Cost-effectiveness analysis of measures to reduce nitrogen loads from agriculture: do secondary benefits matter?

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Abstract

Over the past years a large number of studies have explored cost-effective strategies for reducing nitrogen loads from agriculture. However, the majority of these studies focus on financial costs to agriculture alone, in spite of the fact that a number of relevant measures, e.g. establishment of wetlands and reduced livestock hold, lead to significant secondary environmental benefits. Ignoring these benefits in cost effectiveness analysis leads to a risk of inefficient policy recommendations. In this paper we identify the relevant secondary effects of four measures to reduce nitrogen loads from agriculture and demonstrate the implications of including secondary benefits using an example of cost-effectiveness analysis (CEA) of reduced nitrogen loads based on financial and socio-economic cost estimates.
Keywords: nitrogen loads, cost effectiveness analysis, secondary effects, benefit transfer.

1 Introduction

Over the last 30 years the detrimental environmental effects resulting from nitrate losses from agriculture and other sectors have been in focus of the environmental policy in Northern Europe. Besides various national regulations the problem is addressed by HELCOM [1], in the EU Nitrate Directive, and in the Water Framework Directive, and recommendations on which policy measures to apply for reducing nitrogen losses has been addressed by the EU commission and OECD. This focus has also resulted in a number of economic studies analysing measures for reducing nitrogen losses from agriculture. The

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framework applied most often is cost-effectiveness analysis where the aim is to appoint the cost-minimising strategies resulting in a pre-defined environmental target. Among the large empirical literature e.g. [2–6], a common feature is that all cost estimates represent solely financial cost to the agricultural sector.

Regulating nitrogen emissions from agriculture also influence other environmental pressures such as emissions of ammonia and climate gasses, as well as changes in land use directly influence the supply of goods related to biodiversity and landscape. From a socio-economic point of view these secondary benefits should be reflected in the cost estimates and further, the implementation of the Kyoto protocol and the EU Habitat Directive has led to increased administrative attention to include the secondary effects in policy analysis. Therefore, when preparing the basis for the third Danish Aquatic Action Plan in 2003-04 an attempt was made to quantify the secondary environmental effects in terms of air emissions and include these in the economic analysis using the shadow price approach. In this paper we report the results from this work and, further, we illustrate how recreational and amenity benefits can be included using benefit transfer. Last the consequences of including these in policy analysis are demonstrated by presenting results from financial and socio-economic cost-efficiency analysis of four selected policy measures.

2 Principles of cost measurements and description of measures

In cost-efficiency analysis the monetary value of the improved environmental quality in target is not explicitly included as in a cost-benefit analysis. Therefore the focus of the economic analysis is to establish valid estimates of the costs of implementing different policy measures relevant for reducing nutrient loads. Costs estimates of the policy measures should represent the change in welfare to society caused by implementing the measure. This is approximated by the socio-economic rent, calculated as the difference between income (if any) and total costs from implementing the measure. Further, estimates for the economic value of effects on secondary benefits (e.g. reduction of climate gasses) are included in the net costs. Thus, the welfare-economic analysis focuses on costs as a proxy of the societal loss of consumption possibilities.

The measures that often are discussed and applied when regulating agricultural nitrogen loads lie within two groups; measures regulating input use, livestock production and crop rotation, and measures changing land use. This first type of measures reduces production intensity or gives incentives for implementing environmental friendly production technologies, but agricultural production is basically maintained. This group encompasses reduced nitrogen input, wintergreen fields, restrictions on manure application, etc. The latter type of measures changes land use permanently e.g. by establishment of wetlands, extensive grasslands, buffer strips or afforestration. Note as this analysis focuses on secondary environmental effects the starting point are the actual changes in activities, and therefore the policy leading to the changes (e.g. taxes, subsidies, quotas or command-and-control) is not considered.

We narrow down the analysis to four measures all leading to various scales of secondary environmental effects: mandatory reduction in nitrogen fertiliser input on all farms, reduced livestock hold, establishment of wetlands, and afforestation. In order to make the comparisons of the measures consistent the measures are scaled to result in a yearly reduction in N loads by 5,000 tonnes. In Table 1 the measures are described.

Table 1: Description of the measures.

Measure	Description
Reduced N input	Reduction of total nitrogen input by 5 percent on all farms
Reduced livestock hold	Reduction of agricultural livestock hold by 12 percent
Wetlands	50 000 ha agricultural land converted into wetlands
Afforestation	135 000 ha agricultural land converted into forest

Source: Anon [7].

All of the measures have been analysed as part of the preparation of the third Danish Aquatic Action Plan [8], however, only including secondary benefits with respect to air emissions; see [9] for a evaluation of the first two action plans. In the following section the secondary environmental effects of the measures are presented and the possibilities for including these in the socio-economic analysis are outlined.

3 Secondary environmental effects

When applying measures for reduced nitrogen loads from agriculture a number of secondary environmental effects are likely to occur. These encompass changes in ammonia (NH_4) and climate gas emissions (CO_2, CH_4 and N_2O) and provision of goods related to biodiversity and landscape. The economic value from each type of effect relates to changes in various goods. Ammonia emissions lead to eutrophication of low-nutrient nature locations such as bogs, oligotrophic lakes, dry grasslands and inland heath lands. The effects of changes in ammonia emissions therefore primarily relate to changes in the status for national biodiversity preservation. With respect to climate gas emissions the impacts are of a global scale and range from impacts on urban settlements and agriculture to biodiversity preservation (see for example ExternE [10]).

For both types of effects it is – at least ideally – possible to construct a quantitative system for assessing the welfare consequences of changes in the emissions. If the relationship between production activities, emissions, transport and decomposition, loads, and effects can be modelled, valuation of the effects is possible by use of revealed or stated preference methods or shadow prises. The principles of these types of valuations are found in Freeman [11] and practical examples of dose-response modelling can be found in the EcoSense model [10].

For some of the measures the secondary benefits relate both to effects resulting from changes in emissions and from direct changes in the provision of different goods. This is the case for establishment of wetlands and afforestration for which changes in the provision of recreational and biodiversity goods will occur as a direct result from changing land use. The provision of biodiversity and recreational goods at a given location are of cause correlated, but not unambiguously. Thus, a location with high biodiversity value does not need to have a high recreational value, as the realisation of recreational values is conditional on accessibility. Opposite an area with high recreational value need not possess high biodiversity value (e.g. think of a golf course).

When analysing the economic consequences of changes in land use it is useful to distinguish between use values and non-use values. Using this terminology recreational opportunity is strictly a use value where as biodiversity leads to both use and non-use values. In Table 2 the types of goods related to biodiversity effects of changes in land use are outlined.

Table 2: Types of goods related to biodiversity effects.

Type of good	Value function
Use value (amenity and recreational value)	The range of the value depend on public access to the area and distribution of property rights (e.g. fishing and game shooting)
Existence value (non-use)	The value of knowing that a given nature location, nature type, or species exist to day
Bequest value (non-use)	The value of knowing that a given nature location, nature type, or species are preserved for the benefit of future generations

For wetlands the areas will typically not be subject to public access so far they remain in private property. The secondary values related to the changes in the provision of recreational goods from this measure are, therefore, restricted to the owner in terms of e.g. fishing and game shooting.

The same values (except for fishing) will be affected by afforestation and, further, recreational values to the public are expected to arise. This is because the public access is legally ensured in Denmark to all public forests and all privately owned forests larger than 5 hectares.

4 Benefit transfer of values for non-market goods

Valuation of secondary effects often requires determining a monetary value for goods and services that are not traded on a market. The last decades have seen a rising attention on non-market valuation methods, but implementation of a valuation study is costly and time consuming. A less costly alternative would be to implement a benefit transfer study, i.e. the transfer of monetary estimates of environmental values estimated at one site (study site) to another, so-called policy site.

Benefit transfer refers to the practice of transferring non-market values for environmental goods and services from a “study” or “source” site (i.e. the site where an original valuation study was conducted) to the “policy” or “target” site (i.e. the site where benefit estimates are required for decision making). Benefit transfer as a research area started to gain attention about 12-15 years ago and has since made it into every book covering the issue of non-market valuation of the environment e.g. [12–14]. The US Environmental Protection Agency’s manual for cost-benefit analysis has dedicated a separate chapter to the subject of benefit transfer [15], and a similar OECD handbook is currently under preparation. The different benefit transfer approaches found in the literature can be broadly divided into four categories [16]: Unit value transfer; Unit value transfer with adjustment, e.g. for income; Benefit function transfer; Meta-analysis.

Unit value transfer is the easiest way of transferring benefits. It consists of applying unadjusted mean or median benefit estimates from the study site at the policy site. Simple unit value transfer basically assumes that the utility gain of an average individual at the study site is the same as that of an average individual at the policy site. This supposition will hardly hold in most circumstances as people at study and policy sites might differ from each other in terms of income, education and other socio-economic characteristics that affect their preferences for e.g. recreation. Likewise the good to be valued at study and policy site respectively might not be similar enough to be comparable, as well as the existence supply of the good and of substitutes might not be stable over time and space. Instead of transferring unadjusted unit values the policy analyst can adjust the value estimates to better reflect differences in socio-economic characteristics between policy and study site, e.g. by use of the Purchasing Power Parities.

By transferring the entire benefit function instead of per unit benefit estimates more information can be transferred between study and policy site. Benefit function transfer can directly account for differences in user and site characteristics. This, however, requires access to an original study where benefits are described as a function of different explanatory variables. A related method is to extract information on benefit values from a range of available studies, so-called meta-analysis. Here the relationship between benefit estimates of a number of different studies is quantified by employing regression analysis where the different study results are treated as the dependent variable, while model characteristics, country, etc. are used as explanatory variables.

The assessment of non-market secondary benefits in this study is entirely based on benefit transfer from existing studies. In order to reduce the uncertainty resulting from transferring values as much as possible most original studies are taken from Denmark. Given the large uncertainties associated with benefit transfer sensitivity analyses are conducted that indicate the impact of variations in unit values for the final ranking of the four analysed policy measures.

5 CEA based on financial and socio-economic cost estimates

In Table 3 the units derived from the benefit analysis applied in this paper are shown for the single measures. Amenity values for afforestation projects are transferred using average unit willingness to pay (WTP) values per house for different distances to the forest edge from two Danish hedonic prising studies

[17, 18]. These unit values are calculated as a percentage of the average house price in an area that allows for an adjustment for income differentials as reflected in house price differences. Recreational values are transferred using average WTP values per forest visit from a Danish contingent valuation study [19] and information about forest visitation patterns of the Danish population from [20].

Values for air emissions are estimated as shadow prises. The shadow prise approach holds the same characteristics as unit value transfer, as the shadow prise is calculated as the marginal abatement costs of a current or planned policy. The shadow prise approach can only be applied for including secondary benefits, and it requires an explicit target for reducing the emissions and the existence of a cut-off prise so that a marginal willingness to pay for reducing the emissions can be derived from existing policies.

The estimates for recreational and amenity values reflect differences in visit frequencies and housing prises between rural and urban areas. The range in estimates for climate gas values reflect estimates of the future compliance costs for the European Commission [21] and ExternE [10], where as ranges for fishing and game shooting values are based on marked data. We refer to Schou and Birr-Pedersen [22] for a technical description of how the benefit values are calculated.

The measures for which the secondary benefits with respect to recreation and biodiversity are most significant are establishment of wetlands and afforestration as these result in significant changes in land use. The other two measures primarily result in reductions in production intensity although reductions in livestock hold may lead to reductions in grasslands and grassing potential if cattle stocks are reduced. For the amenity values only the estimates from the rural and urban baseline scenario are shown. Note also that because wetlands typically are kept in private property without public access no amenity or recreational values are attached to this measure.

Table 3: Benefit values.

Secondary effect	Unit	Mean	Min	Max
Ammonia reduction	€/kg NH4-N	1.0	-	-
Climate gas reduction	€/tonne CO2-eqv.	11	11	46
Game shooting, wetland	€/ha	25	25	50
Game shooting, afforestation	€/ha	50	25	63
Amenity value forest urban areas	€/ha	1 976	751	3 200
Amenity value forest rural areas	€/ha	63	26	99
Recreational value forests	€/ha	132	13	660

By multiplying the unit values with the scale of the secondary effects or the scale of the measure and then dividing the aggregate costs by the estimated reductions in N loads of 5,000 tonnes, the range of the secondary effects per kg N load reduction is derived. For the afforestation measure values for the rural area are used, as the need for reducing agricultural nitrogen loads typically originates in rural areas. Therefore, the mean estimate for amenity values for rural areas is applied and the lower annual estimate of 13 € per ha for recreational benefits, which reflects an average annual visit frequency of 20 visits per ha. However in the case where afforestation can be targeted to locations nearby urban areas the recreational values can be increased significantly. As can be seen from Table 4 costs per kg N reduction turn out to be substantially negative when mean amenity values for urban areas and the annual mean value of 132 € per ha for recreational benefits is applied.

In Table 4 the result is shown together with the estimated financial costs of the measures according to Jacobsen [8]. The financial costs are expressed as loss of economic rent.

Table 4: Financial and socio-economic CEA, based on mean values (€/kg N).

Measure	Financial costs	Secondary effects			Socio-economic costs
		Emissions	Market use values	Non-market use values*	
Reduced N input	2.1	0.6	0	0	1.5
Reduced livestock hold	6.9	2.4	0	0	4.5
Wetlands	4.4	0.6	0.3	0	3.5
Afforestation, rural area	6.4	1.3	1.4	1.7 + 0.4	1.7
Afforestation, urban area	6.4	1.3	1.4	53.3 + 3.6	-53.2

* The first value reflects amenity benefits, while the second value reflects recreational benefits.

The inclusion of the secondary environmental effects in the net cost estimates shows two significant consequences for the CEA. First, the abatement costs are reduced significantly compared to those of the pure financial analysis. This indicates that policies formulated based on financial economic analysis alone will overestimate the aggregate costs and, thus, tend to lead to less ambitious policy goals compared to the socio-economic efficient solution. Secondly, the relative cost-efficiency of the possible measures changes. This is especially the case for measures involving land use changes where amenity and recreational values are expected to arise. Thus, the cost-efficient mix of policy measures changes when shifting from financial to socio-economic cost-effectiveness analysis. This indicates that the secondary environmental effects may play an important role when formulating environmental policies.

It is important to notice that the analysis is based on average values. For example if afforestation, for which the current analysis actually indicates a positive socio-economic performance, is implemented to a large extend the marginal recreational value should be expected to fall. Further, because of the increased demand for agricultural land to be used for afforestation the financial costs of this measure will rise. These two effects in combination will reduce the cost-efficiency performance of the measure eventually leading socio-economic costs to be positive. Therefore when analysing the cost-efficient mix of policy measures at the larger scale it is important to include considerations of how the demand functions and, thus, the marginal values, for the different goods will be affected in different policy settings.

6 Discussion and conclusions

In this analysis we demonstrate how non-marketed secondary effects can be included in the cost-efficiency analysis. A general feature of benefit transfer as well as primary valuation studies is that the exactness of the estimates depends on how specific the project is described. If the analysis relates to a well described project at a designated location it usually should be possible to develop a detailed description of the changes in land use, effects on environmental quality and biodiversity, and to which extend the project will change recreational possibilities. Such a description yields a good basis for deriving monetary estimates of the benefits. Further, the uncertainty of the benefit estimates will mostly depend on the benefit functions used and can be subject to a reasonably noncomplex sensitivity analysis.

If the analysis deals with policy choices at the more aggregated level, as is the case in this example, decisions as to where the measures should be implemented and the scale of the measures are often not explicit, and may for political reasons not be desirable to clarify. In this case benefit transfer may be difficult especially when the effects and benefits hold site specific elements. However, this analytical problem is not only related to benefit estimates but also to the financial economic and natural science evaluations. But because of the relatively limited data on benefits and their variations with policy relevant parameters the issue becomes more explicit. Given the limited amount of studies available in Denmark and internationally this project was not able to transfer values for changes in biodiversity particular non-use values related to changes in land-uses. Results from this paper indicate the substantial influence non-market values can have on the ranking of policy initiatives. It is therefore suggested that future research should focus on eliciting these non-use values in order to provide policy makers with an indication of their potential size and variability.

Decision making in an administrative context has a (very natural) tendency to seek simplifications of their processes. With regard to project and policy evaluation this becomes evident in an increasing focus on promoting and using so-called "approved unit values" in the form of € per measurement unit for benefit transfer. Such values are extremely context dependent and a cautious and qualified usage of these benefit estimates is therefore strongly required.

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