CF_2 = G^{CF_2}\left(\sum_{i=1}^{n} l_i^0\left(p_i, c\right) L_i^A\left(p_i, p_{-i}, c, a\right)\right) \tag{5}$$

Taking the difference G^{A} - CF_1 and G^{A} - CF_2 allows us to evaluate the relative role of input use intensities and land use changes of the agri-environmental policy regime A. Counterfactual CF_1 allows us to define the unit effectiveness of the instruments in policy regime A and counterfactual CF_2 in turn defines the preventive impact of the policy A on nutrient loads. In the empirical part we also consider the role of some fine-tunings of policy regime A on environmental impacts; they are our minor counterfactuals that are developed in a similar way as (4) and (5).

3. Empirical Framework

Crop yield response to fertilizer

Per hectare crop yield is modelled as a function of nitrogen fertilization. By assumption, farmers use a compound fertilizer that contains nitrogen and phosphorus in fixed proportions and in the absence of constraints choose the application rate of fertilizer on the basis of yield response to nitrogen application. The crop yield function for spring wheat, barley and oats is assumed to follow the Mitscherlich form,

$$y_i = \mu_i\left(1 - \sigma, e^{-v_i N_i}\right) \tag{6}$$

where y_{i} is yield per hectare, N_{i} is nitrogen use per hectare, and μ_{i}, σ_{i} and ν_{i} are parameters. These parameters are estimated by Bäckman *et al*. (1997) on the basis of Finnish field experiments. The yield function for rape, silage and hay is assumed to have the quadratic form

$$y_i = A_i + \chi_i N_i + \gamma_i N_i^2 \tag{7}$$

where y_i is yield *per* hectare, N_i is nitrogen use *per* hectare, and A_i, χ_i and γ_i are parameters. Parameters for rape have been estimated by Pietola *et al.* (1999) and parameters for silage and dry hay are based on Lehtonen (2001) (see Table A3 in Appendix). The parameter values reflect the average growing conditions in the Southern, Western and Central Finland.

Optimal fertilizer use

Farmer's short-run restricted profits π^i are given by equation (8a) for spring wheat, barley, and oats and by equation (8b) for rape, silage and hay.

$$\pi^i = (1 - \overline{m})[p_i \mu_i (1 - \sigma_i e^{-v_i N_i}) - c_i N_i] \tag{8a}$$

$$\pi^i = (1 - \overline{m})\,[p_i \left(A_i + \chi_i N_i + \gamma_i N_i^2\right) - c_i N_i] \tag{8b}$$

where π^i is farmers' *per* hectare profits, p_i is output price for a given crop (i) and c_i is nitrogen price for a given combined fertilizer (NPK), and $\overline{m}$ finally, denoted the mandatory buffer strip between field and waterways. Output and fertilizer prices come from agricultural statistics (Yearbook of Agricultural Statistics provided by Information Centre of the Ministry of Agriculture and Forestry). Optimal nitrogen application level can be solved by taking first-order conditions with respect to nitrogen application N and setting them to equal zero and then solving for optimal N.

Nutrient runoff

The modelling of nutrient runoff follows Lankoski *et al.* (2006) who modelled nitrogen and phosphorus runoff on the basis of Finnish data. For phosphorus runoff we account for both dissolved reactive phosphorus (DRP) and particulate phosphorus (PP). Farmers use a compound fertilizer (NPK) and as these main nutrients are in fixed proportions, nitrogen fertilizer intensity determines also the amount of phosphorus used. Part of this phosphorus is taken up by the crop, while the rest accumulates and builds up soil P.

We use the following nitrogen runoff function based on Simmelsgaard (1991) and Simmelsgaard and Djurhuus (1998),

$$Z_N^i = \left(1 - \overline{m}^{\alpha}\right) \phi_i \exp\left(b_0 + bN_i \left(1 - \overline{m}\right)\right) \tag{9}$$

where Z_N^i = nitrogen runoff at fertilizer intensity level N_i, kg/ha, ϕ_i = nitrogen runoff at average nitrogen use and taking into account the share of manure in total nutrient use, $b_0 < 0$ and $b_0 > 0$ are constants and N_i = nitrogen fertilization in relation to the normal fertilizer intensity for the crop, $0.5 \leq N \leq 1.5$. This runoff function represents nitrogen runoff generated by a nitrogen application rate of N_i *per* hectare and the parameter $+_i$ reflects differences in crops. The first Right Hand Side (RHS) term of (9) describes nitrogen uptake by the buffer strips. We calibrate to reflect Finnish experimental studies on grass buffer strips (Uusi-Kämppä and Yläranta, 1992, 1996; Uusi-Kämppä and Kilpinen 2000). The second RHS term represents nitrogen runoff generated by a nitrogen application rate of N_i *per* hectare when buffer strips take up a share $\overline{m}$ out of this land.

In the case of phosphorus, both dissolved and particulate runoff is modeled. Drawing on Finnish experiments (Saarela *et al.*, 1995) it is assumed that 1 kg increase in soil phosphorus reserve increases the soil P status (i.e., ammonium acetate-extractable P) by 0.01 mg/l soil. Uusitalo and Jansson (2002) estimated the following linear equation between soil P and the concentration of dissolved phosphorus (DRP) in runoff: *water soluble P in runoff (mg/l) = 0.021*soil_P (mg/l soil) – 0.015 (mg/l)*. The surface runoff of potentially bioavailable particulate phosphorus is approximated from the rate of soil loss and the concentration of potentially bioavailable phosphorus in eroded soil material as follows: potentially bioavailable particulate phosphorus PP (mg/kg eroded soil) = 250 * ln [*soil_P* (mg/l soil)]-150 (Uusitalo 2004). Thus, the parametric description of surface phosphorus runoff is given by

$$Z_{DRP}^i = (1 - \overline{m}^{\kappa})\bar{\omega}_i[\psi_i(0.021(\Phi + 0.01^*(1 - \overline{m})P_i) - 0.015]/100 \tag{10a}$$

$$Z_{pp}^i = \left(1 - \overline{m}^{\kappa}\right) \Delta_i[\zeta_i\{250 \ln(\Phi + 0.01^*(1 - \overline{m})P_i) - 150\}]^* 10^{-6} \tag{10b}$$

where β and κ denote the reductive effect of the buffer strips as regards dissolved and particulate phosphorus runoff, respectively ψ_i is runoff volume (mm), Φ is soil_*P* (common to all crops) and ζ_i is erosion kg/ha, and P_i is the phosphorus application rate. As in the case of nitrogen, the crop, soil textural class, share of manure in the total nutrient input and field slope based differences in the runoff of dissolved and the potentially bioavailable particulate phosphorus are captured by parameters $\bar{\omega}_i$ and Δ_i, respectively. Soil_ *P* is fixed at 12.6 mg/l in 1995, 11.6 mg/l in 2001 and 10.6 mg/l in 2007 on the basis of the average for Finnish soil test samples taken on those respective years (MMM 2004 and Myyrä *et al.*, 2005).

4. Data and nutrient runoff under the Finnish agri-environmental programme

Following the theory, we need to develop the empirical counterparts for equations (4) and (5). We need data on land allocation, fertilizer application intensities and nutrient runoff. Data on the development of actual land-use and its allocation between the main crops is given by farm statistics. We use fertilizer restrictions of the agri-environmental program during the years 1995-2007. Using annual crop and fertilizer prices, we solve the privately optimal fertilizer intensity. Nutrient runoff is estimated by plugging the fertilization limits and the privately optimal fertilization rates in our runoff functions.

4.1. Data on land use and fertilizer application

Table 1 shows the land use change between the main crops in 1994 (just before the beginning of current policy) and during the agri-environmental program. We present the land area of those crops that are external to the analysis under the land use class "Other". It includes crops, such as sugar beet, potatoes and peas.[4]

Table 1. **Land use (ha) in 1994, average 1995-1999, average 2000-2006, and 2007 (*Yearbook of Farm Statistics*)**

Land use	1994	1995-1999	2000-2006	2007
Wheat	77,600	97,700	156,729	167,900
Barley	505,700	560,120	554,757	550,100
Oats	334,300	372,660	395,786	361,500
Rape	67,200	66,980	76,500	90,200
Hay	257,900	224,700	123,100	103,100
Silage	268,400	321,300	393,000	438,100
Fallow	505,100	188,400	214,929	231,500
Other	285,700	314,460	300,414	312,900
Total	**2,301,900**	**2,146,320**	**2,215,214**	**2,255,300**

In 1994 much of arable land was left idle and fallowed because of uncertainty created by the new policy regime. It has been gradually taken back to production: during 1995 - 2007 the total amount of cultivated land increased by 5.3%. A more important feature is that land allocation between different crops has changed much. Land allocated to wheat cultivation has increased from 4.1% to 7.4% and that of silage from 14.1% to 19.4%. between 1995 and 2007. The share of barley, oats and rape has remained quite

[4] In what follows we present the data and analysis in terms of averages of the first and second phase of the Finnish agri-environmental program. Annual data and results for each year are available from the authors upon request.

stable but that of dry hay has decreased from 13.4% to 4.6%. Thus, land area under more fertilizer intensive crops wheat and silage has increased. Finally, relative to land allocation in year 1994, a considerable amount of fallow land was released to cultivation.

Table 2 presents the fertilization limits of the agri-environmental program and the fertilizer intensity under the hypothetical case, in which farmers optimize on the basis of relative prices (labeled as "N private").

Table 2. **Average optimal nitrogen use intensity and nitrogen application constraint in agri-environmental program in 1995-1999, 2000-2006, and 2007.**

Crop	1995-1999	2000-2006	2007	N constraint[5]	N private	N constraint[6]
	N private	N constraint	N private			
Wheat	157	100	127	100	138	120
Barley	120	90	102	90	112	100
Oats	99	90	84	90	96	100
Rape	160	100	132	100	147	110
Silage	166	180	192	180	227	240
Hay	131	90	128	90	156	100

With the exception of silage in the first and third program period and oats in the second and third program period, the nitrogen application constraint has been binding. For wheat, rape and hay the privately optimal application rate has been clearly higher than the constraint.

In order to estimate the nutrient loads, we also need to determine the amount of manure produced and used in fertilization and the land areas allocated to mandatory buffer strips. Drawing on MYTVAS (2010) we report *per* hectare nitrogen and phosphorus produced in livestock manure in Table A4 in Appendix. The amount of mandatory 3 meter wide buffer strip has been stable over time, since the share of farmers participating in the program has been stable, and has covered roughly 0.19% (4,150 ha) of total land area (MMM, 2004). The amount of voluntary buffer zones (15 meter wide) has gradually increased: 2,154 ha in 1995, 4,721 ha in 2001, and 7,518 ha in 2007. Altogether these land areas covered in 2007 only 0.5% of the overall arable land.

[5] Application constraints for years 2000-2006 and 2007 refer to base level fertilization constraints. Figures in Table 2 takes into account differential level of constraints according to region and soil textural class as well as specific constraints related to more accurate fertilization measure and reduced fertilization measure of the program.

[6] Clay and silt soil textural classes in southern and central Finland.

4.2. Estimating nutrient runoff under the agri-environmental programme

We estimate the average *per* hectare nitrogen and phosphorus runoff by employing equations (9)-(10b) for each crop. Total nutrient runoff for each crop is obtained through multiplying the average runoff by the total land area allocated to each crop. Recall, the runoff of dissolved phosphorus depends on the state of soil phosphorus, which behaves dynamically in time (see e.g. Schnitkey and Miranda, 1993 and Iho, 2010). Therefore, we use the reported annual average estimates for soil phosphorus to ensure that runoff of dissolved phosphorus reflects the actual development of both soil phosphorus and phosphorus fertilizer application. For the agri-environmental programme we fix the $soil\_P$ at 12.6 mg/l in 1995-1999, 11.6 mg/l in 2000-2006 and 10.6 mg/l in 2007. For private solution we use constant figure 12.6 mg/l, because the private use of phosphorus fertilizer does not decrease from 1995 to 2007.

Table 3 presents the estimated average *per* hectare nutrient runoff under the fertilizer application constraints reported in Table 2. We calibrated our N and P runoff functions to reflect the runoff figures of the Finnish VIHMA model (a tool to assess nutrient and sediment runoff from agricultural catchments) (Puustinen *et al.*, 2010). We take into account the reductive impact of buffer strips and account for the fact that part of nutrients are given as manure (the shares of manure and chemical fertilizers are given in Table A4 in Appendix) and drawing on Simmelsgaard and Djurhuus (1998) we let manure application cause slightly higher runoff than chemical fertilizers.[7]

Table 3. **Nitrogen, dissolved phosphorus (DRP) and particulate phosphorus (PP) runoff, kg/ha, under constrained fertilizer use intensity in 1995-1999, 2000-2006, and 2007**

Crop	1995-1999			2000-2006			2007		
	N	DRP	PP	N	DRP	PP	N	DRP	PP
Wheat	14.5	0.47	0.60	14.7	0.44	0.59	15.5	0.40	0.57
Barley	13.5	0.47	0.60	13.7	0.44	0.59	13.6	0.41	0.58
Oats	13.5	0.47	0.60	12.9	0.43	0.59	13.6	0.39	0.56
Rape	14.5	0.44	0.59	14.6	0.44	0.59	14.5	0.40	0.57
Silage	6.0	0.78	0.29	6.5	0.73	0.28	7.4	0.63	0.26
Hay	4.6	0.70	0.27	4.7	0.65	0.27	4.9	0.60	0.26
Fallow	10.6	0.40	0.50	10.6	0.37	0.47	8.8	0.38	0.41
Other	10.5	0.69	0.36	10.8	0.65	0.36	11.0	0.61	0.34

[7] The manure created in animal husbandry is applied to the fields within the farm or transported to more distant farms. Traditionally animal husbandry farms have enough arable area to facilitate the application of manure but about 10% is transported outside the farm limits. The fertilization limits hold true for manure application, as well.

Due to the relaxation of the nitrogen application constraints, the last program period witnesses the highest *per* hectare nitrogen runoff for wheat, oats, silage and hay. Instead, the *per* hectare runoff of both particulate and dissolved phosphorus has decreased steadily over time due to the decrease in soil phosphorus.

Table 4 combines the land areas of crops and the *per* hectare average nutrient runoff to produce total nitrogen and phosphorus runoff in tons under the Finnish Agri-Environmental Programme. In terms of our theory, Table 4 corresponds to equation (2). In the Table 4, we employ the actual land allocation and justify it as follows. During the first two program periods the national crop area payments that complemented the CAP area payments were designed to determine the desired land allocation between crops. Crop prices will pay a higher role under the third program period with the single farm payment, which is independent of crops and therefore less distortionary (see e.g. Lichtenberg, 2002, Guyomard *et al*., 2009). Agri-environmental area payment does not impact the land allocation between alternative cultivated crops but it affects land allocation between crops and set-aside as well as entry-exit margin as we see in section 5.3.

Table 4. **Total nitrogen and phosphorus runoff (tons) under constrained fertilizer use in 1995, 1995-1999, 2000-2006, and 2007**

Crop	1995		1995-1999		2000-2006		2007	
	N	P	N	P	N	P	N	P
Wheat	1956	121	2170	135	3531	208	3992	211
Barley	10689	711	11598	772	11653	741	11484	706
Oats	6819	454	7716	514	7849	521	7547	448
Rape	1894	114	1487	89	1717	102	2001	114
Silage	2708	366	2940	391	3901	454	4977	453
Hay	2012	321	1575	251	871	130	773	102
Fallow	2296	190	1938	160	2216	177	1987	180
Other	4903	394	5085	399	4900	360	5274	357
Total	**33278**	**2671**	**34509**	**2711**	**36638**	**2694**	**38035**	**2571**

Nitrogen loads have increased during all three periods of the program due to increased amount of cultivated land, the higher share of land devoted to fertilizer intensive crops, and relaxed nitrogen constraints during the third program period. Phosphorus runoff follows a different path: the gradual decrease of the soil phosphorus content decreases phosphorus runoff in the second and third phase of the program. The reason for the opposite directions of nutrient loads is that phosphorus application is roughly constant across crops, while nitrogen application varies considerably between crops.[8]

[8] How reliable is our approach to developing total load estimates? The Finnish Environment Institute reported that in 2008 agricultural loads were the following: P 2,750 and N 39,500. Thus, our estimates are fairly close to those figures. According

We can, therefore, conclude that the Finnish Agri-Environmental Programme has failed to reduce nitrogen loads to inland waters and the Baltic Sea. This failure is partly explained by the inadequate design of the chosen policy instruments. However, the success of the programme depends ultimately also on the amount of cultivated land and its allocation between crops, which depends on relative crop prices, Common Agricultural Policy and the use of national crop area payments. Therefore, one needs a deeper analysis that extracts one by one the influence of intervening variables and that reveals the true effectiveness of the programme.

5. The impacts of the Finnish agri-environmental programme on nutrient runoff: A counterfactual analysis

We now turn to the analysis of the environmental performance of the program. We first assess the effectiveness of the policy instruments, fertilizer application constraints and mandatory buffer strips, in reducing nutrient runoff. We do this by isolating the impacts of these intensive margin instruments from changes taking place in the extensive margin, that is land allocation, as equation (4) suggests. We then follow equation (5) to trace out how much the agri-environmental program has potentially offset nutrient runoff relative to the non-regulated market-based development, and also assess separately the impact of fertilizer limits and buffer strips.

5.1. The effectiveness of instruments targeting nutrient runoff

In Table 5 we examine what would have happened to nutrient loads if agricultural land use would not have changed but stayed the same as it was either in 1994 or 1995. This represents our counterfactual CF_1 indicating how nutrient loads would have evolved if the amount of cultivated land and its allocation between the crops would have been the same as 1995 (the first year of Finland's membership in the EU) or in 1994 (last year of the old policy regime before the EU membership and introduction of the Finnish agri-environmental program). We call the actual loads under the current policy regime as Baseline. The difference Baseline - CF_1 in the last two rows allows us to assess how effective the chosen instruments actually are.

to Table A4 in Apppendix the total nutrient input *per* ha has been 151.5 kg N/ha in 1995, 131.3 kg N/ha in 2001, and 123.6 kg N/ha in 2007. The difference to the fertilization limits is explained by the fact that a part (6%) of arable land is outside the agri-environmental program and constrained only by Nitrate directive with 170 kg as nitrogen application limit. Also, there are many root crops and vegetables cultivated under much higher nitrogen application constraint (from 120 to 240 kg/ha) than those crops we focus on in more detail in this paper. Taking these factors into account shows that, for instance, in 2007 total nitrogen input was 124 kg/ha (Table A4) and average constraint would have allowed average N application of 130 kg/ha.

Table 5. **Total nitrogen and phosphorus runoff (tons) under constrained fertilizer use in Baseline and in the case of total cultivated land and its allocation fixed to correspond to that of 1994 or 1995**

Land allocation	**1995-1999**		**2000-2006**		**2007**	
	N	**P**	**N**	**P**	**N**	**P**
Baseline	34509	2711	36638	2694	38035	2571
Fixed 1995	33327	2671	32988	2526	33813	2391
Fixed 1994	35262	2813	34849	2662	35051	2523
Difference 1995	1182	40	3650	168	4222	180
Difference 1994	-753	-102	1789	32	2984	48

Table 5 reveals that under either 1995 or 1994 land allocation total nitrogen loads would have increased much less and phosphorus loads decreased slightly more by 2007 than they actually did. Therefore, we immediately conclude that the chosen measure, fertilization limits and buffer strips are far less effective means to reduce nutrient loads than originally thought.

In 2007, the 1995 land allocation could have decreased phosphorus runoff relative to Baseline by 180 tons and the 1994 land allocation 48 tons, representing respectively 7.0% and 1.9% reduction of phosphorus loads. The difference of fixed 1995 land allocation and Baseline represents the increase in total phosphorus load mainly due to the increased land area in cultivation, since changing land allocation between crops has only negligible impact on total phosphorus runoff. Under the Baseline the actual decrease in phosphorus loads from 1995 to 2007 was 100 tons only, representing 3.7% reduction over the three program periods.

Under the Baseline the nitrogen loads increased remarkably by 4,757 (14.3%) from 1995 to 2007. The increase in the loads is explained by three factors: laxer nitrogen fertilization constraints, increased cultivation of fertilizer intensive crops, and the steady increase in the amount of cultivated land. Relaxation of nitrogen fertilization constraints increased nitrogen loads by 752 tons in the Baseline, (15.8% increase), relative to a situation where constraints would not have been relaxed but stayed the same as in 2001. The impact of the increase in the amount of cultivated land and the change in land allocation towards more nitrogen intensive crops was 4005 tons, representing 84.2% of the load increase.

In 2007, the 1995 land allocation could have decreased nitrogen runoff relative to Baseline by 4,222 tons and 1994 land allocation by 2984 tons, representing respectively 11% and 8% reduction in nitrogen loads. During the years 1995–2007 and relative to Baseline in 1995 nitrogen loads would have increased by 535 tons (1.6%) under 1995 land allocation and by 1,773 tons (5.3%) under 1994 land allocation.

We next scrutinize further the impact of the policy and ask what is the separate contribution of the fertilizer constraints and buffer strips on the prevention of nutrient loads under the Baseline. Table 6 provides the answer.

Table 6. **Relative effectiveness of fertilizer constraints and buffer strips**

	1995-1999		2000-2006		2007	
	N	**P**	**N**	**P**	**N**	**P**
Baseline	34509	2711	36638	2694	38035	2571
Fertilizer constraints only	35280	2750	37476	2735	38888	2612
Buffer strips only	40701	2714	39169	2704	42713	2590
Buffer impact	6192	3	2531	10	4678	19
Fertilizer constraint impact	771	39	838	41	853	41

Had the agri-environmental program relied only on buffer strips, nitrogen runoff in the Baseline would have been almost 4,700 tons more (12.3%) and phosphorus runoff 19 tons more (0.7%) in 2007. If only fertilizer use constraints would have been implemented then nitrogen runoff in the Baseline would be 853 tons more (2.2%) and phosphorus runoff 41 tons more (1.6%) in 2007. Thus, fertilizer use constraints are more effective than buffer strips in reducing nitrogen runoff, while opposite holds in the case of phosphorus runoff.

Recall that the Finnish authorities expected 30-40% decrease of phosphorus and nitrogen runoff by 2010. Table 5 shows that for phosphorus a 10% reduction target would have been realistic given the inefficiency of the instruments and provided that no behavioural responses would have taken place. For nitrogen the targets represent just wishful thinking. Thus, Table 5 indirectly indicates how badly environmental authorities anticipated the farmers' behavioural adjustment when the new agri-environmental programme was launched.

5.2. Preventive impact of the Finnish Agri-Environmental Programme

Let us ask next what would have happened without the program when the farmers are allowed to choose their fertilizer application rates freely on the basis of crop and fertilizer prices.[9] This is our second counterfactual that is defined by equation (5). To answer the question, we must solve nutrient runoff *per* hectare under market solution and link this to land allocation between the crops.

We report in Table A2 (Appendix) the estimated *per* hectare nutrient runoff when farmers choose the privately optimal fertilizer application rates freely on the basis of annual crop and fertilizer prices. From Table A2, nitrogen

[9] When Finland joined the EU, the national price support was eliminated. Moreover, Finland had to waiver tariffs that safeguarded domestic agricultural production. Since then the Finnish market prices of crops have followed fairly closely those of the EU internal markets.

runoff *per* hectare is much higher than that under the fertilizer use constraints reported in Table 3; in fact, it is double for most of the crops. Also, both forms of phosphorus runoff are much higher than corresponding figures in Table 3. Thus, fertilizer use constraints have clearly reduced the estimated average *per* hectare nutrient runoff relative to the market-based free optimum.

Agri-environmental support payments compensate for reduced fertilizer use and buffer strips in crop production. CAP policy entails mandatory fallow requirements (up to year 2008) aimed mostly to reduce overproduction. In addition, farmers may voluntarily fallow land above the mandatory limits compensated by the CAP. In Finland the average mandatory fallow has been roughly 7% of the cultivated area of cereals and oilseeds (as Table 1 suggest—between 70,000-80,000 hectares); the rest is voluntary. Given that the Finnish agri-environmental payments exceed the costs of the reduced fertilization and revenue loss from buffer strips, these payments have increased profitability of crop production relative to fallowing. Referring to a recent estimate on the impact of agricultural support payments on land allocation (Laukkanen and Nauges, 2012), we assume that in the absence of agri-environmental program the fallow land area would be 10% higher.

Table 7 conveys information on nutrient loads under the baseline and private solution. The difference between the two figures indicates how much the Finnish Agri-Environmental Programme has prevented nutrients loads by its presence in each program phase.

Table 7. **Total nitrogen and phosphorus runoff (tons) under constrained and free private fertilizer use in 1995-1999, 2000-2006, and 2007**

	1995-1999		2000-2006		2007	
	N	**P**	**N**	**P**	**N**	**P**
Baseline	34509	2711	36638	2694	38035	2571
Private	37143	2554	35792	2598	38861	2666
Difference	-2634	157	846	96	-826	-95

As Table 7 reveals, the preventive effect of the Finnish Agri-Environmental Programme has changed between program periods and nutrients. During the current program period one can witness a reduction in both phosphorus and nitrogen loads. During the first two periods the increase in the amount of land in cultivation increased runoff and it outweighed the reductive effect of input use constraints in the case of phosphorus; private solution would have resulted in lower loads. For nitrogen this happened in the second program period partly due to increased costs of nitrogen fertilization.

Combining Tables 5-7 allows us to make the following conclusion. The programme has been somewhat successful in controlling nitrogen runoff at the intensive margin but this success has been outweighed by a failure to control both extensive margin (land allocation) as well as entry-exit margin (total amount of cultivated land). Consequently, nitrogen loads have increased and phosphorus loads have decreased less than expected.

5.3. Impact of agri-environmental support payment on entry-exit margin

If agri-environmental support covers just the costs of water protection actions (including private transaction costs), the farm profits would remain unchanged and land use at extensive margin as well. However, if the compensation is in excess of costs, the area-based support tends to increase entry of land to agricultural production.[10] This is the case with the Finnish Agri-Environmental program in which the compliance costs of required actions are much lower than the payment for the actions. This means especially that farms with zero or slightly negative profits could make positive profits thanks to the support payment. Hence, agri-environmental support has significant economic impact on production, which has to be considered when assessing both efficiency and effectiveness of the support.

There are three studies that provide us a starting point to assess the impact of the support payment. Siikamäki (1996) assesses that 18% of farmers in Southern Finland (Support region A) and 10-12% of farmers in other parts of Finland would have ceased their production altogether without agri-environmental support. This estimate is confirmed by Koikkalainen and Lankoski (2004) who find that the agri-environmental support represents 8-11% of total return in cereal farms, 3-4% in hog farms and 5-6% in dairy farms. Moreover, as regards farm income the agri-environmental support represents about 27-53% in cereal farms, 10-17% in hog farms and 13-22% in dairy farms, the range depending on the support region. Finally, Vehkasalo (1999) argued that without agri-environmental support 20% of farmers in Southern Finland (where the level of support was highest in the first program period 1995-1999) and 10% of farmers in the other parts of country would have ceased unprofitable production without agri-environmental support.

Following Vehkasalo (1999) we assume here that without agri-environmental support 20% of farmers in Southern Finland and 10% of farmers in the other parts of country would have ceased unprofitable production without agri-environmental support. This means that roughly 10% of cultivated land would have exited in the short run in the absence of agri-environmental support[11]. Table 8 shows the difference between Baseline

[10] That abatement subsidies impact this way is a well-known result in environmental economics. For instance, Baumol and Oates (1988) demonstrated that a subsidy to reduce emissions in an industry leads to higher emissions than original total emissions even though every firm has reduced emissions.

[11] Farmers' actual behaviour may differ from what they stated in the survey. Moreover, at least part of cultivated land could be rented or bought by those farmers who

total nitrogen and phosphorus runoff and the case where entry-exit margin impact of agri-environmental support is taken into account.

Table 8. **Total nitrogen and phosphorus runoff (tons) under baseline and without entry-exit margin impact of agri-environmental support**

Land allocation	1995-1999		2000-2006		2007	
	N	P	N	P	N	P
Baseline	34509	2711	36638	2694	38035	2571
Without entry-exit margin impact	31058	2440	32974	2425	34232	2314
Difference	-3451	-271	-3664	-269	-3804	-257

The impact of the excess compensation shows up on average 3,639 tons of nitrogen and 266 tons of phosphorus. It is useful to relate the negative impact of excess compensation to Finland's nutrient load reduction targets -1,500 tons of nitrogen and 120 tons of phosphorus in the Baltic Sea Action Plan. It shows that the negative impact of the excess compensation is large, clearly more than the Finland's nitrogen target and twice the phosphorus target. Thus, we witness here a serious unintended consequence of the too generous agri-environmental support payment.

6. Cost-benefit analysis of water protection policy in the Finnish agri-environmental programme

We end the analysis by asking whether the benefits from water quality policy in the Finnish agri-environmental program exceed the costs, or not. Costs of the policy is the overall amount of annual support payments to farmers targeted to water quality, while the benefits are given by the reduced nutrient runoff damages. Reductions in nutrient loads reduce damages both in inland waters and in the Baltic Sea. As the main goal is to improve the state of the Baltic Sea, we express phosphorus loads as nitrogen equivalent using the Redfield ratio 7.2. The Redfield ratio describes the optimum N/P ratio for the growth of phytoplankton, relevant for algal growth in sea waters. The marginal damage from nitrogen equivalents is assumed constant, so that the damage function is given by

$$d\left(Z^i\right) = R_n\left(N_i + 7.2P_i\right) \tag{11}$$

continue production and thus the production impact would be smaller than anticipated. Nevertheless, our assumption of 10% increase is well in line with Pufall and Weiss (2009) who found that farmers participating in the respective program in Germany have increased the land area under cultivation by 7.7%. The slightly higher impact in Finland is explained by the fact that the profitability of the Finnish agriculture without support is much lower than in Germany due to severe northern growing conditions.

where R_n is the constant social marginal damage. Drawing on Gren (2001), the willingness to pay for nutrient load reduction in the Baltic Sea is set to be R_n = € 6.70/kg of N equivalent. Hence, this estimate provides social value of reductions in nutrient runoff. Given that the share of the Finnish agriculture in water pollution of the Baltic Sea is small (5-6%), we employ constant marginal damage in our calculations. This is a simplification and reflects merely the water quality impacts in the open seas.

As regards social costs of nutrient runoff reduction we use the budg*et al*located to water protection measures in the agri-environmental program as a primary measure of social costs of nutrient runoff reduction (see Boardman *et al*., 2011, 99-110 for discussion). The reported program outlays represent total agri-environmental support (basic, additional and special measures) for each year. Almost all types of basic and additional measures are directly or indirectly linked to water protection.[12]

Also, we report a more developed social net benefit estimate by including the policy related transaction costs (PRTCs) to the social costs. Our estimate of policy related transaction costs of agri-environmental support is based on Ollikainen *et al*. (2008) who estimated that the PRTCs of Basic measure support (including fertilizer use constraints and buffer strips) are 1.5% of the total transfer. Finally, the most comprehensive social net-benefit estimate takes also into account the so-called marginal cost of taxation MCT (also called marginal cost of public funds) as a measure of economic welfare losses due to raising government revenue with distortionary taxes (such as labor taxes). We employ 10% of the total transfer as our estimate of marginal cost of taxation.

Finally, for the purposes of comparison in Table 10 we provide a hypothetical case assuming that only compliance costs were compensated to farmers. A crude estimation of compliance costs of implementing the program is measured as foregone profits when a farmer complies with fertilizer use constraints and buffer strip establishment. Thus, foregone profits show the minimum level required for compensation payment.

Table 9 provides the three estimates of the social net benefits for the Finnish Agri-Environmental Scheme.

In Table 9 the reduction in nitrogen equivalents has been determined so that we take the difference between the Baseline and Private optimum in Table 6 and apply Redfield ratio in order to derive N-eq nutrient runoff reduction. Table 9 reveals that the social net-benefit of the programme is negative in every program period under all three net-benefit measures. This

[12] The basic measures include among others fertilizer use reduction and buffer strip establishment, additional measures include more accurate nitrogen application, winter cover and reduced tillage, nutrient balance reporting and manure application during growing season. The special measures include e.g. establishment of buffer zones and constructed wetlands, treatment of runoff waters and organic production.

Table 9. **Social net-benefits of the agri-environmental program**

	1995-1999	2000-2006	2007
N-eq reduction, tons	2038	-1038	2227
Program outlays, million €	229.6	233.1	276.0
Value of damage reduction, million €	13.7	-7.0	14.9
Net benefit, million €	-215.9	-240.1	-261.1
Transaction costs (TC), million €	3.4	3.4	4.0
Net benefit - TCs, million €	-219.3	-243.5	-265.1
Net benefit – TCs - MCT, million €	-242.2	-266.8	-292.7

clearly refers to overcompensation of farmers' compliance costs, that is, part of the environmental support payments seems to entail farm income support.

This can also be verified in Table 10, where the direct compliance costs (i.e. short-run profit foregone) are estimated. Thus, Table 10 illustrates what the net-benefits would have been if the Progamme would only have compensated compliance costs. The estimated compliance costs vary over time as a function of crop prices and cultivation costs. These direct costs are quite low relative to the environmental payments, because they underestimate the long-run cost burden on farmers. However, even if these costs were doubled or tripled they would still remain relatively small in comparison to program outlays.

Table 10. **Social net-benefits of efficient agri-environmental program**

	1995-1999	2000-2006	2007
N-eq reduction, tons	7441	3191	7882
Profit foregone, million €	-21.8	-4.4	-15.0
Net benefits, million €	28.0	17.0	37.8

In this case the reduction in nitrogen equivalents has been determined so that we take the difference between the Private optimum in Table 6 and Without entry-exit margin impact in Table 8 and apply Redfield ratio in order to derive N-eq nutrient runoff reduction. Due to zero entry of land due to Progamme (because of no overcompensation of compliance costs this time) the reduction of nitrogen equivalents are now much larger and this makes the social net-benefits of the program positive.

7. A note on the applicability of optimization-based simulation in policy evaluation

We applied an optimization-based simulation model to assess the impacts of agri-environmental policies on nutrient loads. The approach we chose is relatively uncommon in the policy assessment literature, which typically applies sophisticated econometric models. Just like the econometric policy analysis, we rooted our work on statics and on the application of the profit maximization hypothesis. Our approach differs from the econometric models in that instead of using statistically identified relationships, we rooted our approach on calibrated yield response functions. As is well known, both approaches have been utilized much in various (*ex ante*) policy impact and policy design analyses. It is our understanding that under ideal conditions and using comprehensive frameworks our approach outlined in this paper and using econometric techniques to policy assessment should lead to similar outcomes.

The basic reason for resorting to the optimization-based simulation model in this paper was the fact that almost all farmers in Finland participate in the agri-environmental program, which makes it difficult to establish the control group. Using calibrated response functions as the core of the farm profit function allowed us to determine farmers' actions in the absence of the policy (in our case, especially the free private fertilization intensity), which would have been otherwise difficult to develop. While we acknowledge the importance and sophistication of statistical policy analysis, we can see also other cases where our optimization-based simulation approach may be helpful or may complement the statistical policy assessment.

First, data on the actual behaviour of farmers may not always be available, or it may not cover well the aspects of agricultural production in focus. In this case, the optimization-based simulation model provides one alternative to proceed. Second, the data is available but it may not always be reliable due to so many moral hazard aspects present in agriculture. For instance, fertilization levels or yields may be underreported in statistics. Again, our approach provides a possibility to check the impacts of policies from an angle that is not dependent on false reporting. Third, if policy covers practically all farmers (like in our case), the optimization-based simulation model helps to create the required counterfactual. Finally, suppose that the authorities are fine-tuning the design of policies during rather short time intervals. An optimization-based simulation model can be quickly adjusted for this purpose to provide also *ex ante* analysis of policy reforms.

The model used in the paper was simple, because our focus was on the assessment of the agri-environmental program impacts on the aggregate nutrient loads, so that we pay no attention to the specific impacts of the policy. Increasing complexity of the model to explain the impacts of policy program in more detail would, however, be a relatively straightforward task.

8. Conclusions and policy implications

Counterfactual analysis belongs to the basic tools for policy evaluation in economics. It typically requires data on the treatment group and the control group. This requirement may not always be met. We focused on a case where the *ex-post* analysis of agri-environmental policy becomes complicated by the fact that the whole population is subject to the agri-environmental policy. For this case we suggested a formal approach that is based on the use of profit maximization hypothesis. We created the control group allowing the farmers to maximize their profits freely. The treatment group in turn was defined by the (constrained) behaviour under agri-environmental policy. We applied our model to agricultural water protection policy of the Finnish Agri-Environmental Programme, which aims at reducing nutrient runoff from arable lands to the Baltic Sea. Counterfactual analysis allowed us to examine both the unit effectiveness of the measures included in the Programme and its preventive impact.

We find that the Finnish agri-environmental programme has failed to achieve its goals: nitrogen loads have increased and phosphorus loads have decreased only slightly. Our counterfactuals help to trace out the mechanisms leading to this failure. First, we find that Common Agricultural Policy has modified the incentives provided by the Finnish agri-environmental program. Crop area payments and the current single farm payment invite more land in cultivation. Second, the aim of area payments is to let relative prices to guide agricultural production. Relative prices favour land allocation to more fertilizer intensive land use forms (leading increased use of nitrogen). Thus, general development in both extensive and intensive margins tends to increase nutrient loads. Third, environmental support is an area payment. Due to overcompensation of farmer's compliance costs, it also invites more cultivated land to agriculture by keeping low productivity land (land with zero or slightly negative profits) in cultivation. Thus, due to overcompensation the policy instrument works against its water protection aims. These three impacts were not taken into account by the environmental authorities when the agri-environmental programme was launched.

Our analysis also shows that the means included in the programme were in principle effective especially for phosphorus. Unlike constraint on nitrogen, the phosphorus fertilization constraint is independent of crops. Logically, it has led to gradual decrease in soil phosphorus content and reduction in dissolved phosphorus. Therefore, we witness here anything else than a policy failure, a failure to design policies capable to accounting for changes in farmers behaviour as a response to new incentives. The policy failure is complemented by the frustration of the farmers who have taken an effort to reduce loads but do not receive any reward in the form of reduced agricultural loads. This is not to say that the Finnish Agri-Environmental Programme would have failed entirely. We demonstrated that the loads of both nutrients would have been much higher if no policy would have been implemented. The social cost-benefit analysis of the program showed, nevertheless, strongly negative net-benefits. Thus, there is a lot of scope for improving the agri-environmental water protection policies. But as our figures reveal, there is also much promise.

We believe the lesson of our analysis is relevant to all countries trying to reduce nutrient loads. In addition to finding the best means for the programme, both behavioural responses and counteraffecting agricultural policies must be accounted for when designing water protection programmes.

Finally, the counterfactual analysis developed in this paper lead to an interesting analysis. It was capable to reveal many important mechanisms affecting the outcomes of policies. It is a much recommended tool for the analysis of environmental policies.

References

Baumol W. and Oates W. (1988) *The theory of environmental policy*, Second edition, Cambridge University Press, Cambridge, USA, 299 p.

Boardman A., Greenberg D., Vining A., and Weimer D. (2011) *Cost-Benefit Analysis: concepts and practice*, Fourth edition, Pearson-Prentice Hall, Boston, USA, 560 p.

Bäckman S. T., Vermeulen S. and Taavitsainen V.-M. (1997) Long-term fertilizer field trials: comparison of three mathematical response models, *Agricultural and Food Science in Finland* 6(2), 151-160.

Chang-Jin K., Morley J. and Piger J. (2008) Bayesian counterfactual analysis of the sources of the great moderation, *Journal of Applied Econometrics 23*, 173-191.

Dawid A. (2000) Causal inference without counterfactuals, *Journal of the American Statistical Association 95*(450), 407-448.

Galles D. and Pearl J. (1998) An axiomatic characterization of causal counterfactuals, *Foundations of Science 3*(1), 151-182.

Gren M. (2001) International versus national action against pollution at the Baltic Sea, *Environmental and Resource Economics 20*(1), 41-59.

Guyomard H., Lankoski J. and Ollikainen M. (2009) Impacts of agricultural policies on cropland prices, *Food Economics - Acta Agriculturae Scandinavica Section C* 6(2), 88-98.

Hanley N. and Oglethorpe D. (1999) Emerging policies on externalities from agriculture: an analysis for the European Union, *American Journal of Agricultural Economics 81*, 1222-1227.

Hanley N., Whitby M. and Simpson I. (1999) Assessing the success of agri-environmental policy in the UK, *Land Use Policy 16*(2), 67-80.

Heckman J. and Vytlacil E. (2007) Econometric Evaluation of Social Programs, in: *Handbook of Econometrics*, Heckman J. and Leamer E. (eds), Volume 6. Elsevier, Amsterdam, The Netherlands, 1056 p.

Heckman J. (2010) Building bridges between structural and program evaluation approaches to evaluating policy, *Journal of Economic Literature 48*(2), 356-398.

HELCOM (2010) *Atlas of the Baltic Sea*, Helsinki Commission, Baltic Marine Environment Protection Commission, Helsinki, Finland, 192 p.

Iho A. (2010) Spatially optimal steady-state phosphorus policies in crop production, *European Review of Agricultural Economics 37*(2),187-208.

Kleijn D. and Sutherland W. (2003) How effective are European agrienvironmental schemes in conserving and promoting biodiversity, *Journal of Applied Ecology 40*, 947-696.

Koikkalainen K. and Lankoski J. (2004) Maatalouden ympäristötuen taloudellinen merkitys tukialueittain ja tuotantosuunnittain vuosina 2000 ja 2001. *Maatalouden ympäristötuen-seuranta MYTVAS 2: Osahankkeiden 2-7 väliraportit 2000-2003*, In: Eila Turtola ja Riitta Lemola (toim.). Maa- ja elintarviketalous 59, 140-157 (in English: Follow-up of the effectiveness of the agri-environmental programme in Finland, Results of sub-projects in 2000-2003).

Kluve J. (2004) On the role of counterfactuals in inferring causal effects, *Foundations of Science 9*(1), 65-101.

Lankoski J., Ollikainen M. and Uusitalo P. (2006) No-till technology: benefits to farmers and the environment? Theoretical analysis and application to Finnish agriculture, *European Review of Agricultural Economics 33*(2), 193-221.

Laukkanen M. and Nauges C. (2012) Evaluating greening farm policies: A structural model for assessing agri-environmental subsidies. The Government Institute for Economic Research Finland (VATT), Working Paper N° 40.

Lehtonen H. (2001) Principles, structure and application of dynamic regional sector model of Finnish agriculture. PhD thesis, Systems Analysis Laboratory, Helsinki University of Technology. Agrifood Research Finland, Economic Research (MTTL). Publ. *98*. 264 p.

Lichtenberg E. (2002) Agriculture and the environment, in: *Handbook of Agricultural Economics*. Vol 2A: Agriculture and its external linkages, Gardner B. and Rausser G. (eds), Amsterdam: North Holland, 1249-1313.

MMM (2000) *Horisontaalinen maaseudun kehittämisohjelma Manner-Suoma koskevat EU: n yhteisen maatalouspolitiikan liitännäistoimenpiteet vuosille 2000-2006*, Helsinki, Maa- ja metsätalousministeriö, 229 p.

MMM (2004) *Horisontaalisen maaseudun kehittämisohjelman väliarviointi: Manner-Suomi*, Helsinki, Maa- ja metsätalousministeriö 1/2004, 273 p.

Myyrä S., Ketola E., Yli-Halla M. and Pietola K. (2005) Land improvements under land tenure insecurity: the case of pH and phosphate in Finland, *Land Economics 81*(4), 557-569.

OECD (2008) *Environmental performance of agriculture in OECD countries since 1990*, Paris, OECD, France, 210 p.

Ollikainen M., Lankoski J. and Nuutinen S. (2008) Policy-related transaction costs of agricultural policies in Finland, *Agricultural and Food Science 17*(3), 193-209.

Pietola L., Tanni R. and Elonen P. (1999) Responses of yield and N use of spring sown crops to N fertilization, with special reference to the use of plant growth regulators, *Agricultural and Food Science in Finland 8*(4-5), 423-440.

Primdahl J., Peco B., Schramek J., Andersen E. and Oñate J. (2003) Environmental effects of agri-environmental schemes in Western Europe, *Journal of Environmental Management 67*(2), 129-138.

Pufahl A. and Weiss C. (2009) Evaluating the effects of farm programmes: results from propensity score matching, *European Review of Agricultural Economics 36*(1), 79-101.

Puustinen M., Turtola E., Kukkonen M., Koskiaho J., Linjama J., Niinioja R. and Tattari S. (2010) VIHMA—a tool for allocation of measures to control erosion and nutrient loading from Finnish agricultural catchments, *Agriculture, Ecosystems and Environment 138*(3-4), 306-317.

Saarela I., Järvi A., Hakkola H. and Rinne K. (1995) Fosforilannoituksen porraskokeet 1977-1994. Vuosittain annetun fosforimäärän vaikutus maan viljavuuteen ja peltokasvien satoon monivuotisissa kenttäkokeissa, Maatalouden tutkimuskeskus, kasvintuotannon tutkimuslaitos, maanviljelyskemian ja -fysiikan tutkimusala. (In English: Phosphorus fertilizer trials, 1977-1994: effects of the rate of annual phosphorus application on soil fertility and yields of field crops in long-term field experiments), 94 p.

Schnitkey G. and Miranda M. (1993) The impact of pollution controls on livestock-crop producers, *Journal of Agricultural and Resource Economics 18*(1), 25-36.

Siikamäki J. (1996) Finnish agri-environmental programme in practice - Participation and Farm-level Impacts in 1995. p. 83-98. In: First Experiences of Finland in the *CAP*, Kettunen L. (ed.), MTTL Research Publications N° *81*, 157 p.

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

Simmelsgaard S. E. and Djurhuus J. (1998) An empirical model for estimating nitrate leaching as affected by crop type and the long-term fertilizer rate, *Soil Use and Management 14*(1), 37-43.

Uusi-Kämppä J. and Yläranta T. (1992) Reduction of sediment, phosphorus and nitrogen transport on vegetated buffer strips, *Agricultural Science in Finland 1*(6), 569-575.

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

Uusi-Kämppä J. and Kilpinen M. (2000) *Suojakaistat ravinnekuormituksen vähentäjänä*. Maatalouden tutkimuskeskuksen julkaisuja, Sarja A n° *83*, 49 p.

Uusitalo R. (2004) *Potential bioavailability of particulate phosphorus in runoff from arable clayey soils*. Agrifood Research Reports 53, Doctoral Dissertation, MTT Agrifood Research Finland, Jokioinen, MTT Agrifood Research Finland.

Uusitalo R. and Jansson H. (2002) Dissolved reactive phosphorus in runoff assessed by soil extraction with an acetate buffer, *Agricultural and Food Science in Finland 11*(4), 343-353.

Vehkasalo V. (1999) Ympäristötuen yhteiskunnallinen kannattavuus. In: *Maatalouden ympäristöohjelma 1995-1999:n taloudellinen analyysi* : ympäristötukijärjestelmä ja tulevaisuus - tutkimuksen loppuraportti. MTT Agrifood Research Finland, Publications *90*, 42-77.

YM Ympäristöministeriö (2000) *Vesien suojelun toimenpideohjelma vuoteen 2005*, Helsinki, Suomen ympäristö 402, Ympäristönsuojelu, s.98, 47 p.

Appendix A: Detailed result tables

Table A1. **Total nitrogen and phosphorus runoff (tons) under the privately optimal fertilizer use in 1995-1999, 2000-2006, and 2007**

Crop	**1995-1999**		**2000-2006**		**2007**	
	N	**P**	**N**	**P**	**N**	**P**
Wheat	2812	125	3675	195	4300	209
Barley	13033	728	11435	717	12110	731
Oats	7447	473	7162	490	7096	449
Rape	1913	76	1784	91	2364	110
Silage	2708	361	3640	437	4598	492
Hay	1670	243	913	130	842	110
Fallow	2088	173	2396	191	2162	196
Other	5473	376	4787	347	5389	370
Total	**37143**	**2554**	**35792**	**2598**	**38861**	**2666**

Table A2. **Nitrogen, dissolved phosphorus (DRP) and particulate phosphorus (PP) runoff, kg/ha, under privately optimal fertilizer use intensity in 1995-1999, 2000-2006, and 2007**

Crop	**1995-1999**			**2000-2006**			**2007**		
	N	**DRP**	**PP**	**N**	**DRP**	**PP**	**N**	**DRP**	**PP**
Wheat	34.0	0.53	0.98	27.7	0.49	0.95	29.8	0.49	0.96
Barley	26.1	0.50	0.96	23.2	0.50	0.96	24.8	0.52	0.98
Oats	22.5	0.49	0.94	20.4	0.47	0.93	22.3	0.47	0.94
Rape	34.8	0.46	0.92	28.6	0.49	0.96	31.8	0.50	0.97
Silage	9.4	0.80	0.45	10.3	0.79	0.45	11.7	0.80	0.45
Hay	8.3	0.76	0.44	8.2	0.74	0.43	9.1	0.75	0.44

Table A3. **Parameter values of the application**

Parameter	**Symbol**	**Value**
Mitscherlich response function: Barley	μ	5218
	σ	0.8280
	ν	0.0168
Mitscherlich response function: Wheat	μ	4956
	σ	0.7624
	ν	0.0105
Mitscherlich response function: Oats	μ	4760
	σ	0.7075
	ν	0.0197
Quadratic response function: Rape	A	1034
	χ	12.57
	γ	-0.0260
Quadratic response function: Silage (dry matter content 23%)	A	1183
	χ	24.24
	γ	-0.0394
Quadratic response function: Hay	A	1374
	χ	33.8
	γ	-0.078
Soil P:	Φ	
1995		12.6
2001		11.6
2007		10.6
Runoff volume	Ψ	270
Erosion	ζ_i	250-800
Nitrogen runoff at average nitrogen use	Φ_i	10.2-22.0
Reductive effect of grass buffer strips	α	0.3
	β	1.3
	κ	0.3

Table A4. **Total nutrients in chemical fertilizer and livestock manure, kg/ha of arable land. (Source: MYTVAS 2010)**

Year	N, chemical fertilizer, kg/ha	P, chemical fertilizer, kg/ha	N, manure, kg/ha	P, manure, kg/ha	Share of manure% in total N	Share of manure% in total P
1995-1999	102	20	49.5	9.3	33	32
2000-2006	83	10.8	48.3	8.7	37	45
2007	74	7.9	49.6	8.6	40	52