DOES SCALE MATTER? – COST-EFFECTIVENESS OF AGRICULTURAL NUTRIENT ABATEMENT WHEN TARGET LEVEL VARIES

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Abstract

Agricultural production is facing high requirements on nutrient runoff reduction. Furthermore, the reductions should by done efficiently. For instance, the European Water Framework Directive calls for cost-effectiveness from schemes of measures to fulfill the target of good water quality in European river basins. In this paper we analyse the implications of target level variation on efficiency properties of agricutural phosphorus abatement. We analyse the robustness of cost-effectiveness as the scheme of measures is adopted from another, identical river basin with different target level on total phosphorus abatement. We find that even between homogeneous regions the cost-effective scheme of measures is unique for all target levels of reduction, and that the costs of adoting cost-effective allocations out of scale are high.

KEY WORDS: Cost-effectiveness, phosphorus abatement, buffer strips, wetlands, fertilizer use, Water Framework Directive

JEL: Q18, Q25

Introduction

Agricultural environmental policy is typically driven by political agendas with explicit reduction targets for agricultural pollution. For instance the Helsinki convention of 1992 required a 50% reduction of nutrient loads deposited into the Baltic sea (HELCOM, 1994). This affects the Finnish water protection targets until 2005 (Vesiensuojelun tavoitteet vuoteen 2005, 1998) with the goal of reducing the anthropogenic phosphorus load by 45% and nitrogen by 40% from the levels of early 90's. Pressure is placed on reducing agricultural phosphorus runoff, its contribution to anthrophogenic phosphorus levels being 60% (Finland´s Natural Resources and the Environment, 2002).

Finland's program for the protection of the Baltic sea from the year 2002 nationally quantifies levels for acreages of wetlands and buffer strips. It calls for 20 000 hectares of wetlands and 40 000 hectares of buffer strips. Together with agreements for reduced use of fertilizers, controlable drainage systems and organic farming this should guarantee a reduction of about 40% in total phosphorus and nitrogen levels (Suomen Itämeren suojeluohjelma, 2002).

The European Water Framework Directive (WFD) takes a different approach towards setting the targets. River Basin Management Plans will identify basin specific targets, baseline scenarios, gaps, and schemes of measures. Even though the national national goals and programs remain in effect, WFD differentiates regions with respect to reduction targets. Thus the eventual targets will be set on the basis of the needs of one basin only, and can hence be different even between fairly similiar basins.

The importance of achieving the environmental targets cost-effectively is commonly acknowledged. This requires analysing the costs and effects of potential measures. There are significant differences in assessing efficiency nationwide compared to operations of a smaller scale. Simple calculations can show that heterogeneous agricultural regions have different cost-effective solutions to reach the same target (see e.g., Sharpley and Rekolainen, 1997). Nationwide planning looses efficiency due to this heterogeneity of areas. More focused planning, like that induced by WFD, avoids this problem to some extent. But are there major differences in cost-effective allocations even between homogeneous regions with different target levels?

Analysing efficiency of agricultural environmental policy is complicated. Ambient water quality is not directly linked to pollution loads from the basin. The observability of emissions is limited and also impedes the creation of the right incentives for the agents responsible for the emissions. (Segerson 1988, Fleming & Adams, 1997). Uncertainty can be reduced by increasing spatial information on emissions (Xepapadeas, 1995). Benefits gained from this are tempered by the costs incurred in gathering more information. (Helfand & House, 1995).

Even deterministic modeling of costs and effects of agricultural nutrient abatement is difficult. Models are scarece which use simultanously biological and economic data to assess the efficiency of different measures and their levels of protection are scarce. Vatn et al (1999) provide a framework for interdisciplinary modeling, and an application for assessing cost-effectiveness of different strategies for reducing nitrogen and phosphorus losses. Their application uses deterministic and stochastic physical submodels to predict the physical and economic consequences of chosen actions. Their model, however, is primarily meant for analysing various policy scenarious, not for *finding* the cost-effective combinations of policies.

Studies specific to the cost-effectiveness of agricultural nutrient reductions in the Baltic basin include, for instance, Byström (2000) on cost-effectiveness with or without the use of wetlands, Ollikainen & Honkatukia (2001) and Gren (2001) on costeffectiveness with or without international coordination. Eloffson (1999) builds a spatial model of agricultural nitrogen reduction for the whole basin. She estimates cost functions of eleven measures to reduce nitrogen and presents cost-effective solutions for different target levels.

Hart and Brady (2002) include the issue of different target types in their study on efficiency of agricultural policies in reducing pollution of the Baltic Sea. Their focus is on regulator's optimal responses to three alternative ways of setting the target: the three variables being emissions, nutrient stock or ecosystem quality. Their approach includes various measures, with a focus on choice of crop, management practices, and the levels of fertilizer use. The efficient abatement solutions are combinations of these. They find that optimal strategy is strongly dependent on both the types and the levels of targets.

Brady (2003) introduces regional aspects and agricultural policy into a nitrogen abatement model. He defines a least-cost solution based on present levels of agriculture subsidies as second best and compares it with a least-cost solution based on the situation where subsidies are decoupled from production. The optimal solutions differed substantially in land use allocations, abatement intensity and total costs of abatement. Both Brady (2003) and Hart and Brady (2002) used mathematical programming models.

What has not been analysed, however, how robust cost-effectiveness of an abatement allocation is inside one region with varying target levels. How severely do lower abatement allocations fail to be cost-effective if used at the same ratio but on a lower abatement level?

We will use the River Basin approach of WFD to illustrate the idea. Suppose we have two basins, similar in many aspects, but the load affects the receiving watersheds differently. Hence the River Basin Plans (RBP) would pose different targets on phosphorus abatement. WFD requires that cost-effectiveness of supplementary measures has to be assessed in RBPs. Are there any dangers entailed in doing this primarily for one basin only and then the other basin adapting the same, cost-effective scheme of measures, but executing it on a smaller scale? Can efficiency properties be violated? This paper aims at answering this question.

In the present study we quantify the importance of setting the target levels for nonpoint source phophorus abatement correctly. We consider a single type of target, for which we solve the cost-effective scheme and analyse how the cost-effectiveness is retained when using the same allocation on a smaller scale. For this purpose we develop a static nonlinear mathematical programming model that generates cost-effective solutions for various target levels inside a region. With the help of the model we can compare the costs of cost-effective, and any other allocation supporting the same reduction in phosphorus loss. The application is located in the target area of the Yläne river basin, Southwest Finland and considers 3 measures: reducing the use of phosphorus fertiliser, creating buffer strips and creating wetlands.

The rest of the paper is organized as follows: Firstly we present the analytical model, secondly we introduce the numerical model and the application for the Yläne river basin, thirdly we present the results, and finally, we discuss the results.

The analytical model

Cost-effectiveness equalizes marginal abatement costs from all sources of pollution for any target level (e.g. Baumol and Oates, 1988). We use this general result to illustrate the general idea of the paper. Measures to prevent and reduce agricultural nutrient loads are different in terms of intial costs, effectivenesses etc resulting in nonidentical abatement and marginal abatement functions. Furthermore, abatement functions are not linear.

Figure 1 presents three arbitrarily chosen curvatures of marginal abatement cost functions[1]. Readers can think of these as: marginal abatement costs for wetlands, *MCw;* fertilizer reductions for some crop, MCP_i; and construction of buffer strips, *MCs*.

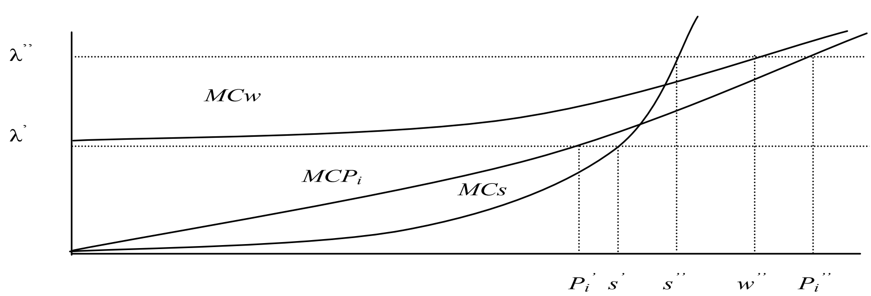

Figure 1. Uniqueness of abatement allocation

[1] The curvatures are chosen for illustrative purposes and thus do not represent the results of this paper.

In the graph we have depicted allocations of abatement measures that satisfy two different pollution constraints: high ($''$) and low ($'$). The horizontal axis denotes the abatement of measures under the two constraints, and the vertical axis the shadow cost of the constraint. For higher level of total abatement the buffer strips reduce the pollution by s'' ,wetlands by w'',and fertilizer reduction by P_i''. The overall level of abatement is a sum of these. The shadow cost of this higher target is λ''. Important to note is, that for the lower reduction target we use no wetlands, and the buffer strips become the most important contributor to overall reduction. The contribution of measures towards abatement differs radically for two different target. Hence using the contribution ratios of the higher target for the lower target would not yield a lowest cost allocation of abatement and vice versa. Our aim is to quantify these failures in their relation to agricultural phosphorus reduction.

We link costs due to changes in agricultural production and introduction of phosphorus abatement measures, with respective changes in phosphorus loss. We consider a single region which can be viewed as a representative farm. The quality of soil varies within the region and the land is allocated to different crops on the basis of these differences. This allocation is assumed to be fixed and we assume that a certain crop is always allocated to a certain soil type. In addition to crop distribution, soil quality distribution mainly determines the phosphorus loss (hereafter refered to as P-loss). The social planner chooses three measures to reduce this P-loss: phosphorus application levels for various crops, buffer strip width and the size of a representative wetland, in a least cost manner to satisfy an exogeneous constraint on total phosphorus loss.

We define the loss of total phosphorus from the region as:

$$E = \sum_{i=1}^{n} e(\theta_i, P_i, w, s), \tag{1}$$

where:

θ: a phosphorus status parameter for soil type $i = 1 \ldots n$

P_i: phosphorus application on a crop allocated for soil $i = 1 \ldots n$

w: acreage of wetlands (or a representative wetland)

s: width of buffer strip

Thus the P-loss from the region is a sum of losses from each soil type. The parameter θ captures two relevant aspects of soil quality: the crop yield potential and the P-loss potential. At this point we do not define it explicitly, but merely state that it affects yield and loss. The representative wetland (w) is common for the whole region, and the buffer strips (s) are constructed of even width on locations suitable for them, independent of crop distribution. The model is thus very simple: it does not allow for crop rotation or targeting the buffer strips according to crop use. The abatement measures affect the average loss from the region, not individual losses separately.

These simplifications can be done without loosing too much information relevant to our research questions. Obviously our assumptions make the exact values of abatement costs less precise. The effects of varying the scale of abatement, however, can be analysed with our model.

The costs include the opportunity cost of land at different levels of phosphorus fertilizer use, equal to net profit; and both initial and maintenance costs of buffer strips and wetlands.The cost of an allocation *P,w,s* is:

$$C = \sum_{i=1}^{n} \left[\pi(\theta_i \ , P_i^*, w^*, s^*) - \pi(\theta_i \ , P_i \ \ , w \ \ , s \ \ ) \right], \tag{2}$$

where P^*, w^* and s^* denote the choices of fertilizer use, acreage of wetland and buffer strip width under no constraint on P-loss. The cost is thus a sum of differences in profits on all soil types, i.e. between the private optimum and the allocation satisfying exogeneous constraint. The profit from a soil type k with fertilizer use k is defined as:

$$\pi_k = (1 - w - s)\left[f(\theta_k, P_k) p_k - t_k P_k \right] - CC(w,s), \tag{3}$$

where p_k is the price of output of the respective crop, t_k is the price of input (P-fertilizer), and CC the operation and maintanence costs of wetlands and buffer strips. The detailed cotrol cost function is presented with the application. The total acreage of land is scaled to 1, hence the opportunity cost of wetlands and buffer strips is captured in (1-w-s).

The abatement cost is obtained by combining the cost of an allocation, C, with the respective reduction in phosphorus loss, i.e. the difference in loss under unconstraint E^* and the loss under the allocation.

We make the following assumptions about cost functions:

$\partial C/\partial w$, $\partial C/\partial s$, and $\partial C/\partial P_i > 0$ for all i
$\partial^2 C/\partial w\partial w$ and $\partial^2 C/\partial s\partial s = 0$, and $\partial^2 C/\partial P_i\,\partial P_i > 0$ for all i

Thus the costs of constructing acreage unit of a buffer strip or wetland are linear and increasing, and the cost function for fertilizer reduction is increasing and convex.

For the loss and abatement functions we assume:

$\partial E/\partial w$ and $\partial E/\partial s < 0$ $\partial^2 E/\partial w\partial w$ and $\partial^2 E/\partial s\partial s > 0$
$\partial E/\partial p$, $\partial E/\partial p > 0$, and $\partial^2 E/\partial p\partial p$, $\partial^2 E/\partial b\partial b > 0$

Thus we assume P-loss to be decreasing and convex with respect to wetlands and buffer strips and increasing and convex w.r.t fertilizer use.

The objective is to:

$$\begin{aligned} &\max_{w,s,p,b} -C(\theta, w, s, P) \\ &s.t. E(\theta, w, s, P) = \bar{E}. \end{aligned} \tag{4}$$

Our assumptions guarantee interior solutions. The FOC:s are:

$$\lambda = \frac{\partial C/\partial w}{\partial E/\partial w} \tag{2a}$$

$$\lambda = \frac{\partial C/\partial s}{\partial E/\partial s} \tag{2b}$$

$$\lambda = \frac{\partial C/\partial P_i}{\partial E/\partial P_i}, \text{ for all } i = 1...n \tag{2c}$$

$$E(\theta, w, s, p, b) = \bar{E} \tag{2d}$$

Conditions (5a-5d) guarantee that marginal abatement costs (here: marginal costs relative to marginal reduction) are equal for all abatement measures. Condition (5d) quarantees the overall reduction target to be satisfied. The shadow cost of the phosphorus loss constraint is equal to λ which is increasing in target reduction.

We solve (4) with a mathematical programming model. In the following, we present this numerical model, and apply it to a case study.

The numerical model and the application fo Yläne basin

A central assumption in our model is that the costs are additively separable and that the abatement measures are independent. Hence the abatement of a buffer strip (wetlands) does not depend on the level of fertilizer use or the construction of wetlands (buffer strips). This can be assumed since buffer strips and wetlands reduce mainly particulate phosphorus from runoff, and because the loss of particulate phosphorus in runoff is quite insensitive to short term decisions about fertilizer use

(Ekholm et al 2005). The combined effects of wetlands and buffer strips are not analysed. If they both affect the same runoff, the interdependencies are obvious. We will discuss later the possible implications of taking these into account.

We apply the model on actual data from one sub-basin of the Yläne basin, Southwest Finland. The data, and its simplifications are presented at the end of respective sections.

The crop yield function

In Finland the short and intensive growing season and strong binding of phosphorus into soil makes fertilizing a long term project. The phosphorous uptaken by crops is mostly from plant available phosphorus fraction of the soil and the purpose of fertilizing is mainly to maintain this fraction (e.g. Saarela & Saarela, 1999, Saarela et al., 2003). To incorporate the long term character of P-use, we postulate the production function as follows.

In the application we assume θ to be captured in the soil test phosphorus (STP). The crop allocation is made according to this and by looking at the actual allocation we can find justifications for assumptions on values of θ. In the Yläne basin over 90% of the farm land is allocated to grain and grass, and ca. 2 % to potato. Less than one percent is allocated to sugarbeet, 5% is allocated to oilseeds, and 2% to peas.

We incorporate these features in our model by roughly assuming that all the soil is either high or low θ, and high θ soils are allocated to potato, low θ soils to barley. The allocation for barley is assumed to be 95% This way we incorporate the differences in soil quality in the model while keeping it as simple as possible. We set the high θ to correspond to a STP value 25 and the low θ to 13.4. Together with chosen soil quality allocation, the average STP value will be the same as the actual values in the Yläne basin as reported in Palva et al (2001).

The crop yield function is constructed of two components. The first one will have the choice of fertiliser use as a choice variable and the second will depand only on θ:

$$f(\theta_i, P_i) = F(P_i) + A(\theta_i), \text{ when } i = \text{barley and potato} \tag{6}$$

For the first component we choose the yield response function of Saarela et al (1995)[2] :

$$F(P) = \frac{1}{STP} y * kk * P(-0.132P + 12.4) + (24 - 0.367 * STP) * \sqrt{P} + 6.97 \tag{7}$$

where STP = soil phosphorus, y = experimental year and kk= crop parameter. The function thus has other variables apart from P, but P is the only choice variable. The crop parameters kk are 2 for potato and 1.05 for barely (Saarela et al., 1995) and the experimental year is fixed to 6 (the experimental periods of Saarela et al (1995) were on average about 12 years).

In equation (2) we define costs as differences between privately optimal allocations and allocations satisfying the phosphorus loss constraint. In the application we use fertilising levels adopted from Palva et al (2001) as optimal levels, and the reductions are conducted from these at 1 kilogram per hectare intervals. These levels are set at 15 kg ha^{-1} for barley and 45 kg ha^{-1} for potato, and will be later referred to as original levels of fertilizing[3].

The second component is determined with the help of actual crop yield data for Yläne region, i.e. the left hand side of equations (6), and (7), which is determined by fertiliser use.

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[2] The yield function having only phosphorus use as a choice variable is unsatisfactory: P fertilizer is usually given in a fixed proportion with other nutrients. We ignore the cost of these by taking into the fertilizer cost only the proportion of P of the fertilizer mix.

[3] This assumption has weaknesses: the original level is influenced be subsidies. Also the rates are based on reported levels, and are thus subject to errors, both intended and unintended. But, in balance, even though this affects the order of magnitude, the influence on our research question is not substantial.

$$A(\theta_i)= f(\theta_i ,P_i^*)- F(P_i^*) \tag{8}$$

Since (8) is determined by the original level of fertilisation, it is constant for both crops. With these levels and assumed STP values, and the crop yield levels reported in Palva et al (2001) we find the constant crop yield for potato to be 21513 kg ha^{-1} and for barley 3728 kg ha^{-1}.

We assume that 75% of the crop yield of both crops is sold at a higher price (0.14 € kg^{-1} for potatoes and 0.131 € kg^{-1} for barley) and 25% at a lower price (0.034 € kg^{-1} and 0.108 € kg^{-1}). For potato, we use Y1 fertiliser (P-ratio of 2/29) and take the price per kilogram of phosphorus to be 0.46 €. For barley, Y3 fertiliser and the phosphorus price 0.32 € kg^{-1}.

P-loss and abatement functions

Among factors contributing to P-loss from agricultural land are: the level of easily soluble P in the soil, the type of fertilizer used, the methods of spreading, the soil type, the slope of the field, the crop used, and rainfall as en exogeneous factor etc. (Sharpley, 1995; Pote et al., 1996;Ylivainio, 2002; Palva et al., 2001; Ekholm, 1998). Our model directly includes only fertilizer use as an explanatory variable. The phosphorus status is accommodated indirectly as most of the P-loss is determined by θ.

The phosphorus used for fertilising affects the P-loss mainly by steering the dynamics of soil phosphorus (McDowell and Sharpley, 2001; Ekholm, 2005; Yli-Halla et al., 2002). Given the exogeneous variables, rainfall in particular, most of the loss occurs on the basis of soil properties that are constant in short run. We capture this again by separating the loss into a constant share, and a share that varies according to fertiliser use . The soils with high θ have higher constant values for P-loss than low. For the application we assume that 5% of loss is determined by P-fertilizer use, 95% by the constant term at the original level of fertilisation.

We calibrate the actual reported total phosphorus loss by Kuusela and Savola (2000) to match the loss induced by our assumed land use allocations. Following an unpublished work by Granlund et al, we assume the P-loss from land on barley to be 1.11 kg ha^{-1} a^{-1} and on potato 3.33 kg ha^{-1} a^{-1}. Further, the contribution of fertilizer use at the original level is assumed to be 5% of the total P-loss[4]. With these assumptions the constants of P-loss are 1.04 kg ha^{-1} a^{-1} and 3.16 kg ha^{-1} a^{-1}. Making such a significant assumption about P-loss coefficient is mainly to enable us to conduct the analysis with two crops that are different in terms of their crop yield and P-loss characteristics.

Buffer strips

Buffer strips are used at field edges susceptible to erosion (Turtola, 1999). We create a coefficient, *s*, which denotes the fraction of cultivated land suitable for buffer strips. Another coefficient, *k*, tells how much more intense the runoff is from these parts of the field.

The buffer strip phosphorus abatement as a function of strip width takes the form:

$$rb(m) = 1 - ((sk) / (sk + (1\text{-}s))) * m^{\alpha}, \tag{9}$$

where *m* is a width of the strip in hundreds of meters. The ratio [0,1], given by the dependent variable, *rb*, is of the share of P-loss not removed by the strip. According to Uusi-Kämppä and Kilpinen (2000) a buffer strip of 10 meters width removes 30-40% of the total phosphorus (TP) surface runoff. Approximately an equal amount is lost in drainage, where the buffer strips have no effect (Uusi-Kämppä and Kilpinen, 2000). Thus we assume that a 10m wide buffer strip removes 17.5% of the TP runoff originating from the field. We also assume concavity of the abatement function with diminishing marginal abatement. We hence calibrate α to be 0.76.

Based on buffer strip plans for the region we assumed that the field acreage from which the runoff can be reduced by buffer strips is 25% of the total field acreage[5], hence *s* gets the value 0.25. The

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[4] The 5% share is the only part of the loss that can be reduced by fertilizer use reductions. The choice of the value changes the way the dynamics is incorporated into our static model.
[5] Anni Karhunen. 1.7.2003. A written note.

runoff is assumed to be three times more intense from the field acreage suitable for buffer strips, and the parameter $k = 3$. In the sensitivity analysis we vary, among other things, both the above mentioned values and observe the results.

Wetlands

We start postulating the abatement function for wetlands from the phosphorus reduction function of Puustinen et al (2001):

$$W(r) = 13.1r + 0.01, \tag{10}$$

where r is the effective size of the wetland, divided by the acreage of the wetland catchment and W denotes TP reduction as a percentage. Obviously, a linear function has its weaknesses: after the relative size of 7.6% all the phosphorus is removed.

We constrain (6) by assuming the maximum TP abatement to be 75%. This is in line with Reinhardt et al (2005). We further assume this reduction to be achieved with the relative size of 10% and a logarithmic abatement function. By fitting these with the original values from Puustinen we have:

$$w(r) = 25.1\ln r + 20.4 \tag{11}$$

which is a concave function.

As with buffer strips we assume that the acreage of wetland bound runoff is smaller than the whole field area. So the ratio of P-loss not removed by a wetland is:

$$rw(r) = 1 - [ss*(25.1\ln r+20.4) + (1-ss)]/100, \tag{12}$$

where ss denotes the fraction of the field acreage through which the runoff flows to the potential wetland[6]. We assume this to be 0.15, and later analyse the implications of the choice.

Control cost functions

All cost functions are implicitely defined in equation (2) as reductions in profit due to abatement decisions. We now define the explicit costs of constructing a wetland or a buffer strip in equation (3). Costs of fertilizer reduction is entirely captured in equation (2) and the production function (6).

The control cost function for buffer strips and constructed wetlands consist of initial costs (divided by the duration), yearly operation and maintenance costs and the opportunity cost of land. The function is:

$$CC(w,s)=w*(1458+59)+s*(9+25), \tag{13}$$

where the first term of each sum is the initial cost per hectare divided by 15 years, and the second term is the annual operations and maintenance cost. The costs of wetlands is adopted from Puustinen et al (2001) and of buffer strips from Maatalouden ympäristöohjelma 1995-1999:n taloudellinen analyysi (1999), and Suojavyöhykkeen perustaminen ja hoito (2002).

Results: The effects of a changing target level on cost-effectiveness

According the results, all the measures were used at some level of total abatement. The marginal costs of abatement were rising, as predicted. For the purposes of this paper it is not practical to present

[6] Using the acreage as a choice variable is dubious. For instance Reinhardt et al (2005) state that the most important factor affecting phosphorus retention in wetlands is the *minimum* water residence time: most of the phosphorus loss occurs during few high discharge events and the average retention capacity is not important (see also Sharpley and Rekolainen 1997). On the other hand, it is the relative size of a wetland that determines indirectly the water residence time. Better modeling of wetland retention would yield more realistic results..

the exact values of total abatement costs or the precise cost-effective allocations at different target levels. Firstly, the functional forms of the abatement cost function were quite uncertain, and thus we tried to avoid giving recommendations for certain measures at certain intensities even for the target region. Secondly, the purpose of this paper was the analyse scale issues of abatement, a task which calls for a different approach.

To present the results is illustratively as possible, we will consider two features. Firstly, we will show what the cost-effective abatement contributions of individual measures are at different levels of total abatement. This way we can show that the ratios do vary strongly between the target levels. Secondly, we will analyse the costs of using the contribution ratios out of scale, that is to say, showing the unit cost differences between a cost-effective solution and a solution that adapts the ratio of measures from a higher (lower) total abatement level.

Figure 2 presents the abatement contributions of the analysed measures for seven reduction target levels. A contribution is defined as the increase in total abatement achieved by introducing the respective measure to its cost-effective allocation level, from its original level (zero), while keeping the other three measures fixed. This was done in turn for all measures at all levels of reduction. This illustrates how the share in total abatement emerges with changing target level.

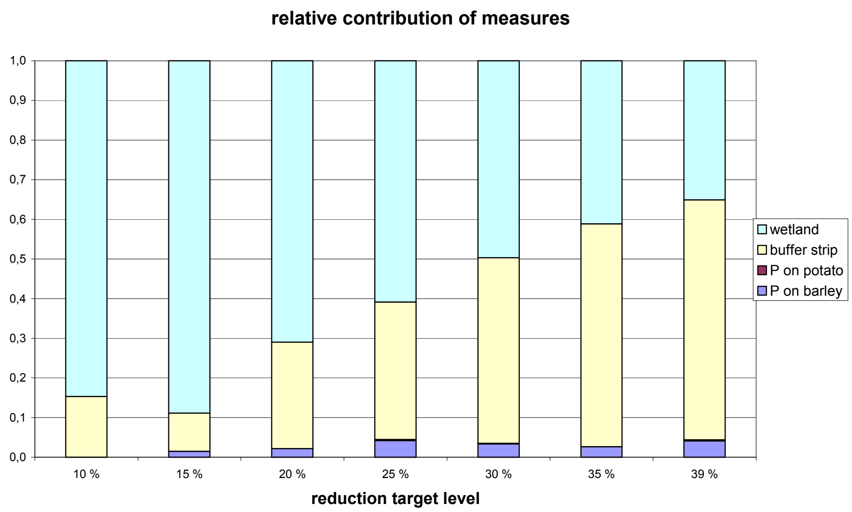

Figure 2. Relative contribution of measures.

From the figure 2 we see that the contributions of individual measures are substantially different under differrent reduction targets. With our modeling assumptions, the wetlands contribute to significantly higher part of total abatement under low levels of abatement, whereas buffer strips contribute more heavily with higher target levels. This means, for instance, that using the contribution ratio of 35% reduction would not be the lowest cost ratio to achieve any other target level. More generally, cost-effective contribution ratios sustain cost-effectiveness only at their own target levels. In the following, we will try to quantify this feature..

The unit cost differences are perhaps the most illustrative way to present these quantifications. In Figure 3 the baseline is the unit cost of a cost-effective 10% abatement on total phosphorus. We then compare this with six different unit costs presented on the horizontal axis, each of them from an allocation that yields a 10% abatement. The first allocation uses the measure contribution ratio of 15% cost-effective abatement, and scales all measures evenly downwards to sustain a 10% abatement. The difference in the unit costs is depicted on the vertical axis. The next allocation uses 20% cost-effective abatement, maintanes the contribution ratio to reduce 10%, and compares the unit costs. This is done until 39% abatement.

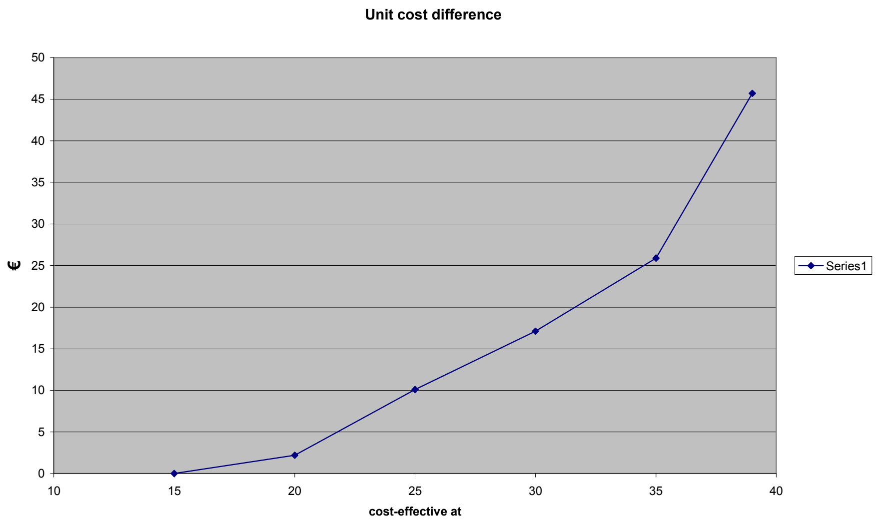

Figure 3. The difference in unit costs

From the graph we see that the difference in unit costs raises rapidly. For instance the average unit cost of reducing 10% of the phosphorus load with a measure contribution ratio taken from 30% cost-effective abatement is 66€/kg, 35% higher than the unit cost of cost-effective 10% abatement (49€/kg).

It is very important to notice that the rising graph in Figure 3 does *not* present rising unit costs of higher abatement levels. All points in the graph contribute to approximately 10% abatement[7]. Unit costs do naturally rise with abatement levels, but much faster than in the graph (for instance, the average unit cost of 30% redution is 120 €/kg). The graph thus shows that the costs of adopting cost-effective schemes of measures out of scale are substantial.

Sensitivity analysis

This section of the sensitivity analysis remains unfinished even though the computations and preliminary analysis are done. For instance, we still have to analyse the effects of scaling up from low levels to high (the effects of which should be of a similar nature to these from high to low, according to Figure 2). We varied input and output prices, land allocation changes, parameter values for crop yield and P-loss functions, and phosphorus abatement cost functions for wetlands and buffer strips (i.e. the parameters of control cost functions and abatement functions).

The central result of the sensitivity analysis is best caputred in the changes in results due to simultanously weakening the phosphorus abatement of the wetland, and increasing the contribution of fertilizer application in the phosphorus loss. This caused a shift in phosphorus abatement from high initial cost wetlands towards lower initial costs (fertiliser reductions and the use of buffer strips).

Recall that the maximum contribution of this years fertilizing on P-loss was originally set at 5%. We increased this to 10% and lowered the wetlands phosphorus abatement to a 50% maximum (originally 75%). Logically, this changes the relative appropriateness of our measures, but what were the implications on the scale issues?

We present the results as we did earlier in Figure 2. Figure 4 depicts the phosphorus abatement contributions of the measures in the new situation.

[7] Not exactly due to the discreteness of the mathematical programming model.

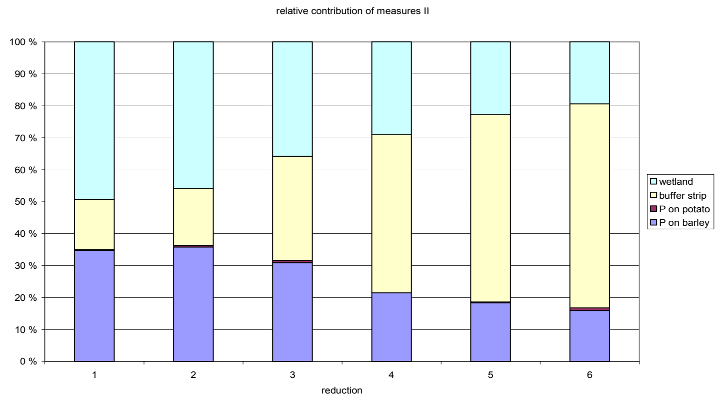

Figure 4. The relative contribution of measures with changed parameter values

We see that, even though the absolute contributions of measures towards phosphorus abatement change from the original situation, the result of substantially different contributions at different levels of total phosphorus abatement remains. This consolidates the feature of cost-effective contribution ratios being cost-effective only at their own target levels. Presumably however, there are quantitative changes due to heavier use of measures of lower initial costs.

Figure 5 depicts the develpement of unit cost difference.

Figure 5. The unit cost difference with changed parameter values

In this case the cost-effective 10% phosphorus abatement has an average unit cost of 71€ kg^{-1}. Using the ratio of 30% cost-effective abatement to reduce only 10% makes the unit costs to raise 24€, that is 34%. Hence, even though the numeric values are different, the percentage variations seem to be very small.

Discussion

There are many things undermining the cost-effectiveness of agricultural environmental policy. Stochasticity in pollution loads, heterogeneity of land, imperfect and asymmetric information, and structural characteristics of agriculture are probably the most important ones.

This paper shows that there are severe difficulties even with the full information and in homogeneous regions. The results of analysis of cost-effectiveness are not directly transferable to other identical regions if the target levels vary. Cost-effective allocations used out of scale are not necessarily more efficient than totally arbitrarily chosen schemes of measures. So, is it important to conduct cost-effectiveness analysis on many homogeneous river basins if they have different target levels? It seems so, as the results of a common cost-effectiveness analysis can be only used if the cost-effective solutions are specified at each level of abatement, as in Hart and Brady (2002), for instance. Then the basins could pick the correct target levels from the plans and these would qualify as cost-effectiveness assesments

This raises two new questions: is the issue only academic? and if not: What recommendations does this give to developing or conducting cost-effectiveness analysis on a basin or national level?

To start by addressing the first question, European river basins face a situation whereby if their baseline scenarios fail to help achieve good water status by 2015, they are required to plan a cost-effective scheme of measures to remedy this. Basins have different baselines and will have limited sets of possible actions for reducing pollution loads resulting from agriculture. Almost certainly, they will not provide general nutrient abatement cost functions that cover a large range of nutrient abatement targets. More likely, they will have few alternative sets of measures, out of which they choose the lowest cost alternative. These schemes of measures do not qualify as cost-effective schemes of measures for any other basin.

This is lesson number one from our research. This result is aggravated by the fact that other basins might have to reduce nitrogen instead phosphorus, or both. The measures affect these two very differently and make the schemes even less transferable to other basins. In our view the river basin level cost-effectiveness analysis should not even aim at providing general results. For a single basin, the only aim is to fulfill the plans in the lowest cost manner. Often this means choosing options from a very limited set of choices. To answer the first question, it very much looks like it is not purely an academic issue.

As for the second question, the problems of this paper and its results originate from the uncertainties in phosphorus abatement functions. For example the abatement function of wetlands is the result of a series of nutrient reductions over a short time period (Puustinen et al 2001). The P-loss values of barley and potato were tentative, especially the loss caused by the phosphorus fertilizer application. Also the stock character of phosphorus is insensitively modelled. Thus at the national, or European level, we should continue with the work on nutrient abatement cost functions of agricultural loads. In the view of this paper it would be particularly important to always conduct the analysis, and present the results, from as large a domain of total abatement as possible. An example of this kind of reporting is found for in Hart and Brady (2002).

In giving recommendations, the scale of reduction should be taking into account. The methods should be ranked according to the phosphorus abatement level, so that they could be applied gradually while maintaining the cost-effective nature of allocations along the abatement path. This could be done in many ways and with any level of accuracy. In its simplest form this would simply mean an order of appearance: as we start reducing phosphorus load from a certain type of agricultural region, we should for instance first construct a metre of buffer strip to reduce the load by n%, then, a second meter to reduce it by m%, then constructing a wetland if suitable for the area to reduc z% etc.

Policy makers and river basin managers should be provided with simple and easily applicable models of cost-effectiveness. These could be models, or even tables or matrices, of the costs and benefits of agricultural phosphrus (nutrient) abatement methods at different target levels, different method combinations, easily applicable for different types of agricultural areas.

Another important issue for further research is the diversified instruments potentially needed due to very different targets in European river basins. Can diversified targets be achieved with uniform instruments, and what are the costs of diversifying the instruments? There are already many studies on these issues, but to our knowledge, not in the context of the Water Framework Directive.

The technical issues of further research include the combined effects of different measures, and different pollutants. We neglected the interlinkages of buffer strip and wetland phosphorus abatement, but also those of phosphorus and nitrogen abatement. Also the dynamics were poorly incorporated in our analysis: none of the costs were discounted, the soil dynamics were extremely simple etc.

The Water Framework Directive calls for integrating economics in a European wide policy process incorperating environmental protection. It is important to avoid doublicating work when assessing costs and effects. Also, the econmists should bring their insights into the analysis processes before the actal work starts.

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