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

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Paper prepared for presentation at the EAAE 2011 Congress
Change and Uncertainty
Challenges for Agriculture,
Food and Natural Resources

August 30 to September 2, 2011
ETH Zurich, Zurich, Switzerland

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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. It is important to assess the success of these voluntary agri-environmental programs. As 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 alternative, hypothetical policies or counterfactual analysis. *Counterfactual* analysis belong to the basic tools for policy evaluation in economics. Counterfactual analysis answers the question: what would have happened if…? Counterfactual analysis builds on a non-observable case, called counterfactual. A comparison of the counterfactual with the actual case sheds light to the critical factors explaining the impacts of policy.

We develop a theoretical frame and derive the counterfactuals for empirical analysis from it. We apply our theoretical frame to agricultural water protection policy which aims at reducing nutrient runoff from arable lands to the Baltic Sea. 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 program would have been implemented at all in 1995 but the rest of the CAP would be in force.

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 x_0 x_A and x_B 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 ($\overline{m}$), the CAP area payment (*a*) and the environmental support payment per hectare (*b*), that is, $x^A = \{\bar{l}^A, \overline{m}, a, b\}$. 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, $x^B = \{s, E\}$.

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 $y = f_i(l_i)$ with $f_i'(l_i) > 0$ but $f_l''(l_i) < 0$. 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 farmers choices under each policy regime.

Under no policy intervention (denoted by supersrcpt 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_{-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 $\pi_i^A = (1-\overline{m})[p_i f_i(l_i) - cl_i] + a + b$ subject to $l_i^A \leq \bar{l}_i$. At the optimal solution output price and fertlizer cost no longer impact fertilizer intensity, because the fertilizer application constraint is binding ($l_i^A = \bar{l}_i$); also the buffer strip is mandated $m^A = \overline{m}^A$. Using these two mandatory figures, the overall amount of fertilizer applied to each hectare is $l_i^A = (1-\overline{m})\bar{l}_i$. 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:

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

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

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

Recall, our aim is to assess the performance of the new agri-environmental policy *A*. Let $\overline{G}^A$ 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, $G^A - \overline{G}^A$, can be any sign and 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 formalize these counterfactuals (*CF*):

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

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

Taking the difference $G^A - CF_1$ and $G^A - CF_2$ allows us 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 fashion 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,

$$y_i = \mu_i (1 - \sigma_i e^{-\nu_i N_i}) \tag{6}$$

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

$$y_i = A_i + \chi_i N_i + \gamma_i N_i^2 \qquad (7)$$

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).

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.

$$\pi^i = p_i \mu_i (1 - \sigma_i e^{-\nu_i N_i}) - c_i N_i \qquad (8a)$$

$$\pi^i = p_i (A_i + \chi_i N_i + \gamma_i N_i^{\,2}) - c_i N_i \qquad (8b)$$

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). 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 modeling of nutrient runoff follows Lankoski *et al*. (2006) who modeled nitrogen and phosphorus runoff on the basis of Finnish data. In the case of phosphorus runoff two forms are distinguished: (i) dissolved reactive phosphorus (DRP) and (ii) particulate phosphorus (PP). As already noted, farmers use a compound fertilizer (NPK) and because 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. The concentration of dissolved phosphorus in surface runoff is found to depend linearly on the easily soluble soil P, and the runoff of particulate phosphorus depends on the rate of soil erosion and the P content of eroded soil material.

The following nitrogen runoff function (Simmelsgaard 1991) is employed,

$$Z_N^i = \phi_i \exp(b_0 + bN_i), \qquad (9)$$

where Z_N^i = nitrogen runoff at fertilizer intensity level N_i, kg/ha, ϕ_i = nitrogen runoff at average nitrogen use, $b_0 < 0$ and $b > 0$ are constants and N_i = nitrogen fertilization in relation to the normal fertilizer intensity for the crop, $0.5 \le N \le 1.5$. This runoff function represents nitrogen runoff generated by a nitrogen application rate of N_i per hectare and the parameter ϕ_i reflects differences in crops.

In the case of phosphorus, both dissolved and particulate runoff is modeled. Drawing on Finnish experiments (e.g. 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

$$Z^{i}_{DRP} = \varpi_i[\psi_i(0.021(\Phi + 0.01 * P_i) - 0.015]/100 \quad (10a)$$

$$Z^{i}_{PP} = \Delta_i[\zeta_i\{250\ln(\Phi + 0.01 * P_i) - 150\}]*10^{-6} \quad (10b)$$

where 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 and field slope based differences in the runoff of dissolved and the potentially bioavailable particulate phosphorus are captured by parameters ϖ_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 2003 and Myyrä et al. 2005).

4. Results

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.

Table 1. *Land use (ha) in 1994, 1995, 2001, and 2007 (Yearbook of Farm Statistics).*

Land use	**1994**	**1995**	**2001**	**2007**
Wheat	77 600	88 100	115 400	167 900
Barley	505 700	516 200	547 200	550 100
Oats	334 300	329 300	422 700	361 500
Rape	67 200	85 300	73 100	90 200
Hay	257 900	287 100	157 500	103 100
Silage	268 400	300 900	380 900	438 100
Fallow	505 100	223 200	201 900	121 200
Other	285 700	316 300	293 200	423 200
Total	2 301 900	2 146 400	2 191 900	2 255 300

Table 1 shows that during the agri-environmental program periods (1995 – 2007) the total cultivated land increased by 2.1% from 1995 to 2001 and 5.1% from 1995 to 2007 in Finland. Also land allocation between different crops has changed much. Land allocation to wheat cultivation has increased from 4.8% to 8.6% between 1995 and 2007. The share of barley, oats and rape has remained quite stable while land allocated to hay has decreased from 15.7% to 5.3% and land allocated to silage increased from 16.4% to 22.6%. Thus, there has been a clear shift in land use towards more fertilizer intensive crops wheat and silage.

Table 2 presents the actual fertilization constraints and the fertilizer intensity under the hypothetical case, in which farmers optimize on the basis of market prices only and do not participate in the agri-environmental programme (labeled as “N private”).

Table 2. *Optimal nitrogen use intensity and nitrogen application constraint in agri-environmental program in 1995, 2001, and 2007.*

Crop	1995		2001		2007	
	N private	**N constraint**	**N private**	**N constraint**	**N private**	**N constraint**
Wheat	155.3	100	140.4	100	158.8	120
Barley	122.1	90	112.1	90	123.3	100
Oats	97.9	90	92.2	90	94.2	100
Rape	156.3	100	147.3	100	137.2	110
Silage	161.6	180	199.3	180	255.6	240
Hay	128.0	90	133.4	90	176.6	100

Table 2 shows that for most focused crops the nitrogen application constraint has been binding throughout program years. For wheat, rape and hay the economically optimal application rate has been clearly higher than the constraint, while in the case of oats farmers’ economic optimum and nitrogen use constraint are sufficiently close to each other. For silage the constraint was not binding in the first agri-environmental program period (1995-1999) but it became binding during the following program periods (2000-2006 and 2007-2013).

Table 3 presents the estimated average per ha nutrient runoff under the fertilizer application constraints reported in Table 2. We calibrated our N ja P runoff functions to reflect the runoff figures of the Finnish VIHMA model (see Puustinen et al. 2010). Calculations in Table 3 account for impact of buffer strips in reducing runoff from field parcels. Moreover, we assume that manure application causes the same runoff propensity as the use of chemical fertilizers.

Table 3. *Nitrogen, dissolved phosphorus (DRP) and particulate phosphorus (PP) runoff, kg/ha, under constrained fertilizer use intensity in 1995, 2001, and 2007.*

Crop	**1995**			**2001**			**2007**		
	N	**DRP**	**PP**	**N**	**DRP**	**PP**	**N**	**DRP**	**PP**
Wheat	14.0	0.376	0.479	14.0	0.347	0.462	16.1	0.319	0.444
Barley	13.1	0.376	0.479	13.1	0.347	0.462	14.0	0.319	0.444
Oats	13.1	0.376	0.479	13.1	0.343	0.459	13.5	0.312	0.438
Rape	14.0	0.356	0.467	14.0	0.347	0.462	15.0	0.319	0.444
Silage	7.3	0.585	0.175	8.2	0.534	0.168	12.4	0.506	0.164
Hay	4.5	0.526	0.166	4.5	0.481	0.160	4.8	0.447	0.154
Fallow	5.4	0.461	0.280	5.4	0.407	0.280	5.4	0.373	0.280
Other	9.7	0.671	0.182	10.4	0.636	0.182	12.6	0.602	0.182

The most alarming feature in Table 3 is that due to the relaxation of the nitrogen application constraints, the last phase of the program witnesses the highest per hectare nitrogen runoff. In contrast to this, the per hectare runoff of particulate and dissolved phosphorus has diminished steadily over time due to the decrease in soil phosphorus. Hence, the total load of phosphorus per hectare has decreased.

Table 4 combines the observed land allocation between the crops and the estimated per hectare average nutrient runoff to produce total nitrogen and phosphorus runoff in tons in 1995, 2001, and 2007 under the Finnish Agri-Environmental Programme.

Table 4. *Total nitrogen and phosphorus runoff (tons) under constrained fertilizer use in 1995, 2001, and 2007.*

	1995		**2001**		**2007**	
Crop	**N**	**P**	**N**	**P**	**N**	**P**
Wheat	1894	97	2481	121	4151	166
Barley	10347	568	10969	572	11825	544
Oats	6601	362	8473	438	7461	352
Rape	1834	91	1571	76	2079	89
Silage	3391	257	4883	301	8544	332
Hay	1962	224	1076	114	756	70
Fallow	1129	155	1021	130	613	74
Other	4693	303	4705	271	8184	376
Total	**31 851**	**2056**	**35 180**	**2024**	**43 613**	**2003**

The time path of the nitrogen load shows that despite all efforts, nitrogen runoff has increased during all three phases of the program. The shift of arable land to more fertilizer intensive crops, such as wheat and silage, increased nitrogen runoff during the second phase of the agri-environmental program. This tendency is re-enforced by the relaxed nitrogen constraint during the third phase. The development of phosphorus runoff follows a different path since the gradual decrease of soil phosphorus content decreases phosphorus runoff in the second and third phase of the program. Thus, nutrient loading evolves to opposite directions; the reason is that phosphorus application is roughly constant across crops, while nitrogen application varies considerably between crops.

In Table 5 we examine what would have happened to nutrient loads if agricultural land use would not have changed over the years but would have stayed 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 exactly same as in 1994 or in 1995. Taking the difference between this and the actual loads under the current policy regime, which is called the Baseline indicates the importance of controlling both entry-exit (total amount of arable land) and extensive (land allocation) margins in addition to intensive margin (fertilizer use intensity and buffer strips).

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	**1995**		**2001**		**2007**	
allocation	**N**	**P**	**N**	**P**	**N**	**P**
Baseline	31 851	2056	35 180	2024	43 613	2003
Fixed 1995	31 851	2056	32 398	1938	36 519	1833
Fixed 1994	29 056	1982	29 499	1864	32 915	1764

Table 5 reveals that under either 1995 or 1994 land allocation total nitrogen load would have increased much less and phosphorus load decrease much more than they actually did. Under 1995 land allocation, decrease in phosphorus runoff by 2007 would have been 223 tons and under 1994 land allocation 218 tons, making 11.0% and 10.8% reduction of nutrient loads. Given that the phosphorus fertilization limit has been the same over all three periods, these figures indicate the true impact of phosphorus limits on loads that is purified from the changes in land allocation. Due to changes in land allocation, the actual decrease in phosphorus loads was 53 tons only, representing 2.6% reduction. The difference of these two figures, 165–170 tons, represents simply the increase in total phosphorus load due to the increased land area in cultivation (note that changing land allocation between crops has only negligible impact on total phosphorus runoff).

During the years 1995 – 2007 nitrogen loads would have increased by 4668 tons (14.7%) under 1995 land allocation and by 3859 tons (13.3%) under 1994 land allocation. Contrast this to the actual increase in nitrogen loads that was 11 762 tons (36.9%). The increase in actual loads is explained by two factors: nitrogen fertilization constraints were relaxed in the third programme period (year 2007); and the amount of cultivated land has increased and more land is allocated to more nitrogen intensive crops. The relaxed nitrogen fertilization constraints increased nitrogen loads by 5753 tons in the baseline, representing 48.9% of the total load increase, relative to a situation where constraints would not have been relaxed. Consequently we can conclude that the impact of the increase in the amount of cultivated land and land allocation change towards more nitrogen intensive crops was 6009 tons, representing 51.1% of the load increase.[1]

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. This is our second counterfactul that is defined by equation (5). Table 6 convays information on nutrient loads under both cases. 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 6. *Total nitrogen and phosphorus runoff (tons) under constrained and free private fertilizer use in 1995, 2001, and 2007.*

	1995		**2001**		**2007**	
	N	**P**	**N**	**P**	**N**	**P**
Constraint	31 851	2056	35 180	2024	43 613	2003
Private	35 175	1948	37 213	1894	46 276	1913
Difference	-3324	+108	-2033	+129	-2663	+90

As Table 6 reveals, the preventive effect of the Finnish Agri-Environmental Programme has been on average 2673 tons of nitrogen while in the case of phosphorus the program has even increased runoff by 109 tons on average relative to private optimum. Thus, in the case of phosphorus entry-exit margin impact, that is, increased amount of land in cultivation under the program and thus increased runoff outweighs the reductive effect of input use constraints.

[1] The small difference in the total nitrogen loads in 1995 and 2001 even though land allocation and nitrogen fertilization constraints were the same for both periods. This difference results from the fact that nitrogen constraint was not binding for silage in 1995 (optimal fertilizer intensity was less than the constraint), while it became binding in the second phase of the programme.

Combining Table 6 with Table 5 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 outweighted 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.

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

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

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.

As regards social costs of nutrient runoff reduction we use the budget allocated to water protection measures in the agri-environmental program as a primary measure of social costs of nutrient runoff reduction. 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 (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.

Table 7. *Social net-benefits of the agri-environmental program.*

	1995	**2001**	**2007**
Social net benefits of the agri-environmental program			
N-eq reduction, tons	2544	1104	2013
Program outlays, million €	229.6	233.1	276.0
Value of damage reduction, million €	17.0	7.4	13.5
Net benefit, million €	-212.5	-225.7	-262.5
Transaction costs (TC), million €	3.4	3.4	4.0
Net benefit - TCs, million €	-215.9	-229.1	-266.5
Net benefit – TCs - MCT, million €	-238.8	-252.4	-294.1

In Table 7 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 7 reveals that the social net benefit of the programme is negative in every program period under all three net benefit measures. This clearly refers to overcompensation of farmers' compliance costs, that is, part of the environmental support payments seem to entail farm income support.

5. Conclusions

We developed a theoretical framework using the interlinkages between the behavior of agents and the response of environmental systems to the economic decisions. 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. 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 tend 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. The social cost-benefit analysis of the program showed strongly negative net benefits.

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