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Reijo Pirttijärvi

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Foreword

Agri-environmental issues are substantial in the research agenda of the Agricultural Economics Research Institute. The focus is on the consequences of agricultural and trade policy measures on farm economy and the environment, the instruments of environmental policy, and the environmental indicators of agriculture.

Agriculture is a significant source of nutrient runoffs in water systems. This fact and the evolution of agricultural policy (shared goals) make it essential to be able to measure the nutrient flows from a point source perspective. Nutrient balances provide a tool to measure farm-specific nutrient surpluses and, consequently, farm-specific environmental load. In addition, nutrient balances provide a tool to farmers to improve the efficiency in using nutrients and, thus, improve the profitability of production.

The main emphasis of this study is to examine the role of nutrient balance calculations in environmental policies at the farm level. The reasoning of agri-environmental policies is based on the definition of a desirable state of the environment, or, desirable trend in the state of the environment. Among other things, this study compares the farm-level economic implications of choosing alternative policy measures aiming at a reduction of up to 30% in nutrient leakage: input tax, output tax, and emission tax based on the nutrient balance calculations.

The study has been undertaken by Reijo Pirttijärvi. It has been presented as Licentiate's thesis of agricultural policy at the Department of Economics and Management of the University of Helsinki. At the Department, the study has been supervised by Acting Professor Jukka Kola, to whom the author wants to express his sincere gratitude.

Major part of the study was carried out at the Agricultural Economics Research Institute. The author wishes to thank his colleagues, especially Jyrki Aakkula, Jussi Lankoski and John Sumelius, for their encouragement and constructive criticism. In addition, the author is grateful to Professor Lauri Kettunen for commenting on the study. Special thanks are due to Sirkka-Liisa Rinne at the Agricultural Research Centre, and Helena Kämäräinen and Päivi Ojanen at the Kainuu Rural Centre, for providing the data sets, and to Jaana Kola for revising the English text.

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Helsinki, August 1998

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NUTRIENT BALANCES IN AGRI-ENVIRONMENTAL POLICY

REIJO PIRTTIJÄRVI

Abstract. Agricultural nutrient runoffs are difficult to measure because of the nonpoint nature of the emissions. Policy solutions are emerging suggesting incentive schemes and farm co-operation to reduce the nutrient runoffs. An alternative way to tackle the problem is to calculate nutrient balances and set economic instruments or direct regulations on the nutrient surplus. However, nutrient balance calculations do have difficulties, e.g. unfavourable weather can increase nutrient surpluses dramatically, also depending on a crop. The sensitivity of crop response to weather conditions leads to similar responses with respect to the nutrient surplus. Unfavourable growing conditions lead to high nutrient losses.

The farm model of assessing the cost-efficiency of input tax, output tax, and emission tax based on the nutrient surplus showed the input tax to be the best one with respect to this criterion. The effect of weather conditions on the cost-efficiency ranking showed that if the weather is unfavourable for crop growth the emission tax is more cost-effective than output tax only at 10% abatement level. Input tax is by far the most cost-efficient policy instrument of these three even under stochastic weather effects

The final choice of a policy instrument is a matter of many perspectives, cost-efficiency being one of them. In addition, criteria such as how easily the instrument is enforced, and how time- and site-specific the environmental responses are, play an important role in the final policy ranking. Furthermore, the information dimension also has to be considered in the policy choice. The better the policy instrument communicates information both to the farmer and the policymaker, the more willing the farmer (the agent) is to provide as precise as possible data on the true impact his production has on the environment, and the better the policymaker (the principal) is conscious of the results of the policy it has established. With this respect, the nutrient balance calculations provide the best information dimension.

Index words: agriculture, environment, information, modelling, nutrient balance, weather

Contents

1. Introduction

1.1. Background

The public interest in finding ways to reduce nutrient runoffs to rivers and lakes has risen considerably during the past few years in Finland, especially during the year 1997. The reason for this is the eutrophication and algae growth in our watercourses due to the increased nutrient levels in them. In the public discussion, agriculture has been labelled to be the main source of water pollution. True, in part, but also a too simplistic point of view.

The characteristics of agricultural nutrient runoffs are often not very well understood. The biological processes of nutrient flows in agriculture are difficult to measure and to control because of the nonpoint nature of the emissions. On the other hand, substantial progress has been made in the control of discharges from point source activities. This is due to the fact that controlling point source discharges, e.g. from chimneys or drainage pipes, is a rather straightforward task; we only need to install a proper filter device or refinement facility at the end of the discharging pipe, and the problem is dealt with.

Therefore, the characteristics and extent of water quality problems caused by nonpoint sources of pollutants emanating from agricultural activities highlights the urge to find efficient policy solutions to these problems, as noted e.g. by Weinberg (1991). From the viewpoint of an economist the pollution problems from agriculture are associated with market and policy failure in which the costs and returns of farm production decisions are inadequately internalised. As a result, private decisions lead to socially sub-optimal allocation of resources and production methods. The economic challenge, then, is to find the missing information to market participants by assessing alternative policy instruments (Weersink 1997). These instruments can modify farmer behaviour with the view of achieving sustainable farming production objectives, or specified nutrient abatement targets.

Thus, the characteristics of nonpoint source pollution make the design of agri-environmental control policies more difficult, yet challenging. A controlling measure such as an emission tax is usually an infeasible measure in the control of agricultural nutrient runoffs, since agricultural emissions are not easily identifiable. Therefore, input taxes or input restrictions are usually the preferred policy choices. However, from the theoretical viewpoint these are but the second-best policies. This leads us to questions, such as; 'What are the best ways to tackle the agricultural nonpoint pollution problem, and can we use emission taxes?'. Hence, the question is not *what* we should do but *how* we should do it.

1.2. Objectives of the study

First, this work aims at understanding the characteristics of agricultural pollution and the different possibilities to reduce it through government or market based control policies. As the nonpoint source discharges have gained more relevance in water pollution, the characteristics of these must be taken into consideration in policy formulation. Environmental control policies are seen to be implemented in the principal-agent framework and applied through economic instruments, direct regulations and information policy.

Secondly, the nutrient balance calculations are seen as a way to be able to assess the farm specific nutrient surpluses, and consequently the environmental load of a farm from a point source perspective. The different methods of the nutrient balance calculations (and their difficulties) are scrutinised and discussed. *A special focus* is on assessing the importance of weather factors influencing the build-up of nutrient surpluses, which thus influence the choice of a policy instrument. In addition, the connection between the farm economic indicators and efficiency in the nutrient use is studied.

The *main emphasis* of the research is to examine the role of the nutrient balance calculations in nutrient abatement policies at farm level. The research problem is; "How does an emission tax based on the nutrient balance calculations compare with other control policies by the cost-efficiency criterion?". In this, the impact of the weather on the policy choice is assessed. The problem is approached through a modelling application on dairy farms, and the results are generalised at a broader level. In addition to the cost-efficiency criterion, several other policy-ranking criteria are discussed in the final policy choice assessment.

1.3. Approach of the study

Environmental problems of agriculture stem from physical phenomena. Therefore, understanding the biological aspects of the nutrient flows also helps the economists to develop functioning agri-environmental policy solutions to tackle the agricultural pollution problem. The approach of the analyses conducted in this research is illustrated in Figure 1.

Choosing and setting up a policy measure is based on biological and economic information. Policymakers first focus on information on the present state of the environment. This information is presented to them by biological scientists (soil and water scientists). They can also present the level of the desirable environment to the policymakers, although they seldom have enough information to assess such a level precisely.

The next step is to judge what is 'the best', i.e. the optimal, way to achieve the aforementioned desirable state of the environment. This research focuses on this problem by comparing the economic implications of using different environmental policy measures at farm level. The term optimal is split into five policy goals: cost-efficiency, information dimension, site- and time-specificity, enforceability, and correlation to water quality. The main focus is on assessing the cost-efficiency of the measures. Furthermore, the effect of weather factors to the policy choice is studied.

This research draws a comparison of different policy choices. In addition to the two widely used environmental controlling measures, i.e. an input tax and a product tax, a new concept of nutrient balances is introduced; and its usefulness and implications as an output tax are considered as the third policy option to reduce environmental load from agricultural sources.

The focus of the research is to set up a policy measure which fulfils best the specified criteria in a situation in which a government policy aiming at a reduction of 30% in agricultural nutrient leakages is established. The assessment is conducted by modelling silage production.

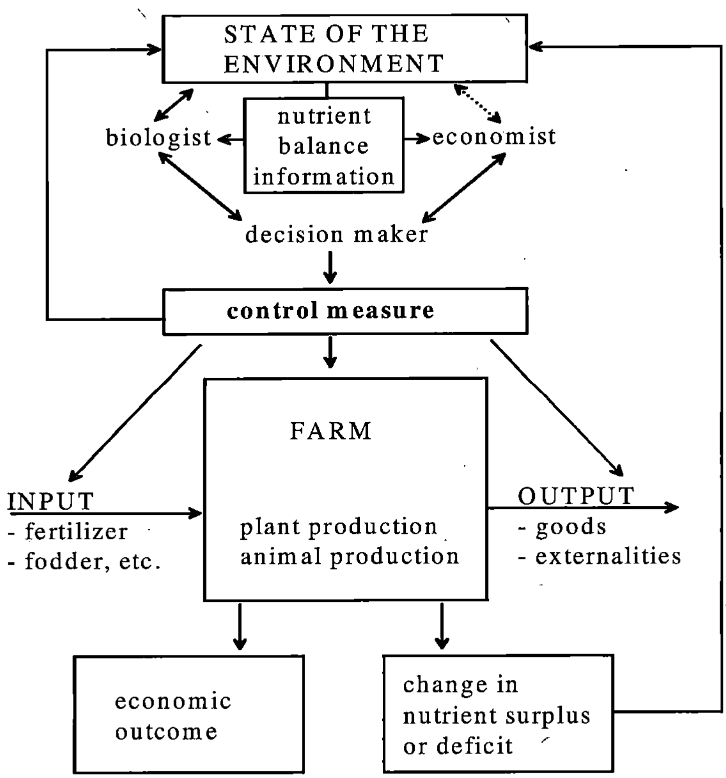

Figure 1. The approach of the study.

The decrease in the leakage is measured in the change of the nutrient surplus or deficit, i.e. as the difference between the nutrients entering the farm and the nutrients leaving the farm. The advantage of using the nutrient balance at farm level is that this makes it possible to assess agricultural load to the environment from the perspective of point source pollution, and not from the nonpoint source pollution perspective as it is usually treated.

In addition, nutrient balances give information to farmers on the efficiency of using the inputs at their disposal, and work as an incentive to improve the usage of nutrients – especially in the manure. Furthermore, the role of information is specifically analysed in this context, since the information dimension of a policy measure has recently gained more relevance.

1.4. Plan of the study

This work is structured as follows. Chapter 2 highlights first the main impacts (both negative and positive) agriculture has on the environment. Secondly, the chapter introduces the methods of the nutrient balance calculations (the NBCs). The previous experience in using the NBCs is presented in this part of the study. This paper discusses the research on the NBCs as policy tools. Lastly, the problems and biases which obstruct the calculation methods are covered, followed by a description of the approach of the study.

Chapters 3 and 4 form the theoretical framework of the study. In Chapter 3 the agricultural production economics are described in a formal way, and the definitions for e.g. externalities, public goods, and property rights are reviewed. Furthermore, the theoretical solutions for internalising the externalities are illustrated. Chapter 4 scrutinises the characteristics of agricultural pollution. The main interest is to assess the economics of nonpoint pollution. The theory of nonpoint source pollution has recently been progressing, and some of these findings are reviewed here. The review outlines some of the policy methods available in controlling environmental pollution. The role of information in policy formulation is also discussed. Furthermore, the criteria for the choice of a control instrument are analysed in more detail.

The rest of the study scrutinises more closely the feasibility of the nutrient balance calculations in the Finnish conditions. In Chapter 5 the feasibility and the accuracy of the balance calculations are studied with respect to the weather conditions. This is done using the crop level data from the Agricultural Research Centre. In Chapter 6 a case study utilising the NBCs on dairy farms in Kainuu is conducted. A farm level model of using different policy instruments in pollution abatement is presented. The model is a simulation of a profit maximising farmer who produces silage. The leakage function of the model for nitrogen is based on the NBCs. Different scenarios are conducted to assess the policy instrument choice. The impact of weather factors on the choice of the policy instrument is also examined. Chapter 7 concludes and discusses the findings of the modelling in the framework presented above. Chapter 8 presents a summary of the study.

2. Agriculture and the environment

2.1. The impact of agriculture on the environment in Finland

2.1.1. Negative effects

Water quality problems caused by farming have become a major environmental policy issue in many countries, including Finland. This can be seen from the heated discussion that took place in early 1997 when the Ministry of the Environment put forward the recommendation for the nitrate directive. The extensive algae growth in the Finnish lakes and rivers and the Gulf of Finland during the warm summer of 1997 raised the eutrophication discussion into the limelight. Eutrophication of watersystems is the principal negative impact agriculture has on the environment in Finland. However, there are also other negative pressures caused by agriculture on the environment, e.g. atmospheric pollution (volatilisation), erosion, soil compaction, and detrimental effects on biodiversity.

Various Finnish studies (e.g. Rekolainen et al. 1992, Kauppi 1997) indicate that agriculture is the most important sector causing environmental burden to watersystems. Presently, agriculture accounts for over 1/3 of the total nitrogen load and over 1/2 of the phosphorous[1] load to watercourses in Finland. Although environmental policy measures such as the Finnish agri-environmental programme (FAEP, established in 1995), based on EU regulation EEC/2078/92, are having positive impacts on the reduction of nutrient load from agricultural sources, nutrient runoff problems are far from solved. The key reason for this is that the point pollution (PP) of industries or settlements is much more easily controlled than the nonpoint pollution (NPP)[2] of agriculture.

The total phosphorous load from cultivation of arable land is estimated to range between 2,000 and 4,000 tonnes per year, and, correspondingly, the nitrogen load between 20,000 to 40,000 tonnes per year. These figures do not contain the direct load from manure storage facilities (Rekolainen et al. 1992), and they are from the era before the FAEP.

[1] Usually, phosphorous is the limiting factor for algae growth in Finnish lakes and rivers.

[2] In this research the terms 'nonpoint pollution' and 'point pollution' are used although the longer terms 'nonpoint *source* pollution' and 'point *source* pollution' are also used in the literature.

Most of the agricultural production is concentrated to Southern and Western Finland. Thus, the impact of agriculture on surface waters in those areas is greater than in the northern regions. It has been estimated that 20% of lakes and about 55% of rivers in Finland are eutrophic. Indeed, the quality of the groundwater is quite good in Finland. The nitrate levels in groundwater supplies remain in most cases below 25 mg/l (MMM 1994), in contrast to the EU nitrate directive which sets an upper-level of 50 mg/l for nitrates in drinking water.

In addition, agriculture creates gaseous emissions. The most relevant ones are ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4). More than half of the NH_3 depositions originate from abroad, but local emissions also contribute to eutrophication and acidification (MMM 1995). In 1990 the ammonia emissions were about 50,000 tonnes per year (YM 1991).

Agriculture accounts for almost half of the anthropogenic (i.e. from human activities) nitrous oxide emissions in Finland. These total emissions were 18,000 tonnes per year in 1994. The total annual N_2O emissions were estimated at about 22,000 tonnes (Pipatti 1997).

The anthropogenic methane emissions in 1990 and 1994 were estimated at about 250,000 tonnes per year. The main sources were landfills and agriculture. Agricultural methane emissions (mainly from enteric fermentation) were about 100,000 tonnes. In contrast, the CH_4 emissions from nature (mainly from marshes) were 820,000 tonnes in 1994 (Pipatti 1997).

But, as mentioned earlier, agriculture accounts for the major part of the nutrient losses to watercourses, as can be seen in Figure 2. Therefore, the interest to find the proper ways to tackle the problem of controlling agricultural nutrient runoffs is growing; and this forms the basis for the present work.

Regional studies indicate that phosphoric load is considerably greater in Southern Finland than in other parts of the country. In areas close to certain rivers the load is at present over 2 kg/ha/year (MMM 1994).

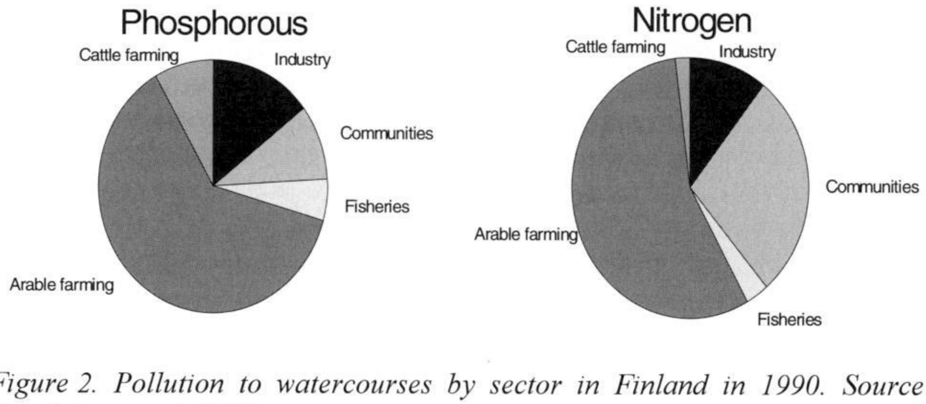

Figure 2. Pollution to watercourses by sector in Finland in 1990. Source: Rekolainen et al. 1992.

During the recent years 25 kg of phosphorous has annually accumulated into the soil (MMM 1995). The FAEP will bring this level down as the base level fertilising of phosphorous will be lowered to 15-20 kg/ha for cereals, for example. However, the research results from 1995 show just a slight decrease in the level of fertilising (Grönroos et al. 1997), but this is partly caused by the late implementation of the FAEP during that year. The coming years will show more clearly the true impact the programme has on the Finnish environment.

During the past decades, the intensity of input use in arable farming has risen. This can be seen by looking at the development in fertiliser usage. The fertiliser sales (equals the actual use) have risen considerably from the 1950s (Figure 3). However, at the beginning of this decade there has been a substantial decrease in fertiliser sales in terms of all three nutrients. It is important to note that the tax on fertilisers was raised considerably at that time. The tax revenue (FIM 650 mill. in 1992)[3] was used to cover the exporting costs of the oversupply of agricultural products. Some FIM 44 mill. of the tax revenue was used for agri-environmental investments (YM 1992).

In 1990 the tax on nitrogen in compound fertilisers was 0.15 FIM/kg, but it was raised to 2.90 FIM/kg in 1992 (Sumelius 1994). This seems to indicate that there exists a negative price elasticity on fertiliser purchases, although Ryhänen (1994) estimated the fertiliser prices to be fairly inelastic in Finland. Presently, there is no tax on fertilisers as Finland joined the EU in 1995 and the earlier tax was abolished.

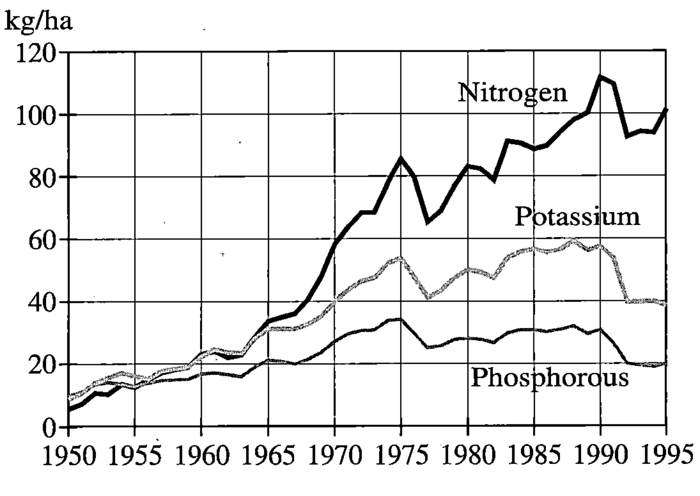

Figure 3. Fertiliser sales in Finland in 1950-1995.

[3] 1 FIM = 0.19 USD or 0.17 ECU in mid 1998.

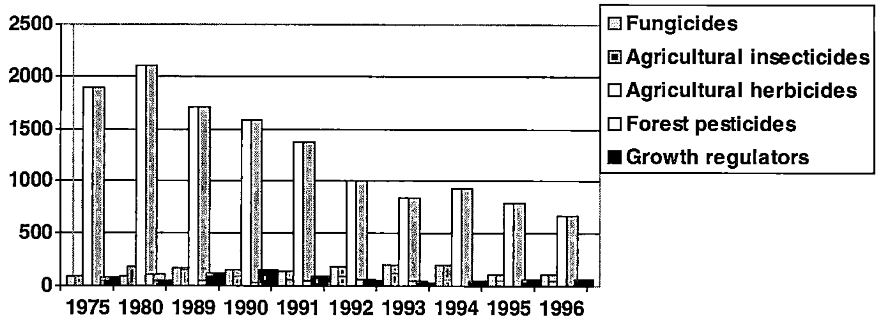

Figure 4. Pesticide use in Finland in 1975-1996; active ingredients. Source: KTTL 1997 ref. MT 1997.

With respect to the use of plant protection substances, the use of active ingredients has declined considerably during the past two decades (Figure 4). It is mostly herbicides which have been used in larger quantities in the Finnish agriculture, as the cool climate gives 'natural protection' against many pests and diseases common in the southern countries. In international comparison Finland has very low pesticide use (only 0.6-0.7 kg/ha). It is estimated that the pesticide emissions from arable land to watercourses are about 0.1-1.0% of the used pesticides (Laitinen et al. 1996).

The drastic decrease in the use of pesticides is partly due to the introduction of new pesticide products, such as small dose herbicides (MMM 1995). In addition, the training of farmers in the use of pesticides and testing of the spraying equipment, which are criteria for the agri-environmental support, have a similar impact on the use of pesticides.

The concentration of the cattle production in Central, Eastern, and Northern Finland has decreased the share of grass in the cultivation areas of Southern Finland. This has led to erosion, which increases the phosphoric load. Soil compaction has also increased erosion and surface runoff as rainwater is not absorbed to the soil but gets flushed away eroding soil particles and, thus, nutrients to watercourses (MMM 1995). Compaction has been caused by intensive production, especially mechanisation and the increase in heavy machinery. Both erosion and soil compaction reduce the soil fertility and lower the yields.

It is also estimated that changes in agriculture have caused altogether 300 plant and animal species to become endangered. In 1991 it was estimated that there are almost 1,700 species in Finland that are being endangered. One fifth of these are species living in connection to human activities, like in agricultural biotopes (YM 1995). The majority of endangered species in cultivated areas occur in the traditional biotopes, and others have become endangered as a result of changes in the cultivation methods.

2.1.2. Positive effects

In the long run an ever increasing share of the total economic value of agriculture will be derived from public goods produced by ecologically more beneficial farming. It is clear that agricultural practices maintain the agricultural countryside environment (ACE). It is an array of many public goods, amenities, and services. Agriculture should no longer be regarded merely as a producer of foodstuffs but a producer of many environmental public goods, of which agricultural *landscape* and *biodiversity* are the two most crucial ones (Aakkula 1996, Pirttijärvi et al. 1995).

Finnish rural landscape is greatly valued by Finns. (Of course, the beauty is in the eye of the beholder and not everybody values rural landscape or finds open field areas attractive.) Only nine per cent of the land area is cultivated. Particularly in Southern Finland open arable land areas are characteristic of the rural landscape. The greatest threat to the rural landscapes is caused by discontinuing of agricultural production and depopulation of rural areas. Agriculture is the backbone of the countryside in Finland.

Finnish rural landscape consists of many small features, and it is diversified in its details. Arable land is often located like a mosaic on lake shores or in the middle of forests. The size of open arable land areas varies from a few hectares to tens of hectares. The large open arable land areas in the river valleys in Southern Finland, which can be 500 to 1,000 hectares, form an exception to this. (MMM 1994).

In all, animal husbandry has created the most versatile types of landscape among the rural cultural landscapes in Finland. Natural pastures and woodlands used for grazing, as well as natural meadows cleared in forests and on shores for the harvesting of winter fodder with their barns and fences have been the dominating elements of the cultural landscapes of rural areas. At the beginning of this century cultivation by clearing and burning-over woodland was still the form of cultivation that dominated the landscape in eastern parts of the country. In the past few decades specialisation in cereal cultivation and decrease in animal husbandry, especially on the southern coast, has reduced the versatility of the rural landscape. As Finland is dominated by closed forest landscape, both small and large fields and pastures in all parts of the country are important for preserving an open landscape.

Species that favour the cultural environment of rural areas constitute a considerable share of the species in Finland. The managed traditional biotopes created by grazing and cutting have the greatest variety of species. Over a third of vascular plant species (400 to 500 species) have benefited from grazing and cutting. Apart from plants, managed natural meadows and pastures are important for butterflies and other insects (MMM 1994).

In the beginning of 1995 there were almost 2.9 mill. hectares of land and water areas which were under some kind of protection scheme (MMM 1995). This is about 8% of the total area of Finland. The proposal for the adaptation of the EU habitats directive aims to further extend these areas to cover over 10% of Finland under the 'Natura 2000' programme. This has aroused a number of arguments because the proposal (covering altogether 469,000 ha of land and water area) also comprises private land area (107,000 ha) under the scheme, and the economic use of this land would be restricted.

2.2. Nutrient balances as environmental indicators

2.2.1. Nutrient balance calculations

The nutrient balance calculations (NBCs), also referred as mineral balance, nutrient budget or excess nutrient calculations, have become very commonly used during this decade. The idea behind these calculations is rather old and simple notion; we compare the nutrients entering a system to the nutrients leaving a system at certain observation points. If the balance (i.e. the nutrient input minus their output) is positive, there is an oversupply of nutrients entering the system. In the opposite case, the negative balance indicates that the system has lost some of the nutrients in the production process. Huang and LeBlanc (1994) call this 'the nutrient mining'. This kind of calculation makes it possible to measure the amount of nutrients emitted to the environment – a very attractive point of view from the perspective of a policymaker.

The first major international effort to set a common basis for the calculation methods was the EU funded three-year (1993-1995) project, in which the balances were calculated for N, P_2O_5 and K_2O on regional and on national levels, as well as by farming type (Brouwer et al. 1995, Schleef and Kleinhanß 1994). Based on the principles of this project, the OECD (1997) started a harmonising work on the nutrient balance calculations when setting up international environmental indicators for comparison purposes (much like the PSE indicator). In addition, Eurostat (1997) has started collecting data from the EU member states for the NBCs.

In Finland the balances have been calculated already in 1970s and 1980s (Väisänen 1996). However, these have started to gain more relevance only during the past 10 years, especially in organic farming where the NBCs have become a fundamental factor in the farm production planning.

In many countries the balance calculations have been used to estimate the nutrient flows at farm, regional, and national level. As the information derived from the calculations is used for various purposes, many different methods were, and still are, used in calculating the nutrient balances. This is true both in Finland and in the international context.

Table 1. General ways to calculate the nutrient balances.

Nutrients in	Balance: Farm gate	Surface	Cattle
inputs			
1 Fertilisers	*	*	
2 Purchased manure	*	*	
3 Farm's own manure		*	
4 Purchased fodder	*		*
5 Farm's own fodder			*
6 Purchased seeds	*	*	
7 Farm's own seed		*	
8 Purchased livestock	*		
9 Biological fixation	(*)	*	
10 Atmospheric deposition	(*)	*	
outputs			
11 Crops sold	*	*	
12 Crops for feed		*	
13 Crop residues		*	
14 Livestock products sold (including manure)	*		*
15 N evaporation		(*)	
Nutrient surplus or deficit = (1+...+10)-(11+...+15)			
Nutrient efficiency ratio = (11+...+15)/(1+...+10)			

The * mark in the table indicates that the item is included in the corresponding balance formula.

Table 1 shows the schematics of three different methods which are used in Finland to calculate the nutrient balances: farm gate balance, surface balance, and cattle balance.

The idea behind the *farm gate balance* is to measure how much the farm uses purchased nutrient inputs in its production, and to compare these figures to the nutrient contents in the outputs sold (or given away) from the farm. The level of observation is the actual inflows and outflows of nutrients through the farm's 'gate'. In other words, the nitrogen and phosphorous[4] contents of the inputs entering the farm are compared to the corresponding figures exiting the farm in the outputs sold. Natural phenomena, such as atmospheric deposition of nitrogen in the soil and biological fixation by leguminous crops (in the parentheses)

[4] We limit ourselves to examining only N and P nutrients as these are the two most important forms of agricultural runoffs.

can also be included in the calculation[5]. The difference between the inputs and the outputs gives us the balance, i.e. a surplus or a deficit of nutrients.

The *surface balance* is calculated as a difference between the nutrients entering into and exiting from the soil surface. Within the inputs, the main components the calculation takes into account are the use of chemical fertilisers and manure. For the part of the outputs, the nutrient contents of the harvested crops are calculated. Subtracting the latter from the former gives a *gross* surface balance. If we also take into account the evaporation of the nitrogen in the manure and the fact that all nitrogen in the manure is in a infeasible form for plants to be utilised, we get a *net* surface balance. Usually a 30% estimate for non-available nitrogen is subtracted from the nitrogen in the manure then bearing then a closer link to the water pollution issues (e.g. Brouwer et al. 1995). In addition, atmospheric deposition is taken into account in the surface balance as well as biological nitrogen fixation and the use of sewage sludge in agriculture. The nitrogen fixation, in particular, is often significant at farm – e.g. in organic farming.

The *cattle balance* indicates the difference between the nutrients in the animal foodstuffs and the nutrients in the animal products leaving the production facility. Theoretically the balance should yield the amount of nutrients excreted in the manure during the production process. This type of balance calculation is used in the Finnish production control system run by the extension agencies (see e.g. Helander 1996).

2.2.2. Information in the nutrient balance calculations

In all three calculation methods the subtracted figure indicates a farm-specific nutrient surplus or deficit[6]. The surplus here means the amount of nutrients that the production process is unable to capture. Dietz (1992) defines the surplus slightly differently as the amount of nutrients actually lost on a farm minus the amount of nutrients nature can sustainably process at the location concerned. With this latter definition, though, we encounter the problem of defining the assimilative capacity of nature which, no doubt, varies by each location. Therefore, the former definition is used, also because of its closer link to the production process.

Theoretically the first two balances should yield the same result, but normally these balances differ. According to Brouwer et al. (1995), the difference probably stems from using normative coefficients in assessing the nutrient

[5] Strictly speaking these do not enter the farm through the gate. However, including these in the calculation enhances the information design of the calculation method.

[6] For simplicity we talk here mainly about surplus because that is the most common case.

content in animal manure, crop production, and livestock production. The nutrient surplus indicates in the short run the potential of polluting charges. In the long run, the averages of nutrient surpluses approximate the average farm environmental load.

Most likely the balance (i.e. the inputs minus the outputs) is positive, which means that some of the nutrients have accumulated in the soil, evaporated into the air, or leached to the watersystems. In organic farming the nutrient balances usually show a very small surplus or even deficit when calculated by the farm gate balance (Väisänen 1996). However, if biological nitrogen fixation, which is very difficult to measure reliably, is included, surpluses also emerge there (i.e. the second law of thermodynamics holds).

The NBCs lack the exact information on where the nutrients have exited. Therefore, the nutrient balance calculations are more like indicators than exact measurement devices of nutrient runoffs. The different ways of calculating the balances have their benefits, but also their difficulties, as will be discussed later.

Calculating nutrient balances at farm level is a rather straightforward procedure, but it can be a tedious one. Keeping account of all nutrients entering the farm causes extra work in animal husbandry farms. It is particularly difficult to keep track on the nutrients in the feeding stuffs, especially if these are not indicated in the feeding stuff packages.

Coefficients must be used in calculating the nutrient contents in the harvested crops and in the manure. Expressing unequivocal nutrient coefficients for different animals is a difficult task. Much of the magnitude of the coefficients depends on the feeding process and on the capability of the animals to process their feed.

The Finnish agri-environmental programme requires a farmer to carry out a nutrient analysis of manure once in every five years (MMM 1994). This measure makes it easier for the farmer to also assess the nutrient flows of a farm. A list of the Finnish nutrient coefficients in crops and manure is given in Annex 1. These coefficients are used in OECD and Eurostat calculations as well as in the Finnish regional calculations presented in Section 2.3.

In general, nutrient balances are rather ineffectual indicators unless related to agricultural land area or the total input of nutrients, for example. The nutrient use efficiency can be expressed by comparing the nutrient output to the input of them. The quotient tells how well the nutrients entering the system are captured by the production system.

The surpluses per hectare, as indicated by the balance calculations, vary a great deal across different countries, but the efficiency quotient is more even in different countries. The quotient, however, varies considerably by farming type. In Finland the efficiency ratio for cereal farms varies around 50-70%, whereas for dairy farms the corresponding figure remains in the range of 20-40% (Pirttijärvi 1996, Lankoski 1996).

In this connection, it is important to note the results of the Finnish study by Esala (1995) on crop level nutrient utilisation. He estimated that even under prosperous growth conditions only about 50% of the fertiliser nitrogen is utilised by the crop, 15% to 20% remains in the straw, and 30% accumulates into the soil. During a less prosperous year, the crop uptake is only 25% to 30% of the fertiliser nitrogen.

In order for a farmer to better realise the importance of nutrient bookkeeping as a good management tool, monetarisation of the nutrient surplus is needed. For example, a relatively small decrease of 10% in the nutrient surplus can be converted to the monetary value. Let us look at a farm which has surpluses of 100 kg/ha of nitrogen, 30 kg/ha of phosphorous and 70 kg/ha of potassium. If the prices for these nutrients are 4 FIM/kg for nitrogen, 9 FIM/kg for phosphorous and 2 FIM/kg for potassium[7], a 20 hectare farm reducing all these surpluses by 10% saves FIM 1,600 in nutrient values.

We have to note that the nutrient balances do not tell precisely where the nutrient losses emerge from and to what extent the losses leach to the watercourses or discharge into the air. Nevertheless, substantial nutrient losses tell the farmer that he has an efficiency problem in using the inputs in his disposal; and that there is also an environmental problem. A farmer then faces many options to choose from to reduce the nutrient losses. Consequently, nutrient balances elicit a farmer to opt for environmentally sound management practices. Therefore NBCs act as an information instrument in pointing out the environmental bottlenecks in the farm management.

As the nutrient balance calculations alone are usually insufficient to tell what to do, the balance sheet should be closely integrated to other farm planning and monitoring procedures, such as soil analysis. Götz and Zethner (1996) give an extensive (for Austrian conditions, but also applicable to Finland) list on the measures through which the nutrient surplus can be reduced:

- minimising the use of commercial fertilisers
- improving the use and application of manure
- soil analyses of N_{min} and N_{org} to plan fertiliser use
- increasing consideration of nutrient return from crop residues
- reducing feed imports (e.g. protein feeds)
- enlarging grasslands, converting arable land into meadows
- protecting the soil by a vegetation cover (crop rotation and underseed)
- reducing ammonia emissions through measures concerning livestock

[7] These prices are calculated from compound fertilisers having the highest amount of the corresponding nutrient.

- reducing the use of nitrogen fertilisers to curb denitrification
- tillage and landscape planning measures to minimise soil erosions
- supporting biological farming which uses substantially fewer commercial nutrients

2.3. Experiences and research on nutrient balances

Calculating the net surface balance levels out the differences of nutrient usage efficiency between cereal farms and animal husbandry farms. In the studies of Brouwer et al. (1995) and Schleef and Kleinhanß (1994) 30% of the nitrogen in manure is subtracted to get the net surface balance calculation. This net balance is more closely related to the problem of nutrient losses to watersystems. However, this approach biases the real problem of agricultural nutrient losses, as losses to the air and partly to the ground are neglected.

According to the studies of Brouwer et al. (1995) and Schleef and Kleinhanß (1994), the nitrogen losses from cereal farms are about one fourth of the losses compared to dairy farms. The nutrient balances from poultry and pig farms show the highest nutrient losses.

The magnitude of nutrient losses to the environment varies between countries and within countries. For example, in the Netherlands the nitrogen losses per hectare of arable land are, on average, almost three times higher than in Germany, and six times higher than in Finland (Figure 5).

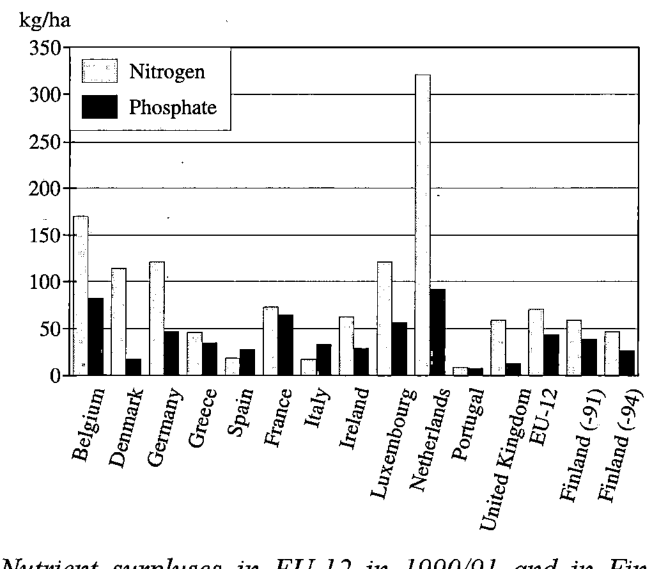

Figure 5. Nutrient surpluses in EU-12 in 1990/91 and in Finland 1991/94. Source: Brouwer et al. 1995 and Pirttijärvi 1996.

In EU-12 there also occurs considerable regional variation, i.e. within countries, in the nutrient balances. The main reason for high nutrient losses stems from intensive animal husbandry, and thus from the use of manure. Large scale livestock farms, which rely on imported feeding stuffs, like many pig farms in the Netherlands, produce huge amounts of manure. Instead of being a valuable production input, manure has turned out to be a mere waste in many cases (Dietz 1992).

Finnish nutrient balance calculations also point out that nutrient losses from the animal husbandry farms surpass those from the cereal farms (Lankoski 1996). Regional variation in nutrient losses is related to animal density. In intensive milk production areas the nitrogen losses are the highest. Map in Figure 6 shows how the nitrogen surpluses have changed in Finland from 1991 to 1994.

Exceptional weather conditions, such as the drought in some parts of Finland in 1992, also lower the usage efficiency of nutrients. Phosphorous surpluses are fairly similar in all parts of the country, and they have decreased in the past few years; which is a result of the decrease in the use of phosphorous fertilisers in general (Pirttijärvi 1996).

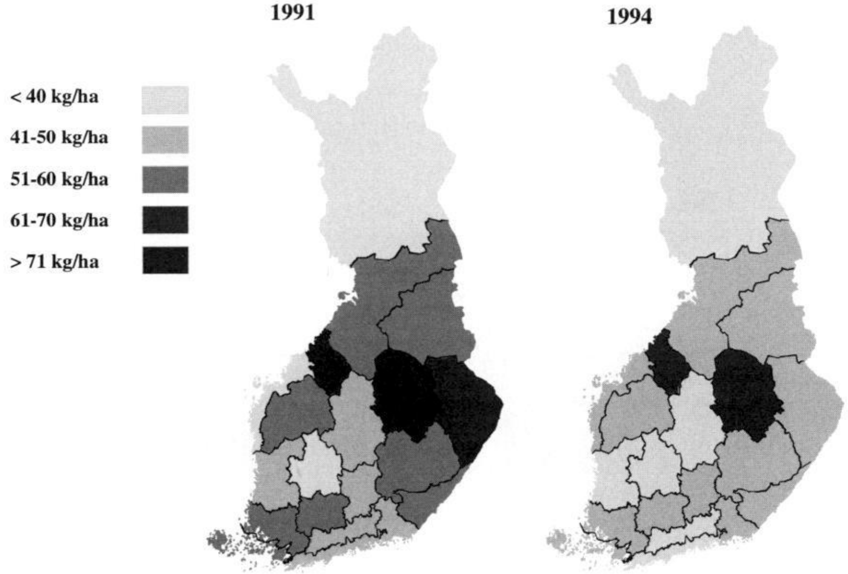

Figure 6. (Net) nitrogen surpluses in Finland in 1991 and 1994. Source: Pirttijärvi 1996.

OECD has done work on nutrient balances amongst the other 12 environmental indicator areas (OECD 1997).[8]

Figure 7 shows the Finnish nitrogen surpluses over an 13-year period according to the OECD schematics. As can be seen in the figure, a downward trend takes place in the nitrogen surplus during this period. The slope of the regression curve estimated from the series and shown in the figure is -3.0, i.e. nitrogen surplus per hectare is decreasing, on average, close to 3 kg/ha per year, although in the last few years there has been an upward trend. The same kind of trend is taking place in most other OECD countries (OECD 1997).

Another evident feature in the chart is that the year 1987 has a considerable deviation from the trend line (this, in part, makes the regression line steeper). This is due to the unfavourable weather conditions for crop growth during that year. Nitrogen application via fertiliser and manure were about the same as in 1986, but the uptake of nitrogen by the crops was about 30% lower. Therefore, the nitrogen surplus in 1987 was almost 20 kg/ha higher than in 1986.

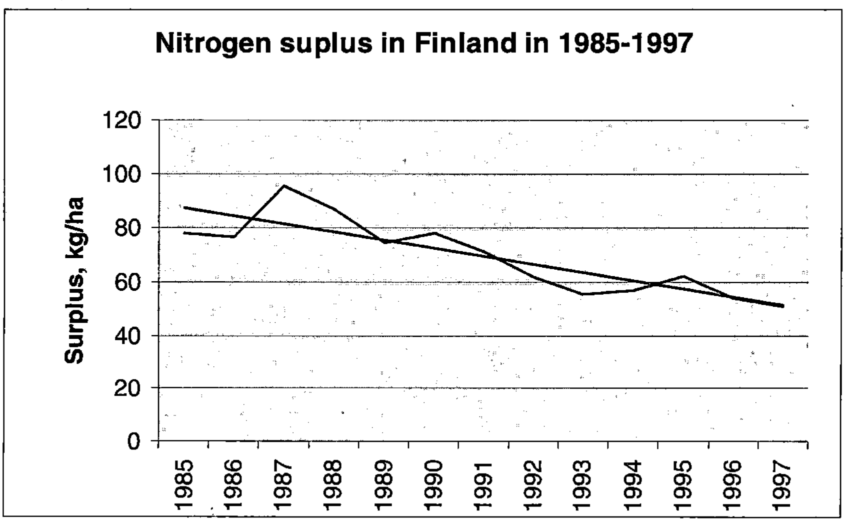

Figure 7. Nitrogen surplus and its trend in Finland in 1985-1995.

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8 The method used in OECD calculations is the surface balance method taking into account ammonia losses from manure which take place before spreading. The other indicator areas are: agricultural pesticide use, agricultural water use, agricultural land use and conservation, agricultural soil quality, agriculture and water quality, agricultural greenhouse gases, agriculture and biodiversity, agriculture and wildlife habitats, agricultural landscape, farm management, farm financial resources, and socio-cultural issues in relation to agriculture.

At the policy level this kind of weather-influenced deviation in nutrient losses has important implications. For example, if there had been a policy measure established in 1986 with the objective of reducing the nitrogen surplus, the 1987 balance figure could have been incorrectly interpreted as an indication of policy failure. This is very important to note in international context where uniform policies are often sought for, and national characteristics are often not fully understood.

This type of problem could be avoided by calculating, for example, three-year moving averages, but then the effect of a particular policy becomes difficult to pinpoint as the impact of a policy measure is smoothened. Alternatively, some sort of 'tolerance limits' for the random factor in the nutrient surplus could be introduced. However, there is no unambiguous way for setting the tolerance limits as the annual changes can be considerable. And, if the tolerance gap is too wide, again, the effect of a policy measure will be shadowed. This phenomenon will be discussed later in Section 4.1. in the context of nonpoint pollution.

In the Finnish pork production some 500 farms are involved in the production planning of the Centre of Rural Advisory Services (MKL). In the MKL production supervising system nutrient balances have also been calculated since 1995 (Helander 1996). The relationship between the production costs and utilisation percentage of nutrients is illustrated in Figure 8.

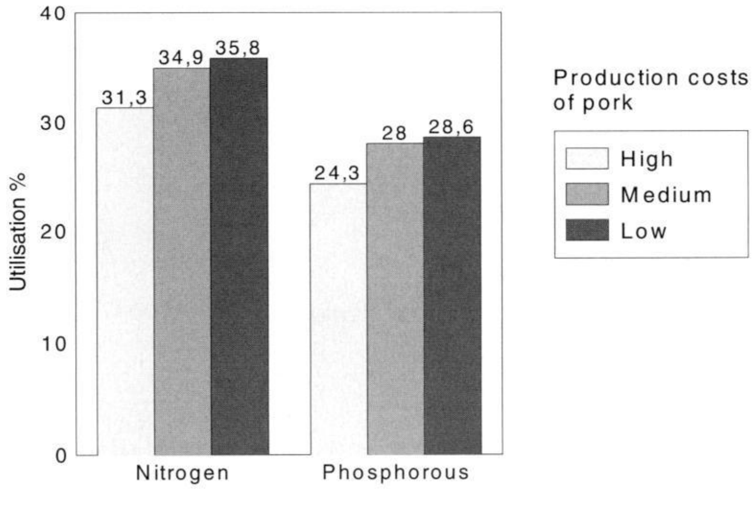

Figure 8. The relationship between production costs and nutrient utilisation at Finnish combination pork farms in 1995. Source: Helander (1996).

The nutrient balance calculated using the MKL data depicts the relationship between the nutrients entering the herd and exiting it in the form of meat (i.e. the cattle balance of nutrients). There is a positive correlation between nutrient utilisation and production costs; in other words, the lower the production costs are the better the farm's utilisation of nutrients is. This demonstrates a link between good farm management and good environmental management, although this relationship is not very clearly pronounced.

In the Finnish pork production the average utilisation percentage for nitrogen is 35 and for phosphorous 28 (Helander 1996). In cattle production the corresponding figures on farms participating in the production supervision in 1996 were 24% for nitrogen and 31% for phosphorus. The average nutrient surpluses by cattle balances were 114 kg of nitrogen and 15 kg of phosphorous per animal per year (Mälkiä 1997).

2.4. Nutrient balances as policy tools

In addition to the information dimension of the indicator, the NBCs are being applied in environmental control policies, too. The Netherlands (together with Denmark) is the pioneer in taking advantage of the NBCs in agri-environmental policy. In 1998 a new manure law based on a levy system on farm level nutrient surpluses will come into force (MLNV 1995). The main principle of the law will be a levy on surplus phosphate on farms having more than 2.5 livestock units (LU) per hectare (in 2002 this will be lowered to 2.0 LU/ha). table b presents the contents of the manure law. The farm will be charged for the part of the phosphate surplus which exceeds a threshold value of 40 kg/ha in 1998. Later, the threshold levels will become stricter. Two levels of levies will be introduced; a lower 5 NLG/P_2O_5 kg and a higher 20 NLG/P_2O_5 kg, depending on the magnitude of the surplus[9]. Furthermore, nitrogen balances are introduced, but no levies for them are set. Nitrogen surpluses are controlled indirectly with the phosphate levy system.

It is estimated that the average income losses in the year 2000 will be on pig farms NLG 7,000 and NLG 1,000 on cattle farms. At the beginning, about 50% of the farms will face the levies, and later, as the threshold levels will be lowered, almost every livestock farm will be covered by the levy system. In the short run, this system will not completely solve the country's manure and nitrate problem, but yet it is a serious attempt to tackle the most serious problem of the Dutch agriculture.

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[9] 1 NLG (Dutch guilder) = 0.52 USD in mid 1998. Taking an example, a 50 hectar farm which has a 55 kg/ha phosphate surplus will in 1998 have to pay a levy of about NLG 7,500 levy (USD 3,870 or FIM 20,400).

Table 2. The thresholds of phosphate surpluses and spreading norms in the Netherlands. Source: MLNV 1995.

	1998	2000	2002	2005	2008/10
Allowed surplus, P_2O_5 kg/ha	40	35	30	25	20
Low P_2O_5 -levy, 5 NLG/kg	40-50	35-45	30-40	25-30	(1)
High P_2O_5 -levy, 20 NLG/kg	over 50	over 45	over 40	over 30	(1)
Nitrogen surplus*	300	275	250	200	180
Spreading norm, P_2O_5 kg/ha		85	80	80	80
for grass	120				
for grain	100				
Livestock unit density per hectare	2.5	2.5	2.0	2.0	(1)

* for grass; nitrogen deposition and mineralisation are not included.
(1) will be decided later.

The development of the NBCs as policy tools has also created research on their use as policy tools. For example, Berentsen and Giessen (1993) studied the efficacy of a control policy based on the NBCs. They present an agri-environmental linear programming model for dairy farms. They consider four different policies (Table 3) available to reduce nitrate emissions on three different farms.

As this research was conducted before the introduction of the manure law, the limits (e.g. 400 kg/ha N-surplus) do not correspond with the criteria in the forthcoming legislation. Yet, some useful information is gained here. From the table above it becomes clear that the different policy instruments work better on different goals. If the objective is the reduction of NH_3 emissions, it is obvious that the policy involving closed manure storage coupled with injection of manure to the soil is by far the least-cost policy. However, concerning the reduction of other N-losses, the levy on nitrogen losses (calculated by the nutrient balance) appears to be the most cost-effective policy, although the results are to some extent ambiguous in this respect.

Fontein et al. (1994) developed an econometric dual-approach model on the Dutch pig sector to assess how levy systems (on feed or on surplus) would reduce nitrogen surpluses. They conclude that a levy on the surplus of nitrogen is a more cost-effective way to reduce the output of nitrogen than a levy on feed. In the case of a levy on feed the profit reduction associated with a 40% reduction in the nitrogen excretion is larger than in the case of a levy on the surplus.

The technical and environmental efficiency of Dutch dairy farms was studied by Reinhard et al. (1997). They concluded that, although the technical efficiency was very high (0.89), the environmental efficiency measured by nitrogen surplus was fairly low (0.38). Their data also shows that intensive farms are

Table 3. The effect of different policy instruments. Source: Berentsen and Giessen 1993.

	Farm	closed manure storage+ injection	legal N-limit (400 kg/ ha)	levy on fertiliser-N (NLG 4/ kg)	levy on N-losses (NLG 4/ kg)
Net decrease of labour income (NLG.):	1	2,299	0	1,589	890
	2	2,694	5,730	6,881	8,006
	3	3,433	13,910	10,830	7,759
Reduction of NH_3-emission (kg N/ha):	1	7.4	0	1.5	1.4
	2	18	16	16	18
	3	29	36	32	31
Decrease of income/kg reduction of NH_3 (NLG/kg):	1	13	-	44	26.5
	2	6.2	14.9	17.9	18.5
	3	4.9	16.1	14.1	10.4
Reduction of other N-losses (kg/ha):	1	0	0	42	40
	2	0	112	114	130
	3	0	267	245	221
Decrease of income/kg reduction of other N-losses (NLG/kg):	1	-	-	1.6	0.9
	2	-	2.1	2.5	2.5
	3	-	2.2	1.8	1.5

environmentally more efficient than extensive farms with respect to the generation of nitrogen surplus, although this tendency is not very pronounced.

In Finland Lankoski (1996) has studied the cost-efficiency of a nitrogen fertiliser tax compared to an effluent tax based on nutrient balances. The cost-efficiency of a direct instrument, effluent tax, was better at lower abatement levels (i.e. at 10% - 30% abatement levels). However, at higher abatement levels the tax on nitrogen input was more cost-effective. If nitrogen surpluses are high, taxing inputs is a more cost-effective policy.

The nutrient balance calculations are beginning to be used in agri-environmental policy. However, so far there is little evidence to support the NBCs-based levies as a more effective policy choice over the conventional input or output taxes. The relative superiority of a policy instrument depends on different factors, such as the expected and realised costs functions (see Section 3.3.). This leads us to the question: How reliable is the methodological base of the NBCs? The policy evaluation is biased in cases where the expected and the realised costs and benefits differ. The following section underlines some problems that hinder the use of the NBCs both for farm level assessment and in policy analyses.

2.5. Inherent problems of the nutrient balance calculations

Even though the nutrient balance calculations seem to be a very attractive way of circumventing the nonpoint pollution problem, thus allowing the policymaker to use economic instruments such as levies on nutrient surpluses (as in the Dutch case), some inherent limitations still exist. These limitations can be calculation specific, information specific, or application specific.

The *calculation specific problems* are those related to the data accuracy and to the components which need to be included in the calculation. For example, a set of coefficients must be used in calculating the nutrient contents in the manure and in the harvested crops. Expressing unequivocal nutrient coefficients for the nutrient content in manure for different livestock is a difficult task. Yearly changes, as well as regional or international differences, can be considerable. Much of the magnitude of the coefficients depends on the feeding process, and on the efficiency of the animals to process their feed. The same kind of variation, but to a lesser extent, is present in the nutrient uptake coefficients of different crops.

The biological nitrogen fixation is the cornerstone of nutrient economy in organic farming, but, in most cases, it bears negligible meaning in conventional farming. In addition, nitrogen deposition can be as high as 35-40 kg/ha in Central Europe, but in Portugal or in Finland it remains in the range of 5 kg/ha (Brouwer et al. 1995, Ympäristö 1997). Thus, the type of farming as well as local conditions must be kept in mind when assessing the relevance of the factors to be included in the NBCs.

One of the major questions concerning the feasibility of nutrient balances in policy use is the problem of external factors, such as the weather and soil type, that influence the build-up of the surplus. Because of the weather, yield levels from year to year vary considerably in Finland which is located at the margin of cultivation for many crops. The question of how important a role the weather has on the build-up of nutrient surpluses is studied in more detail in Chapter 5.

In some of the NBCs (e.g. Brouwer et al. 1995 and Schleef and Kleinhanß 1994), 30% of the nitrogen of manure is subtracted to derive the net surface balance, as explained in Section 2.2.1. Calculating the net surface balance of nitrogen levels out the differences of the nutrient usage efficiency between cereal farms and animal husbandry farms. The net balance is more closely related to the problem of nitrogen losses to watercourses. This approach, though, distorts the real problem of agricultural nutrient losses, as losses into the air and partly to the soil are ignored. At farm level the 'evaporation percentage' may well be quite something else than 30% due to differences in production techniques.

The NBCs also contain *information specific problems*. In general, nutrient balances are not very useful indicators unless related to the agricultural land area, number of livestock units, or the total input of nutrients, for example. In the international context the choice of the elements in the denominator plays an important role. If all possible agricultural land is added to the denominator, the balance per hectare gets smaller (i.e. we get a more favourable figure). Indeed, it would be better to consider only the active arable land area (where manure or fertilisers are spread) in the agricultural land area to produce more meaningful figures.

Furthermore, we have to note that the NBCs do not indicate very precisely where the nutrient emission occurs, i.e. to what extent the losses leach to watercourses or discharge into the air. Therefore, nutrient surplus is more of an indicator than an exact measure of nutrient emissions. Substantial nutrient surpluses tell the farmer that there is an efficiency problem in using the inputs at his disposal; and that there is also an environmental problem. The farmer then faces a number of options for reducing nutrient losses. From this perspective, the NBCs can be seen as an information tool in the production planning.

The NBCs alone usually convey insufficient information to the farmer in terms of what to do and how to improve the nutrient utilisation on the farm. Therefore, the balance sheet should be closely integrated to other farm planning and monitoring procedures. As a result, nutrient balances will influence the farmer to choose environmentally sound management practices; and, thus, the NBCs work as an information policy instrument.

The question concerning the purpose of the actual use of the NBCs falls into to the set of *application specific problems.* The NBCs can be used for different purposes, and it is the purpose that justifies the way of calculating the balance. That is, there is no single correct way to do the calculation. If the NBCs are used only as a production planning tool in the farm management, the completeness and use of the best estimates is more important than strict administrative comparability, as put forward by Lord and Anthony (1997).

However, if we need to get comparable figures, the definitions of the calculation methods must be set so that the database is consistent and the coefficients are derived using consistent methods. This is the basis for the work the OECD (1997) and Eurostat (1997) are doing for their parts. Indeed, we have to be very careful in interpreting the results if different methods are used.

In administrative use the balance calculations face the reliability bias due to farmers' strategic behaviour. If farmers know that the NBCs are to be used, for example, for setting a levy based on the nutrient surplus, they tend to overestimate the quantity of outputs and underestimate the quantity of inputs. This is a hidden information setting, and will easily lead to adverse selection (see Section 4.). The enforcement costs for policymaker to verify the NBCs' information, e.g. using the information from the taxation database, will not be insignificant.

Hence, the actual use of the nutrient balance calculations also influences the accuracy of the data. If farmers consider that the NBCs provide them with new and valuable information they have an incentive to report the balance sheet as accurately as possible. But if they fear they are to be penalised by the outcome of the calculation, they have an incentive to report less reliably. This is an issue that needs to be taken into consideration in the final policy analysis.

2.6. The framework of the study

The theoretical framework of this study is presented in Figure 9. As noted in the previous sections, producing agricultural commodities has negative effects on the environment, along with the positive ones. However, the market mechanism lacks the ability to capture these effects because most of the environmental goods are public goods, the use of which is nonexclusive and nonrival. The task is, consequently, to define agri-environmental policy instruments to internalise these effects in such a way that social welfare is maximised (or improved).

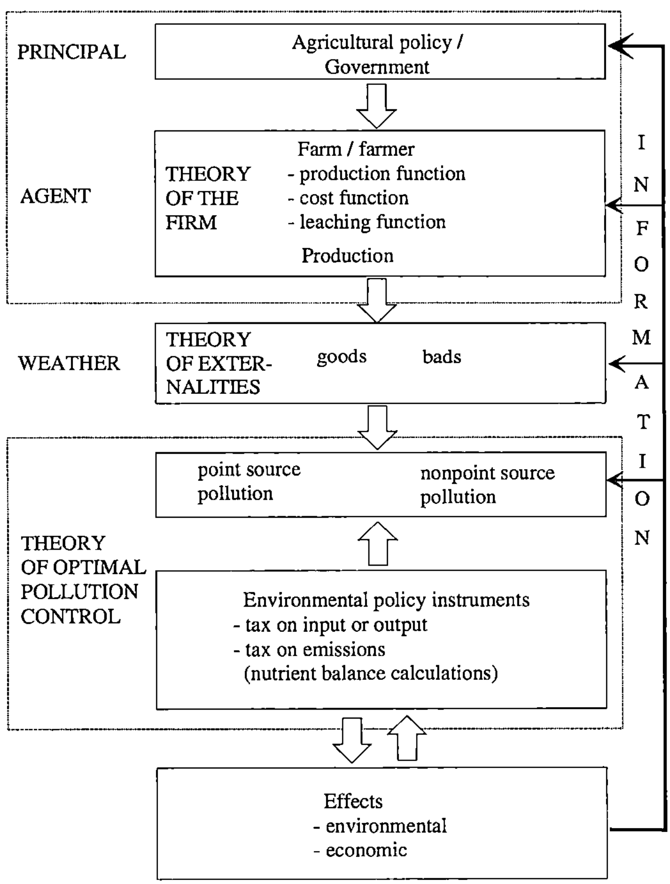

Figure 9. Theoretical framework of the study.

When economists are talking about control policies they usually mean the principal-agent approach (Siebert 1995). This means that the policymaker, the government (in a usual case) as the principal is seen to be the authority reflecting the people's view on the desired environmental quality, and it sets specific quality goals for it. The principal attempts to influence the decisions of the agents, e.g. the farmers, in such a way that the target is eventually reached. Thus, the task for the principal is to design such policy instruments (i.e. institutional arrangements and/or incentives) which make individual agents' behaviour contribute according to the specified overall target.

The principal usually sets taxes or standards on inputs or emissions to make the marginal private benefits meet the marginal social cost. However, emission-based first-best solutions are seldom possible in agricultural pollution control due to its nonpoint nature. In nonpoint pollution problems, the monitoring of individual polluting actions is difficult and such actions cannot usually be inferred from observed ambient pollution because of the following two reasons (Segerson 1988, Helfand and House 1995):

1) ambient pollutant levels have a random distribution that is contingent on the level of abatement undertaken; and/or
2) the actions of several polluters contribute to the ambient levels and only combined effects are observable

Because of these reasons, the solutions to control point source pollution problems do not work in the case of NPP. Therefore, workable policy instruments for NPP must recognise these characteristics.

In addition to the nonpoint problem of agricultural emissions, there exists another problem that makes agri-environmental policy formulation difficult. The *weather* is an especially important factor influencing the growth of crops in Finland. Therefore, emissions from the agricultural sector can be very extensive if the growing conditions are unfavourable, resulting in inefficient nutrient utilisation. The effects of weather factors on nutrient surpluses are studied using data from the research stations of the Agricultural Research Centre. A second set of data is from dairy farms in Eastern Finland. This latter data is used in assessing the relationship between farm economic factors and nutrient use efficiency.

The information from crop level data is incorporated into farm-models to assess agri-environmental policy formulation. A cost-efficiency model for silage production is constructed. In the final policy assessment one important factor to be taken into consideration is how well the system dispatches information and whether this has an effect on the policy ranking, i.e. information is in a key role when assessing policy alternatives.

3. Producing agricultural goods and bads

3.1. Producing agricultural commodities

As discussed earlier, agricultural production has manifold impacts on the environment. To be able to assess the economic implications of these impacts we have to inspect the production economics of the agricultural production process. Production is an activity of combining and using goods and services called *inputs* in a technological process to produce goods and services called *outputs* (Gravelle and Rees 1990). These goods are used privately or sold on the markets. By definition, trade is an exchange process of goods and services (both inputs and outputs) on the markets.

The neoclassical approach of economics, which is used in this study, sees agriultural producer to toil on his (or her) farm aiming at profit maximisation (alternatively, he can also aim at minimising costs). The profit is the difference between the production costs and the price farmer gets for the outputs sold from the farm. Of course, the profit maximising approach can (and should) be questioned and criticised. Ritson (1985) clarifies this issue stating that we are concerned with what happens instead of what should happen – whether or not it is in the 'society's interest'. Profit maximising approach is a hypothesis concerning the behaviour of a farm firm. Based on this, we can identify changes in the marginal costs and revenues when a change takes place e.g. in the prices of the inputs the farm uses in its production process.

According to Ritson (1985), the four main factors affecting the change in the firm's supply of agricultural outputs are: the state of technology, the price of a product, the price of competing products, and the price of inputs. A change in any of these will have an effect on the farm profit. In policy analyses, simulations are made to estimate the magnitude of the profit change when one or more of these four factors are allowed to vary. The following example clarifies the issue.

In Figure 10, the TC1 curve indicates the total costs of producing Product A before the price change of the input. The profit is indicated by the difference of total revenue and total cost, i.e. TR - TC1. The corresponding marginal costs curve is MC1, and the maximum profit level of Product A is Q1, i.e. at the point where marginal cost and marginal return (MR) meet.

If a price change, e.g. of fertilisers, takes place, the total cost curve of an agricultural commodity, *ceteris paribus*, shifts to the left, i.e. from TC1 to TC2. This means that if the policymaker wants to set a tax on fertilising inputs when hoping to reduce agricultural nutrient runoffs, the total costs of producing any given level of Product A will rise. In addition, since the quantity of fertilisers used decreases along with the level of output, the rise in the total cost will itself decrease as the output decreases. In other words, the marginal cost of producing

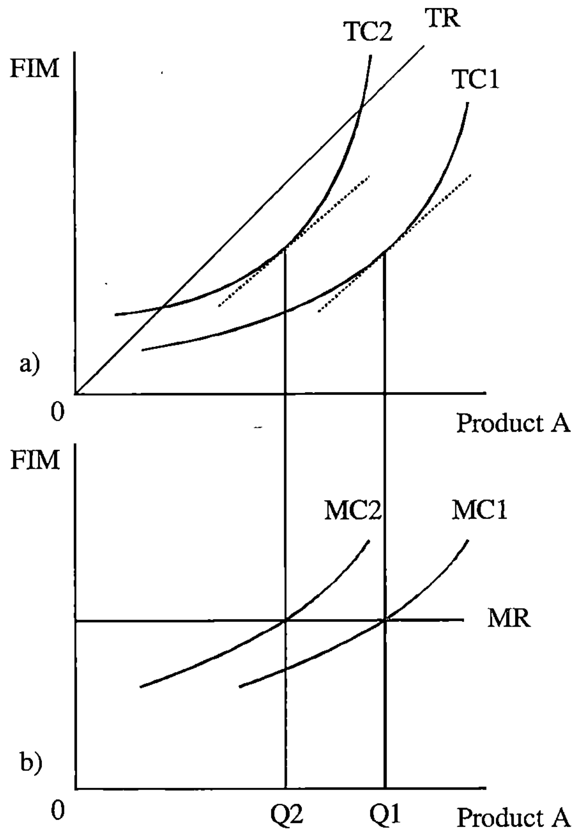

Figure 10. The effect of a price change on production. Source: Ritson (1985).

any level of Product A also rises, i.e. MC1 shifts to MC2, and the maximum profit levels of the production of Product A will move from Q1 to Q2 (Figure 10b). Thus, an increase in the price of one input, other things remaining unaltered, will cause a decrease in supply.

The effects of technology change, output price change, and change in the price of competing products can be analysed in the same way. For example, productivity improvement due to technical innovation will result in a decrease in total costs. Further, a fall in the price of a competing product will cause a rise in supply. Finally, the change in the price of a product shifts the total revenue line. A rise in the product price increases the quantity of Product A produced, i.e. the farm is willing to supply more of a good when the price of it increases. The responsiveness of the quantity supplied to changes in the price of a product is called the price elasticity of supply. More formally, $E_s = (\delta Q/Q)/(\delta P/P)$.

Let us now look at a situation where a farm produces one crop using two inputs, e.g. mineral fertiliser and manure, in its production process. Following Debertin (1986) we get;

(1) $$\pi = pf(x_f, x_m) - w_f x_f - w_m x_m$$

where π = profit
p = price of output
$f(x)$ = production function
x_f = quantity of nitrogen fertiliser input in fertiliser
x_m = quantity of nitrogen fertiliser input in manure
w_f = price of nitrogen fertiliser
w_m = price of nitrogen in manure

To find an optimal solution to this maximisation problem the equation is differentiated first with respect to x_f and then to x_m to get the two first order conditions, i.e.:

(2) $$\pi_1 = pf(x_f) - w_f;\ \text{and}$$

(3) $$\pi_2 = pf(x_m) - w_m$$

setting these to zero and rewriting:

(4) $$f(x_f) = w_f / p = x*_f$$

(5) $$f(x_m) = w_m / p = x*_m$$

These mean that when the profit is at the maximum, the marginal product equals the ratio between the input price and output price. The sufficient second order conditions to ensure a local maximum for the concave production function are:

(6) $$\delta^2\pi / \delta x_f \delta x_f < 0$$

(7) $$\delta^2\pi / \delta x_m \delta x_m < 0$$

(8) $$(\delta\pi / \delta x_f)(\delta\pi / \delta x_m) - (\delta\pi / \delta x_f \delta x_m)^2 > 0$$

However, the production process is usually more complex, because in this setting the production is assumed to have no external effects either on the environment or on the utility of some other producer or consumer. Usually these effects exist, i.e. the production creates *externalities*.

3.2. External effects of agricultural production

3.2.1. Externalities and market failure

As stated above, the aim of a producer in his production process is to create goods for consumption either for his own use or to other people's use through the markets. The rules of thermodynamics, however, dictate that it is inevitable to lose some input flow along the production process (even if this is actually not lost here). A classic example of this is pollution, e.g. nutrient runoffs from agricultural fertilisers. Therefore, pollution is not merely a symptom of a market failure, but a pervasive phenomenon in itself. This viewpoint is called a *materials balance approach* (Pearce and Turner 1990, Siebert 1995). Market failure exists when markets are not maximising the social welfare. In this context, economists usually talk about production (or consumption) externalities[10], which exist when an activity of one agent causes an uncompensated change in some other agent's welfare.

In an economy of two activities i and j, externality exists if the output Q_i in activity i depends on the output Q_j or on the inputs R_j of the other activity (Siebert 1995). Thus;

$$(9) \qquad Q_i = F_i(R_i; Q_j, R_j)$$

where $\delta Q_i / \delta Q_j \neq 0$ or $\delta Q_i / \delta R_j \neq 0$

It must be noted here that the output i often decreases when facing externalities, but it can also increase. If the output of good i increases while the output of good j is rising, *positive externalities* exist. Similarly, if the output of good i decreases while the output of good j is rising, *negative externalities* will prevail. Therefore, we can also talk about positive externalities; not only the negative ones.

[10] Vatn and Bromley (1997) redefine the existence of externalities from a different viewpoint as; "*given the markets*, the presence of externalities can be interpreted as a rational result and thus cannot properly be called a 'failure' of the market".

But why do we seldom encounter positive externalities? The answer is simple: if somebody creates a positive externality, he tries to sell it (in case he has the property right on the externality, see next section) to the person who benefits from it, and we are talking about ordinary goods sold on market. On the other hand, the producer has a tendency to neglect the negative externalities, because taking care of them (pollution abatement) imposes additional costs to the production.

At the competitive markets the producer finds the equilibrium between supply and demand at the point where his marginal *private* costs equal the marginal private benefits. In the case of externalities, the producer's supply curve does not reflect marginal *social* costs which exceed the marginal private costs. This will entail excessive production of the commodity, and too much pollution is produced.

If the environmental effects of the production are taken into account, the emissions can be considered to be joint outputs of the production, which are then emitted to the environment. Then, the emission function (*E*) can be written as;

(10) $$E_i = H_i(Q_i), H_i' > 0, H_i'' > 0$$

This emission function assumes that, at a given technology, the quantity of emissions increases proportionally or progressively along with output Q_i, but it excludes the case in which the quantity of pollutants increases regressively (Siebert 1995). If $H_i'' = 0$, the emission function is a linear curve, and if $H_i'' > 0$, the emission function is strictly convex.

The production function Q_i is characterised by a declining marginal product and does not distinguish among different production factors. For simplicity, only one type of resource *R* is assumed (instead of having manure and fertiliser as in Equation *(1)*.

(11) $$Q_i = F_i(R_i), F_i' > 0, F_i'' < 0$$

Compiling these two equations above the emission function is rewritten to;

(12) $$E_i = H_i[F_i(R_i)] = Z_i(R_i), Z_i' > 0, Z_i'' >=< 0$$

This function defines the statement that the pollutants are joint products of the input. Overuse of inputs, and consequently environmental deterioration, is one symptom of the failure of the market system to internalise the external effects agriculture has on the environment. From the social point of view agriculture produces too much pollution (e.g. nutrient runoffs) since it ignores the impact the pollution has on the environment (e.g. on watercourses).

3.2.2. Public goods and property rights

According to a definition by Randall (1987), environmental goods, such as environmental quality, are *public goods* which are nonexclusive and nonrival. The definition nonexclusive means that the users of the environment cannot be prevented from benefiting[11] from the public good in question. Nonrivarly means that benefiting or consuming of the public good does not reduce the possibility for someone else to use the public good. In practise, the consumption of a public good can to some extent have an effect on the quantity of the supply available to somebody else if congestion of the public good takes place.

As environmental quality is a public good, farmers using nitrogen fertilisers cause nitrate pollution of watercourses because the price charged for the fertiliser usually does not reflect the cost of water pollution to the society (OECD 1994). Therefore, market mechanisms do not capture the external effects of the production if no one owns (exclusively) the right to use the environment and no one claims for compensation based on these harmful effects, i.e. the property rights of the environment are inadequately defined.

In general, environmental externalities can emerge because of failures in the *property rights* system (Hanley 1991). Property rights are a set of rules (rights, privileges and limitations) which specify the *use* of scarce resources and goods. It is the right to use a resource that property rights relate to (Pearce and Turner 1990). Tietenberg (1992) defines four characteristics for an efficient structure of property rights; universality, exclusivity, transferability, and enforceability. As long as property rights are well defined, trade between agents will result in an efficient allocation of externality (Varian 1990). However, problems arise when one tries to define property rights for public goods, such as the environment. Siebert (1995) presents two approaches to the environmental problem; the public-goods approach and the property-rights approach.

The *public-goods approach* states that, if environmental quality is a public good, property rights cannot be defined and government intervention becomes necessary. This approach claims that, if property rights are adequately defined, optimal allocation will be reached through private decisions, and government intervention becomes necessary only in order to define and secure property rights.

[11] Of course, somebody can also feel the environment to reduce his utility, and thus considers the environment to be a negative externality, i.e. a 'public bad', for himself.

The public-goods approach requires assessment of environmental quality in order to attain the government goal of welfare maximisation. Environmental quality can be assessed using different methods, like the travel cost method, hedonic pricing, or contingent valuation method. The approach sees the social welfare function as reflecting the benefits and costs of environmental quality. The social welfare function is thus a guideline for benefit-cost analysis.

The *property-rights approach* can be interpreted as a contribution to the theory of institutions, where an institution is defined as a set of rules that specify how things are done in a society (Siebert 1995). If institutions cannot define the property rights for the use of the environment in agricultural practices, and the cost of using fertilisers does not reflect its social cost, farmers have an incentive to overuse fertilisers relative to the community's net interest (OECD 1994). Thus, if property rights are well established, the market will find the correct allocation of resources e.g. by negotiating over the compensation the externality creates to the property right holder.

3.3. Internalising the externalities

What are the solutions to the externality problems? For the economists pollution is an external cost, which occurs only when one or more individuals suffer a loss of welfare. The neoclassical solution is to find (or at least move towards) a *Pareto-optimum* of society, i.e. a situation where there is no alternative allocation of resources that leaves everyone at least as well off and makes some people strictly better off (Varian 1990). Thus, a Pareto-inefficient situation occurs when there exists some way to make somebody better off without hurting anyone else. In theory, the Pareto-optimum is reached, at the competitive markets, when the marginal social costs are set equal to marginal private benefits. This will yield the optimal level of pollution.

We have two approaches to find the optimal solution to externality problem: taxes and standards, and Coase solution. These also reflect the public-goods approach and property-rights approach.

Taxes and standards

In the idealistic economic environment of prefect information, choosing between a tax and a quota system (or market permit) is indifferent from the polluter's viewpoint. This can be seen in Figure 11, which depicts the situation in a simple way. The horizontal axis describes the level of reduction in the emissions, and the vertical axis the costs and benefits of an activity. The origin represents the zero level of abatement (no emissions reduction). The upward sloping curve represents marginal social costs (MSC) of pollution. The positive slope of the curve is logical, as the costs of pollution reduction are likely to increase when we approach the level of zero emissions. The downward sloping curve of marginal social benefits (MSB) is also presented in the figure. The negative sign of the MSB curve is also obvious, because the further we reduce the emissions the smaller is the marginal benefit of an incremental emission decrease.

The optimal point of emission reduction is at E, where the marginal benefits and costs are equal. This point can be reached in two ways; first, by setting a tax T upon each unit of emissions, or second, by setting a quantity level to emissions at Q. Both approaches result to the same optimal outcome (Baumol and Oates 1988). This analysis relies on perfect information on the cost and benefit curves. However, in reality the situation is usually not that simple.

Simultaneous uncertainty concerning marginal costs and marginal benefits makes the policy choice more complex. In the presence of uncertainty, the expected value of social welfare can differ significantly between the price vs. quantities control systems. When the regulator is unaware of the true position of the cost and/or benefit curves, the policy will, in general, differ from optimal (Baumol and Oates 1988).

However, benefit uncertainty on its own does not have an effect on the optimal control instrument (Weitzman 1974). Thus, it is important to know that uncertainty on the location of the MSB curve only does not affect the choice of the instrument, as equilibrium prices and costs depend exclusively on the cost function. If marginal benefits are fairly constant over the relevant range of waste emissions, the tax on emissions will provide close to the right signal as a measure of external costs (Baumol and Oates 1988). A more formal account of this is presented in Annex 2.

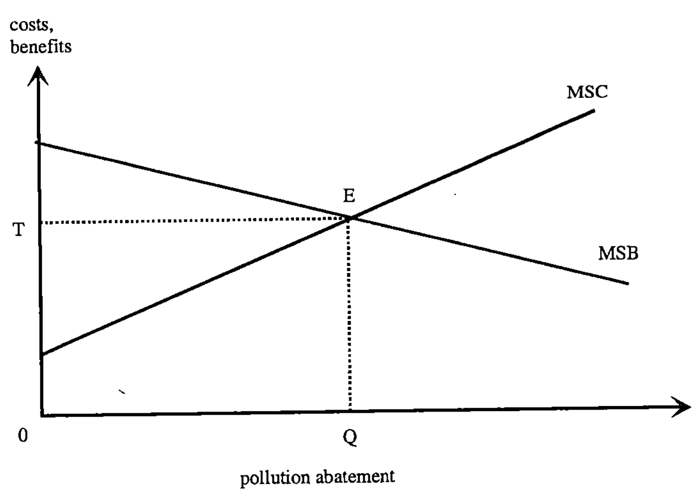

Figure 11. Marginal costs and benefits in pollution abatement.

Coase solution

Coase (1960, ref. Siebert 1995) has presented an alternative solution to optimal environmental allocation;

> "Let exclusive property titles to the environment be defined, and let them be transferable. Let there be no transaction costs. Let individuals maximise their utilities, and let them be nonaltruistic. Then a bargaining solution among different users of the environment will result in a Pareto-optimal solution among different users of the environment. The resulting allocation is independent of the initial distribution of property titles."

Coase pointed out that pollution control situations have a certain symmetry. Inefficient pollution imposes costs on victims which exceed the costs of controlling the pollution in question. In other words, the marginal benefits of pollution control exceed the marginal costs. The existence of inefficient pollution damage therefore provides a motivation for the victims to take corrective action even in the absence of any such incentives by the polluters (Tietenberg 1997).

Thus, if property rights are well established, regardless of who holds the property rights, there is a tendency to approach the social optimum (Pearce and Turner 1990). Therefore there is no need for government intervention, for the market will take care of the internalising of the externality. However, many problems undermine this theoretical approach, too.

For one, transaction costs, such as bringing involved parties together and the actual bargaining, are not likely to be zero. Further, negotiation is difficult to apply when the number of people affected by the externality is large (i.e. the free riding problem may arise).

Coase theorem assumes perfect competition where marginal net private benefits equal price minus marginal costs (i.e. $MNPB = P - MC$). Under imperfect competition, however, the optimal level of externality is achieved when marginal net private benefits equal the marginal revenue minus marginal costs (i.e. $MNPB = MR - MC$). Clearly, marginal revenue does not in this case equal the price because the demand curve is above the marginal revenue curve. Therefore, bargaining solution does not apply under imperfect competition (Pearce and Turner 1990). These may be some of the reasons why government intervention is likely to happen in policies aiming to internalise the environmental externalities.

In practise, environmental control policies are implemented through different policy instruments which will be discussed in the next chapter. However, before scrutinising the policy instruments we need to look at the nonpoint pollution characteristics of agricultural pollution. These characteristics have to be understood before applying policy instruments to the agricultural production process.

4. Control of agricultural pollution

4.1. Characteristics of nonpoint pollution

Nonpoint pollution (NPP) (or dispersed pollution) is defined as sources of water pollution not associated with a distinct discharge source. These include rainwater, erosion, runoff from roads, farms, and parking lots, and seepage from soil-based wastewater disposal systems. In comparison, point pollution (PP) is defined as a specific discharge that is traceable to a distinct source (i.e. pipe, ditch, container, or well.) such as discharge from wastewater treatment plants or industrial facilities (Cox et al. 1996). The presence of NPP can be indicated by measures concerning the water quality.

The nutrient runoffs from arable land are nonpoint pollution since the exact origin of these discharges is difficult (or even impossible) to define. Instead, runoffs from e.g. a farm's manure storage facility are considered point pollution as these originate from a specific source. Therefore, agricultural production bears the characteristics of both the nonpoint and the point source pollution. However, it is primarily the NPP through arable farming that is of interest here.

The inherent problem with respect to agricultural nonpoint pollution is that the ambient pollutant levels from farming depend on a number of climatic and topographic conditions in a manner that cannot be predicted with certainty. This implies that there will be a range of possible ambient levels associated with any given abatement practice or discharge level at any given time (Segerson 1988).

The range of possible ambient levels can be presented by a probability density function (PDF) that is conditional to the abatement practice. The usual objective of pollution control policies is to increase the probability that the ambient levels will fall below a certain tolerance level. This is presented in Figure 12, where the implementation of a certain policy measure causes the PDF_0 to shift to the left to the PDF_1, and to a reduced level of ambient pollution.

It must also be noted that, if the gap between the two PDFs is fairly narrow, i.e. the shift in the PDF_0 is only marginal, it becomes difficult to conclude if the abatement policy has really had any effect on the pollution level. Thus, uncertainty concerning the prevailing stochastic effects makes the policy assessment much more complex.

In the context of agricultural nonpoint pollution, there are some information problems related to imperfect monitoring. According to Braden and Segerson (1993), these are: 1) the inability to observe emissions, 2) the inability to infer emissions from observable inputs, and 3) the inability to infer emissions from ambient environmental quality. The combination of these problems can hinder the design of efficient pollution control instrument, but this also makes the policy design very challenging.

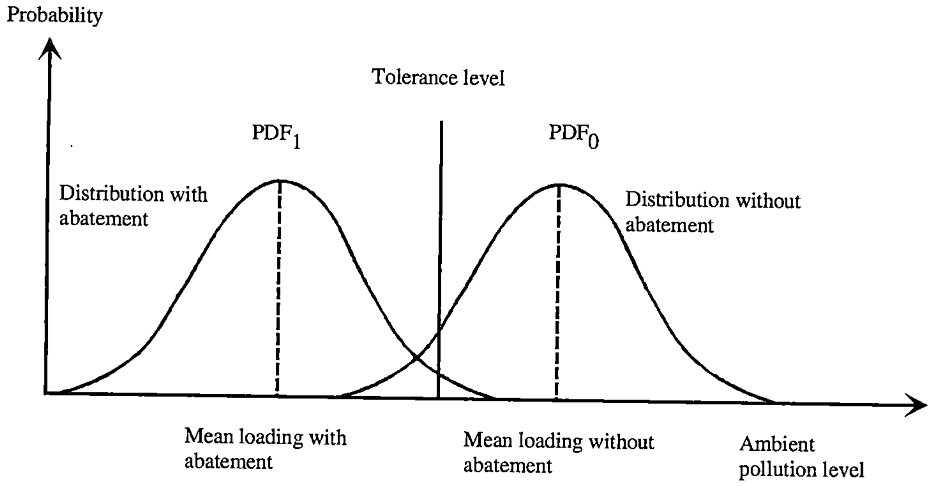

Figure 12. Probability density functions in pollution abatement. Source: Segerson (1988).

Shortle and Dunn (1991) give a useful presentation of the impact the weather has on the yield and consequently on the optimal fertilisation levels (Figure 13). They use a simple farm model where grain is produced, and the only factors having an influence on the crop yield are the weather and nitrogen application. The curve R_g represents the revenue from grain sales and nitrogen application if the weather conditions are good. Curve R_b represents the situation in a bad year, and R_e represents the situation in a normal year. C_f is the cost curve of fertilising. The dotted vertical lines indicate the points where profit is maximised at a corresponding revenue expectation and fertiliser application, i.e. the distance between the revenue curve and cost line is at maximum.

As the farmer is unable to foresee the weather in advance, he fertilises on the basis of what he expects to occur. If good and bad years are assumed to be equally likely, the expected revenue line R_e is the one according to which the farmer acts. Thus, the farmer uses the amount of N_e of nitrogen to maximise the profit, which is the distance between the curves R_e and C_f.

However, if the weather turns out to be bad instead of the normal one, the farmer over-fertilises by the difference between N_e- N_b. Furthermore, instead of getting the revenue of GB he gets EB, which is smaller than the optimal level DA in a bad year. Therefore, both the farmer and the environment lose in this setting.

If the weather turns out to be better than expected, the farmer would have been better off using more fertilisers, i.e. up to N_g, since the profit JC is bigger than IB. However, the farmer will get an extra revenue of IG. The environmental effects are positive because the good weather creates high yields and effi-

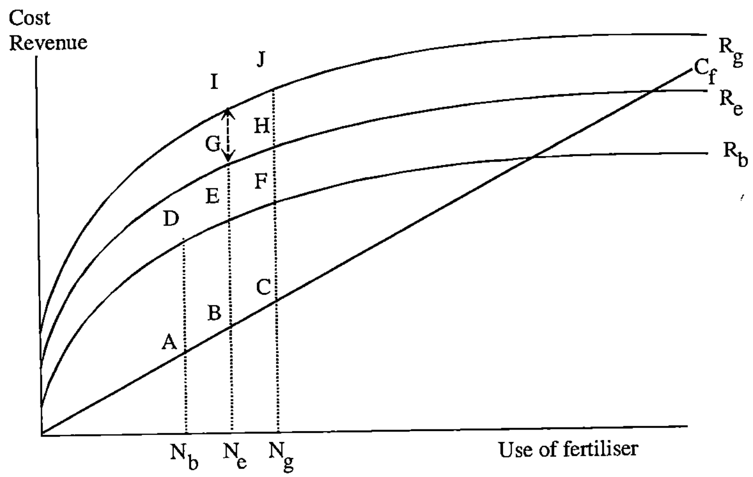

Figure 13. Profit maximising and optimal fertilisation levels. Source: Shortle and Dunn (1991).

cient nutrient uptake by the crop. Thus, the stochastic nature of the weather has an effect on the farmer's revenue, the crop response, as well as the nutrient emissions to the environment. Therefore the creation of policy instruments to tackle these phenomena proves to be very challenging.

4.2. Different ways to tackle the nonpoint pollution problem

It is clear that the characteristics of nonpoint pollution challenge the policymaker much more than in the case of point pollution. *Land use control* as well as technology-based *best management practices* (BMPs) are the two most widely used tools for controlling NPP and protecting designated uses of watercourses. The BMPs are defined as a structural or non-structural method, activity, maintenance procedure, or other management practice used singularly or in combination to reduce nonpoint source inputs to receiving waters in order to achieve water quality protection goals (Cox et al. 1996). Examples include animal waste management systems, conservation tillage systems, vegetated filter strips, manure and soil testing, following recommended manure and effluent application rates, and halting application of animal waste nutrients in winter months. Many of these measures are included in the present Finnish agri-environmental programme.

However, as e.g. Segerson (1988) points out, the BMPs may not allow for flexibility and cost-minimising abatement strategies unless applied on a site-specific basis. This is true also in the Finnish case. For example, the winter plant-cover criterion in Southern Finland is 30% regardless of the proximity to the watercourses. Therefore, farms located further away from rivers or lakes are carrying the same burden set by the criterion in spite of their lower potential for nutrient leakage. Braden et al. (1989) also point out that spatially uniform policies lack the cost-efficiency characteristics. This means that more emphasis should be laid on constructing locally differentiated or flexible policies instead of uniform policies. The drawback of this may be that controlling and monitoring costs become very high.

Instead of the BMPs, different pollution control policies have been introduced to tackle the NPP problem. The theoretical literature on controlling nonpoint pollution focuses on achieving optimal solutions (Helfand and House 1995). The pioneering research on agricultural NPP focused on examining the efficiency of different policy instruments. Griffin and Bromley (1982) evaluated the effect of effluent instruments and input instruments. They found out that properly designed *standards or incentive methods* perform equally well under complete information.

Shortle and Dunn (1986) examined the relative efficiency of different control strategies for achieving agricultural nonpoint pollution abatement under incomplete information. They concluded that appropriately specified *management practice incentives* (MPIs) outperform incentives or standards based on runoff estimates. The MPIs have an intrinsic potential to induce farmers to allocate their production methods to maximise the expected social net benefits of their decisions. Shortle and Dunn also emphasise the information quality conveyed by the MPIs, because the MPIs permit the farmer to utilise the informational advantage about the alternative management practices to maximise his welfare.

Furthermore, Shortle and Dunn suggest that when actual emissions cannot be measured, pollution should be monitored by controlling the use of inputs (i.e. the second-best solution). However, even though this method might reduce water pollution, it fails to incorporate potentially important and cost-effective abatement measures, such as improved management practices (Byström and Bromley 1996).

Braden and Segerson (1993) present a model showing that in the presence of information problems (e.g. the measuring problems of nonpoint pollution), the use of *multiple indirect instruments* may promote efficiency. The use of multiple instruments may be redundant in the world of first-best solutions of single instruments, but it may have an important role to play in improving efficiency when single instruments are imperfect.

In addition, various other *incentive schemes*, such as suggested by Segerson (1988) or Xepapadeas (1992), are proposed for controlling the NPP. In these models it is the *ambient concentration of the pollutants* according to which the proper policy measures are set. The inconvenience of these is that each polluter pays a tax which is independent of his own contribution to ambient pollution.

To overcome this drawback, Romstad (1997) has presented an incentive scheme where e.g. taxes are linked to the ambient quality of watercourses and the solution relies on a local team response. In the solution by Romstad, each polluter (agent) has superior information relative to the policymaker (principal) regarding his own as well as other agents' emissions. Then, properly designed incentives induce the agents as a team to make a contract with the principal to approach the policy goals set by the principal.

Another way to overcome the problem of ambient taxes is proposed by Xepapadeas (1997), who presents an *incentive scheme* where farmer is given a menu of tax schemes from which he can choose. The payments are not related to the deviations between desired and observed ambient pollution but instead relate to *observed choices of farmers*. This approach also relies on the asymmetry of information to reach the optimal emission levels. Xepapadeas shows that when farmers differ with respect to their emission characteristics, ambient taxes can lead to a sub-optimal emission level compared with the socially optimal regulatory scheme.

Thus, it seems that new and functioning (second-best) policy solutions to overcome the NPP problem of agriculture are emerging. Many of these solutions involve a special attention to the quality of information. However, a way to totally circumvent the problem of agricultural nonpoint pollution exists. We need to approach the problem from a different perspective; and the proposed way to do is to use the *nutrient balance calculations* in assessing a farm's nutrient loads. As described in Section 2.2. the environmental load of an individual farm (or a larger agricultural system) can be assessed by means of a balance calculation of nutrients. Thus, by measuring the incoming and outgoing nutrients (e.g. N, P and K), and deriving the balance, the problem of nonpoint pollution is truncated into a point pollution problem, which facilitates the first-best policy instrument assessment.

The NBCs yield farm-specific environmental discharges of different nutrients, and avoid the difficulties that obstruct the control of NPP. Of course, the nutrient balance calculations are not exempt from difficulties (as discussed in earlier sections), and they are by no means 'the only right way' to address the nonpoint pollution problems, yet they offer new possibilities to tackle the issue.

4.3. Environmental control policies

4.3.1. Overview of control policies

Government (i.e. the principal) uses different environmental policy instruments to reach the environmental objectives. These instruments can be categorised in different ways. The traditional twofold way is to talk about the economic instruments and the direct regulations (sometimes referred to as command-and-control instruments) (Russell and Powell 1996). Segerson (1990) defines the environmental control policies as incentive policies (economic) and regulatory policies.

In this study the classic dual classification is extended by one more element, and the three-category definition recommended by the OECD (1994) is used:

1. Economic instruments, sometimes referred to as market-based instruments
2. Direct regulations, also known as legislative approach; and
3. Information policy

The role of the information policy is highlighted in the approach of the OECD. Tietenberg (1997) calls this the third wave in the pollution control policy. In the following sections these three categories are reviewed in more detail. However, this study does not scrutinise the two traditional categories of environmental policy instruments very thoroughly, as the reader will find ample literature on these elsewhere.

4.3.2. Economic instruments

According to Segerson (1990), economic instruments to control nonpoint pollution are policy tools which create financial ex ante (i.e. prohibitive) incentives for producers to contain effluent leakage. In a farming situation this means that a farmer adapts his production according to the financial incentive he encounters e.g. in the prices of the inputs.

Economic instruments can be broadly defined as the policy measures that aim to achieve the environmental objectives through market-based measures. Market-based policies pursue to equalise the level of marginal costs of control among firms, rather than the level of control. These measures include: 1) charges or taxes, 2) tradable input quotas and emission permits, 3) direct payments, 4) environmental management agreements, 5) tax concessions, and 6) reassessing property rights (OECD 1994).

The first item mentioned in the list above, *charges or taxes*, is perhaps the most advocated one by economists but the least desired by the property rights holders. Charges and taxes on emissions are in line with the Polluter Pays Principle (PPP), but they are difficult to set to the optimal level especially in the case of agricultural pollution (as will be discussed later in Section 4.5.).

There are several reasons favouring the use of environmental taxes. According to EEA (1996), the first main economic reason for using taxes in the environmental policy is to bring the costs of pollution and other costs of using the environment into the prices of the goods and services produced by the economic activity. Further, an environmental tax provides an incentive to avoid the tax by using (or generating less of) the substance being taxed. This notion leads to the 'Porter hypothesis' which postulates that environmental regulation generates positive effects to firms' profits by inducing the firms to improve both their technical and environmental efficiency (see e.g. Hetemäki 1995).

An environmental tax allows each polluter to decide whether it is cheaper to pay the tax or to reduce pollution. Those polluters who face the highest costs of pollution reduction will tend to pay more of the tax, while those facing low reduction costs will reduce pollution, instead. The costs of achieving any given level of overall pollution reduction through a tax will therefore be cheaper than through a regulation. Furthermore, taxes work as innovations mechanisms in encouraging to operationalise new techniques to pollution control. And lastly, taxes raise revenues of the government. Therefore, policymakers are eager to use this option.

In both the *emission permit* and the *tradable input quota* systems a market for the right to use certain inputs or the emission levels is created. In the former case, the regulating authority allows only a certain level of pollutant emissions, and issues tradable permits for this amount (Pearce and Turner 1990). Tradability of pollution rights stems in essence from the Coase theorem, according to which the polluter and the victim negotiate over the compensation for the adverse effect the externality has on the other agent's welfare. Emission permits are already widely used e.g. in California in controlling sulphur emissions of industries. However, in the agricultural sector the adaptation of the emission permits has so far been very limited.

In an input quota system there will also be an intrinsic incentive to limit the input use as farmers could sell unused permits to other farmers. However, the administrative costs, especially in case of the emission permits may become very high as agricultural emission levels are not easily measured.

A policymaker can use *direct payments* in expressing its preferences concerning the desired environmental goods it considers worthwhile. An evident shift towards this direction is taking place in the Finnish society. Finnish agriculture is seen not only as a producer of food fibre, but also as a producer of environmental goods, such as the agricultural countryside environment, ACE (Aakkula 1996). Because of this multifunctional character, agriculture plays a particularly important role in the economic life of rural areas. The Finnish agri-environmental programme, according to the EU regulation 2078/92, can be seen as a way to produce environmental benefits.

The FAEP also bears some characteristics of *environmental management agreements*. In order to be eligible for FAEP support the farmer is obliged to fulfil several criteria, such as meeting certain fertilising base levels and establishing buffer strips.

Tax concessions can be used to encourage farmers to adopt particular farming practices to favour sustainable agriculture. This policy is similar to direct payments, but does not have such a heavy administrative burden. Typically, tax concessions are used to encourage investments in environmental purposes.

In many ways the problems related to pollution stem from the poorly or inadequately defined property rights. Therefore, *reassigning property rights* is one of the most central ways to tackle the problem. In the literature on environmental economics the failure to define a 'good' or a 'non-attenuated' set of property rights is seen as the reason for the existence of externalities and a market failure in this respect (Randall 1987) (see Section 3.2.).

The control measures based on economic instruments or regulations are often blind in taking into account the farm characteristics, i.e. they lack the site- and time-specific qualities of good environmental control. For example, a tax on fertilisers treats all farmers in the same way, even though the efficiency in the use of the input varies across the farms. In a situation involving two similar farms using the same amount of fertilisers on same crops but with different output levels, the farmer producing more efficiently pays the same amount of a tax as the other (more polluting) farmer. In addition, the least polluting crops are penalised more heavily than the crops from which the leaching is proportionally greater.

4.3.3. Direct regulations

Instead of market-based policy measures, legislative policies can be used in controlling agricultural pollution. It is the standards or regulations which have become the usual policies to control agricultural environmental pollution. For example, the nitrate directive of the EU (91/676/EEC) sets upper limits to nitrogen for fertilisation in order to secure proper drinking water quality. Regulations concerning certain manure storage capacity or guidelines for the right manure spreading are often used in this context.

Shortle and Dunn (1991) divide standards directed to reducing agricultural NPP into *performance standards* and *design standards*. Design standards are *ex ante* measures directed to the ways farmers produce agricultural commodities and manage their land. These can be considered the command-and-control policy instruments. Performance standards, on the other hand, are measures which use *ex post* information, e.g. the levels of chemical residues in water, to assess the environmental performance of the farm production.

Regulations can take many different forms, such as environmental norms for effluent emissions for farms or standards for food safety, total ban on certain inputs (e.g. dangerous pesticides) or practices, land use planning controls, and in certain cases the public ownership of resources (OECD 1994). Regulation minimises the risks and uncertainty of an outcome and it can often be targeted more precisely than other measures. In the case of e.g. safeguarding a habitat of an endangered species it might be necessary to ban certain farming practices to ensure the desired outcome.

Direct regulations are often coupled with sanctions in case of violating the regulation. This leaves the polluter the option to continue polluting up to the limit where his marginal net benefits equal the expected fine charged for violations.

Although economic instruments will often be the preferred choice by governments because of the extra costs involved in enforcing and monitoring direct regulations, direct regulations may be required to reinforce the effectiveness of economic instruments.

In Finnish agriculture, direct regulations are closely knit into fabric of the Finnish agri-environmental programme. In order to be eligible for the FAEP a farmer has to fulfil several environmental criteria, e.g. the certain base level fertilising must be followed, stocking density must be below 1.5 lu/ha and buffer strips must be left on the sides of main ditches (MMM 1994).

4.3.4. Information policy

In many cases both an accented as well as exercised way to address the environmental problems of agriculture is to apply information policies, sometimes also referred as moral suasion. Tamminen (1997) states that the main means to control agricultural pollution in Finland have, in fact, been the information policies. The concept 'information policy' is a somewhat nondescript expression, and needs to be defined more precisely.

The Finnish Environment Agency (SYKE 1995) defines the information policy as the set of public laws, regulations, and policies that encourage (or discourage) or regulate the creation, use, storage, and communication of information. Different interest groups and governmental or non-governmental institutions are the main actors in executing and developing information policy measures.

The OECD (1994) gives a brief explanation on the concept of information policy as a measure aiming at providing research and the establishment of technical indicators, a well as communication of information to farmers (extension, advisory and training) and to the society as a whole (education). All these will induce a voluntary change in the behaviour of the agent's production methods and techniques.

In agri-environmental information policy the goal is to reduce pollution by producing and sharing relevant information on the impact the agriculture has on the environment, as well as on the ways to reduce this impact. Most often the information is produced by research workers; and their results are delivered to farmers by agricultural advisory services or through the administrative bodies. The objective is to influence the farmers' attitudes and values through these different channels.

Information policy is in practise used as a supplementary measure; i.e. it is used in connection with other policy measures. Of course, when a policymaker sets, for example, direct regulations it also has a certain information dimension in its message. However, information policy is starting to gain more importance as a policy measure in itself (Juntti 1995, Tamminen 1997). Citing Tietenberg (1997), this is called the third phase of pollution control policies (the other two being economic instruments and direct regulations). According to Shortle and Dunn (1991), economists have, so far, been rather dubious about the effectiveness of voluntary control, except in certain individual cases. Yet, there seems to be new 'faith' emerging for the relevance concerning information policies in pollution control.

Usually, the starting point for this policy measure is the personal perception and realisation on the environmental matters and, through this, influencing the behaviour of an individual (SYKE 1995). In the Finnish agricultural sector it has mainly been the rural advisory centres which have carried out the information sharing tasks, thus being a workhorse for implementing governmental information policies.

During the 1990s information policies in connection with the agri-environmental issues have started to flourish. In 1991 the Centre for Rural Advisory Centres (in Finnish, MKL) made the environmental issues the main theme for the year through the 'Our common environment' campaign. In 1993 the Ministry of Agriculture and Forestry prepared the 'Code of good agricultural practice', which describes environmentally sustainable production methods for agriculture. The Code refers to such production methods and practices in which the coupling of benefits related to the environmental and production economics is sought for in line with the principles of sustainable development (MMM 1993).

A control measure based solely on information policy does not work very well, if tangible or evident entrepreneurial or environmental benefit is missing for the farmer to be realised. Hence, if the farmer does cannot observe any environmental effects to take place as a result of the change in his behaviour, or in the farm practice, the information does not mature into knowledge and, further, into environmentally friendly behaviour. Therefore, a feedback system to link the benefits to the actions is needed.

4.4. The role of information

Economists often talk about equilibrium on competitive markets at the point where the demand and supply meet. Competitive markets exists in the idealistic world of complete (or full) information and other simplistic assumptions such as free access to markets and multiple buyers and sellers on the markets. In reality these presumptions seldom hold very well. For example, information is seldom complete, which leads to inefficiency on markets, and this, in turn, often manifests itself through externalities. According to Rasmusen (1994), information problems have come to dominate research in both microeconomics and macroeconomics during the past ten years. Therefore, information is in a key role when defining workable policies for agricultural pollution abatement. Let us first inspect some definitions relating to information.

It is very costly (or even impossible) to acquire complete information on the price or the quality of goods sold on markets. Thus, information remains in most all cases incomplete[12]. (And yet we assume it to be perfect in our models!)

Information is said to be symmetric if all actors on the market have the same knowledge e.g. on the quality and/or prices of goods (or bads in case of a negative externality such as pollution) sold on markets. Information is asymmetric if some parties have knowledge relating to goods that the others do not possess (Rasmusen 1994 and Varian 1990); usually it is about the quality issues. In agri-environmental context: if farmers know the pollution abatement costs better (if not fully) than the policymaker, information is asymmetric. This is the typical principal-agent model in the game theory.

According to Byström and Bromley (1996), this asymmetry of information motivates the principal-agent approach. In order to maximise welfare the government (principal) needs to set up an incentive system that makes the farmer (agent) improve the water quality. However, there is also some common knowledge. The government can observe the outcome of farmers' factor use. Furthermore, the output and water quality are observable *ex post.* But despite this information, the government cannot observe the factor use directly, and hence will have less information than the farmers.

Two other concepts predominating the information theory are moral hazard and adverse selection. Moral hazard refers to situations where one side of the market cannot observe the actions of the other. Xepapadeas (1997) sees the moral hazard to arise when the principal is unable to observe the nutrient emissions of each agent. Adverse selection (or hidden information) is related to a situation where one side of the market cannot observe the type or quality of the goods on the other side of the market (Varian 1990). In a farming situation, this stems from a situation where the principal is unaware of the characteristics or type of each farm, which is private information known only to the farmer. Consequently, in lacking this information, the principal is unable to design policy measures which take into account the site-specific farming conditions.

In agri-environmental policy formulation the government possesses environmental information on national or regional nutrient problems. This information is aggregated, and it is collected either through research experiments or from

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12 A typical real life situation is that when I finally buy a new pair of jogging shoes, the following day I discover a yet cheaper deal for the same shoes in another shop.

measuring the ambient concentration of nutrients e.g. in the air or water. However, the government lacks the farm level (site-specific) information on nutrient emissions as the farmer does not know it either (i.e. in the traditional sense of NPP problem).

If farmer knew his emissions, the situation would turn out to be a game of asymmetric information. In a situation where the government, in tackling with the goal of improving water quality, sets e.g. a tax on fertiliser inputs, it lacks the knowledge of how well marginal benefits and marginal costs coincide. Then, the polluters (i.e. the producers of emissions) can react to the tax by applying the best suitable techniques (also cost-effective) to the farm production process. Berentsen and Giesen (1997) conclude in their analysis that a substantial decrease of nutrient losses can be realised by research, education, and extension, i.e. through elevated information level, while the income may actually increase.[13]

The third phase in the evolution of a pollution control policy, according to Tietenberg's (1997) terminology, involves investment in the provision of information. This augmenting role of information strategies, he argues, does not arise only from the increasing need for more regulatory tools, but also from the falling cost of information collection, aggregation, and dissemination. Lower information costs for producers and administration also mean lower overall social transaction costs for nutrient abatement (Huang and LeBlanc 1994). It should also be noted that 'pure' information policies do not alter the property rights and, therefore, policy solutions bearing information design encounter less resistance than conventional command-and-control policies.

New emphasis is thus arising concerning the importance of information in the policy formulation. A policy which respects and utilises the information the farmer possesses is an alternative way to tackle the pollution problem. This sort of non-regulatory approach allows farmers to decide the best management practices suitable for their conditions to improve the nutrient use efficiency. One promising new concept in this arena is the method of the nutrient balance calculations.

4.5. The choice of a policy instrument

Choosing between different policy instruments is a difficult task. According to Conway (1991), we have three main criteria which guide the selection of an environmental policy instrument. The first is the environmental effectiveness, the second is the administrative practicability, and the third is the cost-effectiveness of gains in environmental quality. Lankoski (1996) presents a framework

[13] This may also implicate an evidence of the Porter hyphotesis described in Section 4.3.2.

for the assessment of the optimality of the instrument, in which the key criteria are environmental effectiveness, economic efficiency, fairness and equity considerations, political acceptability, administrative practicability, and information content.

Braden and Segerson (1993) present three criteria which are important in evaluating polices in nonpoint pollution cases: ability to target, enforceability, and correlation with water quality (Table 4). They also argue that no single instrument appears to dominate in terms of all three criteria, and that trade-offs are unavoidable in the selection of policy instruments. The ability to target is high if the instrument efficiently correlates with the site- or time-specific environmental responses. Enforceability requires the principal to be able to set and oversee that compliance with the terms of the policy is met. High correlation with water quality is usually reached in case of instruments that are well linked with the variables closely correlated to the water quality.

Hanley (1990) points out that *optimality* (i.e. marginal net private benefits equalling marginal costs) itself is not a very good policy option. A policy aiming at e.g. reducing nitrates in groundwater should be targeted in such a way that given 'arbitrary' standards of water quality and/or nitrate input limits are efficiently achieved at the least cost. This is because nitrates take a long time to travel from topsoil to groundwater, and thus the methodological problems associated with predicting the marginal external cost to some future citizens are considerable .

In the case of large numbers of small polluters, in which case monitoring involves practical difficulties, a tax on polluting inputs may be a second-best policy (Hanley 1990). However, the efficiency of such a tax would depend on whether the quantity of an input purchased is closely correlated with the volume of pollution emissions generated. Such a close correlation is usually rather

Table 4. Evaluation of policy instruments. Source: Braden and Segerson 1993.

	Rating with respect to		
Tax or Subsidy	ability to target	enforce-ability	correlation with water quality
Output	L	H	L
Input	L/M	H/M	M
Emissions/Management practices	H	M	M
Ambient concentration	H	L/M	H
Use of liability	H	L	H

L means low, M means medium, and H means high rating.

unlikely, as Braden and Segerson (1993) point out. More importantly, the effectiveness of this instrument depends on the elasticity of the input demand response to the higher input prices. In the case of fertilisers, the demand elasticity is quite inelastic in Finland (Ryhänen 1994).

In general, economic incentives have many advantages over regulations (Karp et al. 1995). First, they can achieve the desired effect at the least possible cost. Second, they are easier to enforce. Third, they present fewer opportunities for rent-seeking behaviour; therefore, they are likely to be more effective and more equitable. Finally, unlike regulations, economic incentives generate revenues which may be used to finance the incentive programme.

Market-based incentive systems 'automatically' lead to the cost-effective allocation of the pollution-control burden among firms. By forcing firms to factor environmental costs into their decision-making, these systems create powerful incentives for firms to find cleaner production technologies, i.e. to internalise production externalities. Market-based incentives also make the environmental debate more understandable to the general public by focusing attention to what our environmental goals should be, rather than on difficult technical questions about different means for reaching these goals. (Stavins 1992).

5. Nutrient balances and weather

5.1. Data description

As already mentioned in Section 2.3., we have observed the importance of the climate to the nutrient balances in Finland. A question emerged on the feasibility of the nutrient balances when facing the problem of external factors, such as the weather and soil type, having an effect on the build-up of the surplus. Ideally the nutrient balance calculations would allow us to inspect the agricultural pollution from a point source perspective as the balance calculation would give us the information on the emissions of the farm. However, there are evident problems to make such simplistic assumptions. This chapter explores the effect of the weather on nutrient balances.

Because of the weather, yield levels vary considerably from year to year in Finland as Finland is located in a fringe area for the cultivation of many crops. Wheat and rye can be grown in Southern Finland but not in Northern Finland. In addition, the border for growing e.g. barley is met in Lapland. The question of how important a role the external factors (especially the weather) have in terms of the nutrient balance needs to be studied in detail. If external dictate the level of the nutrient balance to any greater extent, setting a levy on the surplus loses some of its justification. This problem bears connection to the general theme of the impact of the climate change to agricultural production.

The importance of external factors in the formation of the nutrient surpluses is often questioned in this context (e.g. Lankoski 1996). Mukula and Rantanen (1989) have studied the risks to the yield of arable crops in Finland. The estimated variation coefficients[14] were 15% for barley, 19% for winter wheat, 16% for spring wheat, 11% for barley, and 9% for oats.

The yield levels are closely linked to nutrient balances, especially to the nitrogen balance, where the correlation is over 90%. Therefore the variation coefficients also reflect the magnitude of the variation of the nutrient balances due to climatic factors. However, in order to get more updated figures an analysis of more recent data was conducted.

This question was studied based on the data from the Agricultural Research Centre (MTT). The data were originally used in comparing the crop production in organic (or the so-called self-sufficient) and conventional farming. However, only the data from conventional farming were used in this study.

The data cover the years 1983-1993. Four crops in four research stations were cultivated. The geographical location of these stations is presented in the map in Figure 14. The two southern research stations have fairly similar climatic conditions, and, correspondingly, so have the two northern stations.

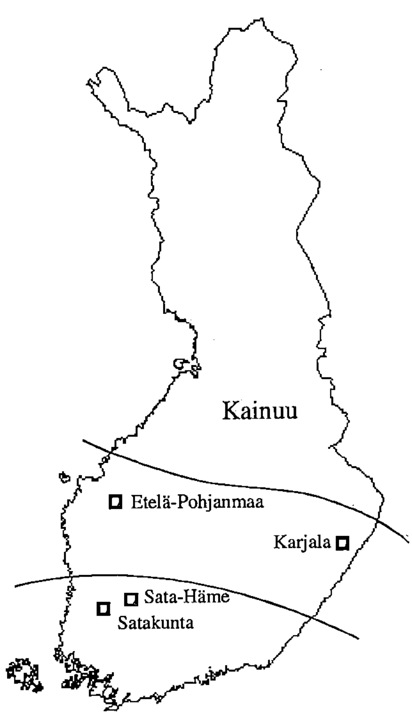

Figure 14. The four Finnish research stations.

The data consist of altogether 413 observations. Monthly data on the temperature and precipitation were available during this period from all four research stations. Crops were fertilised at normal fertilising levels and, no manure was used. Tables 5 and 6 depict some key elements of the data.

In Table 5 the nitrogen surplus in kg/ha has a very high standard deviation, especially in case of potatoes. This means that there appears considerable variance in the actual crop uptake of nitrogen. Both rye (in average, almost 91 kg/ha of nitrogen) and potatoes (88 kg/ha in phosphorous) have very high nutrient surpluses.

The variation coefficients for nutrient surpluses are rather high, particularly for

[14] The percentage of standard deviation of the average yield, i.e. vc = s/x_{avg}

Table 5. Mean, standard deviation, and variation coefficient for nutrient surpluses by research station and crop in 1983-1993.

Research station	Crop	obs.	Nitrogen			Phosphorous		
			Mean	StdDev	VarCo	Mean	StdDev	VarCo
Satakunta	Rye	18	84.7	12.3	15%	30.2	7.5	25%
	Barley	36	35.0	21.5	61%	27.3	3.7	14%
	Oats	36	18.4	16.5	90%	24.0	4.5	19%
	Potatoes	18	28.5	18.8	66%	89.7	7.6	8%
Sata-Häme	Rye	18	93.8	14.4	15%	31.2	6.2	20%
	Barley	33	54.2	15.7	29%	30.6	4.7	15%
	Oats	30	38.1	15.3	40%	28.3	4.9	17%
	Potatoes	17	11.2	30.9	276%	87.7	9.6	11%
Etelä-Pohjanmaa	Rye	18	84.4	11.2	13%	29.5	5.5	19%
	Barley	36	38.7	12.4	32%	26.8	4.7	18%
	Oats	36	9.7	16.7	172%	22.4	5.4	24%
	Potatoes	18	19.8	26.1	132%	88.6	7.0	8%
Karjala	Rye	18	99.9	18.5	19%	32.3	4.7	15%
	Barley	33	43.0	20.4	47%	28.1	5.4	19%
	Oats	30	27.3	26.3	96%	26.0	6.8	26%
	Potatoes	18	15.8	30.3	192%	86.2	8.6	10%
Average	Rye	72	90.7	15.5	17%	30.8	6.0	19%
	Barley	138	42.5	19.1	45%	28.1	4.8	17%
	Oats	132	22.5	21.5	96%	25.0	5.8	23%
	Potatoes	71	18.9	27.1	143%	88.0	8.2	9%

nitrogen. Even though potatoes are very good in utilising nitrogen, it is very fragile in terms of weather conditions as summer night frost may cause almost a total loss of crop. This took place in a few cases, and the yields were under 15,000 kg/ha, which caused a very high nutrient surplus. This is also true for oats, where night frost can cause a dramatic decrease in the yield levels (see also Kettunen et al. 1987).

The next question is to what extent the weather factors contribute to the variance in nutrient surpluses and, therefore, to the ability of the plants to use fertilisers. The weather data during the summer months at the research stations is shown in Table 6.

It is clear that weather conditions vary considerably from year to year (Table 6). For example, the year 1987 was very poor for crop growth because of the low effective temperature sum (ETS). It was some 20% lower than on average on these four research stations during the time period of 1983-1993. The summer 1991 was colder than the average. In 1988 the temperature sum in May-August was 16% higher than normal, and as much as 33% higher than in 1987. In comparing the data on corresponding nitrogen surpluses at national level as

Table 6. The effective temperature sum and rainfall (mm) in May-August in four Finnish research stations in 1983-1993.

Year	Satakunta temp.	Satakunta rain	Etelä-Pohjanmaa temp.	Etelä-Pohjanmaa rain	Sata-Häme temp.	Sata-Häme rain	Karjala temp.	Karjala rain
1983	1,067	206	1,002	182	1,079	223	1,089	199
1984	1,091	296	1,068	268	1,081	248	1,085	190
1985	1,026	216	974	203	1,011	245	960	313
1986	1,098	187	1,049	235	1,092	285	1,066	293
1987	807	320	780	189	786	304	844	428
1988	1,230	320	1,160	286	1,211	429	1,177	303
1989	1,091	230	1,052	259	1,093	227	1,122	186
1990	1,072	184	992	221	1,032	211	890	196
1991	993	254	960	288	984	242	981	395
1992	1,104	279	1,041	212	1,097	257	990	217
1993	1,015	320	932	308	1,001	305	904	386
aver.	1,054	256	1,001	241	1,042	271	1,010	282
std.dev	103	54	96	43	105	61	106	91

shown in Figure 7 on page 25, we can conclude that at aggregated level there is some evidence on the link between the temperature sum and nitrogen surplus. The same can be said for the part of the precipitation. The year 1987, in particular, shows a clear connection. Next, the situation was scrutinised at a more disaggregated level.

5.2. The effect of weather on the nutrient balance calculations

The data were analysed using PC-SPSS programme. First, correlation matrices for each four crops were calculated to see if there exists a linear correlation between the nutrient balances with respect to precipitation and ETS. Secondly, variance analyses were performed to distinguish the relevant weather factors having an effect on the formation of nutrient surpluses. The analyses were done with respect to nitrogen and phosphorous.

The highest, statistically significant at 5% level, correlations are shown in Table 7. The data suggest that proper *precipitation* is clearly significant for the build-up of the nutrient surpluses. Excessive rain in May and August, in particular, result in high nutrient surpluses. In the case of the other months the correlations are also mostly positive, i.e. the more it rains the greater are the nutrient surpluses. There also occur a few negative correlations. These indicate that when drought takes place, rain after the dry season improves the overall nutrient utilisation.

Table 7. Most important statistically significant correlations between nutrient surplus and weather factors in May-August.

Nutrient surplus			Correlation of nutrient surplus with respect to									
			Precipitation in May-August					Effective temperature sum in May-August				
Area	Crop		month					month				
			5	6	7	8	5-8	5	6	7	8	5-8
1	Rye	N										.56
		P	.55		.53				.67			.73
2	Rye	N									.53	
		P		-.47	.52	.73		.47	.59	.61		.77
3	Rye	N				.59	.72			-.63	-.72	-.49
		P								-.83		
4	Rye	N		.66	.74	.80	.81				-.54	
		P								-.64		
1	Barley	N	-.53			.36		-.60				
		P	.37							-.55		
2	Barley	N				.38					-.38	-.55
		P				-.45					-.45	
3	Barley	N										
		P		.36	.36			.50				.37
4	Barley	N	.61		.55	.42	.69	-.53				
		P	.64		.37		.50			.40		
1	Oats	N				.75	.66				-.37	
		P	.54						.34		-.44	
2	Oats	N									-.55	
		P	.47	-.40							-.54	
3	Oats	N		.66				.54	.48			.49
		P		.50			.40	.38	.59			.47
4	Oats	N	.76						.53	.42		.48
		P	.70						.67	.52		
1	Potatoes	N	.58	.85	-.56							
		P		.62								
2	Potatoes	N									-.62	
		P	.65								-.79	
3	Potatoes	N	.82		-.82			-.83				
		P	.77		-.76	-.51		-.56				
4	Potatoes	N	.54				.58				-.52	
		P	.67				.51				-.52	

at 5% risk level

Areas: 1 = Satakunta, 2 = Etelä-Pohjanmaa, 3 = Sata-Häme, 4 = Karjala

The *temperature*, especially in August, is significant for the build-up of nutrient surpluses, as the very consistent negative correlations for that month indicate. The higher the degree-days in August is the lower is the nutrient surplus, and the better the nutrients are utilised. The same holds true to some extent for the ETS in May. However, there exists positive correlations for oats and rye mainly in June and July, indicating that these crops do not benefit from high ETS.

In the table above potatoes seem to have very uniform correlations. The relationship between the nitrogen surplus and precipitation in May by research unit is plotted in Figure 15.

Figure 15 shows that there appears a clear tendency to get high nitrogen surpluses for potatoes if May is a wet month. The correlations for this relationship shown in Table 7 are significant at 5% risk level for all research units except that of Etelä-Pohjanmaa.

In general, only few correlations over 60% exist. Therefore, it can also be assumed that the correlation between e.g. precipitation and nutrient surplus is not linear. Too little rain e.g. in May or too much rain in May probably have a similar effect on nutrient surplus, i.e. extreme rainfall conditions result in high nutrient surpluses and poor nutrient utilisation. Thus this relationship is most

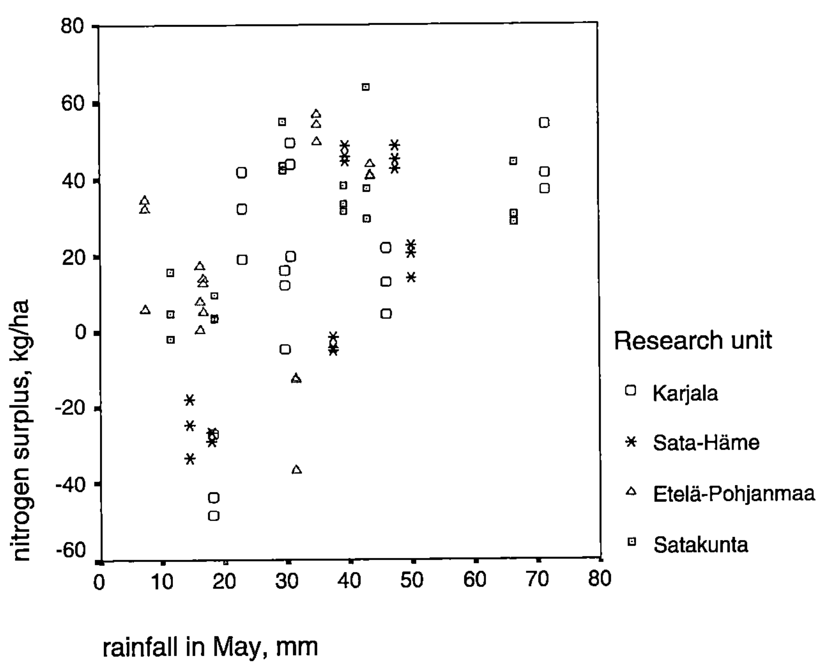

Figure 15. Scatterplot of nitrogen surplus for potatoes and precipitation in May by research area.

likely to be non-linear as has also been noted by Teigen and Thomas (1995), who postulate that the weather is the single most important factor influencing crop production. They present models showing weather-yield response to explain more than 90 percent of yield variation.

Kettunen et al. (1987) conducted linear regressions analyses to find out weather response to crop growth. They found out that temperature or precipitation summed on a monthly basis could not capture the bio-physical phases in the crop growth thus resulting in poor regression models. As the data used here was solely calculated on a monthly basis the more revealing models could not be used.

5.3. The magnitude of weather factors

In order to quantify the effects of weather related factors to the build-up of the nutrient balances, weather variables were categorised and analysed against the nutrient balances in the variance analyses. The total effective temperature sum from May to August was split into three groups in 1:2:1 proportions. This method created the groups of 'cold', 'normal' and 'warm' summer. The total rainfall for the same period was categorised accordingly. In the variance analyses the t-tests (both Tukey's t-test and Student-Newman-Keuls t-test) were conducted at 5% risk level to find out the differences between the groups.

The findings of the variance analyses with respect to the total data are shown in Table 8. There usually exists a statistically (at 5% level) significant difference between the mean of nutrient surpluses when comparing the cold summer

Table 8. The effect of temperature on nutrient surpluses.

Crop	Nutrient surplus	The difference* in the nutrient surpluses between the temperature groups, kg/ha (%) The type of summer			Average surplus
		cold vs. normal	cold vs. warm	normal vs. warm	(kg/ha)
Rye	N	12.1 (12%)	15.8 (16%)		90.7
	P				30.8
Barley	N	10.5 (20%)	12.4 (24%)		42.5
	P				28.1
Oats	N		-13.6 (82%)	-13.3 (70%)	22.5
	P		-4.4 (19%)	-3.7 (15%)	25.0
Potatoes	N	31.9 (90%)		-21.4 (86%)	18.9
	P	6.5 (7%)			88.0

* statistically significant differences at 5% level.

to the normal summer, and between the cold summer and warm summer. For example, in a cold summer, nitrogen surplus for rye is 12.1 kg/ha higher (that is 12% higher) than in the case of a normal summer. In a cold summer the nitrogen surplus for potatoes is 31.9 kg/ha higher than in a normal summer. These mean that a cold summer usually leads to high nutrient losses.

According to Table 8, it seems evident that the higher effective temperature sum is beneficial for the efficient uptake (i.e. low nutrient surplus) of nutrients in the case of rye and barley. However, in the case of oats the situation is different, and the negative signs for the difference indicate that higher ETS yields higher nutrient losses in case of both nitrogen and phosphorous. The same is true for the nitrogen surplus of potatoes when comparing a normal summer to a warm summer (i.e., 21.4 kg/ha higher N-surplus in a warm summer).

Furthermore, a variance analysis of a similar type was conducted to see if the precipitation has an impact on nutrient surpluses (Table 9). The table indicates that precipitation is of lesser importance to the build-up of the nutrient surpluses. However, wet growing conditions increase the nitrogen surpluses in the case of rye and oats. Rye has, on average, 24.6 kg/ha higher nitrogen surpluses during a wet summer compared to a dry summer, and 23.1 kg/ha higher surplus when comparing a wet summer to a normal summer.

The relationship between the weather and the nutrient surplus was not explored more thoroughly, as it is not in the scope of an economic study, but in that of a biological one. From the analyses above it can be concluded that biological processes are very complex as well as very sensitive to precipitation and, especially, to temperature. The sensitivity of crop response to weather conditions leads to similar responses with respect to the nutrient surplus. However, this finding somewhat shadows the potential efficacy of using the NBCs as a policy instrument, and it should also curb the policymakers' eagerness to set policy measures, such as levies, on nitrogen surplus. The system is unjust if a

Table 9. The effect of precipitation on nutrient surpluses.

Crop	Nutrient surplus	The difference* in the nutrient surpluses between the precipitation groups, kg/ha (%) The type of summer			Average surplus
		wet vs. dry	wet vs. normal	normal vs. dry	(kg/ha)
Rye	N	24.6 (23%)	23.1 (21%)		90.7
	P				30.8
Oats	N	21.6 (69%)		13.5 (58%)	22.5
	P				25.0

* statistically significant differences at 5% level.

farmer who is cultivating according to his expectations of a normal summer and faces a bad year for crop growth is penalised by his higher than average nutrient surpluses. Furthermore, the ambient level of pollution can increase despite the farmer's own contribution to abate the pollution. At least these characteristics need to be taken into consideration in the policy formulation.

The analyses above were conducted on the basis of a data from research stations. Next, the situation is looked at the farm level, based on farm level data from Eastern Finland.

6. Nutrient balances on dairy farms in Kainuu

6.1. Data description

Weather factors are significant in terms of the build-up of nutrient balances. The previous section shed some light on the crop level magnitude of this. At farm level the situation is more complex than on research parcels, and even more complex in a situation where manure is applied to the soil.

The nutrient balance data from Kainuu[15] Rural Agricultural Centre were collected during 1995 and 1996 for the year 1994. All three balances (i.e. farm gate balance, surface balance, and cattle balance) were calculated for the farms. Altogether some 250 dairy farms are involved in a local dairy project, which has developed an environmental index where agricultural nutrient balance is one of the factors in the index (Hämäläinen and Kaataja 1996). Of these farms, 191 farms were included in the analysis. The data are summarised in Table 10.

These farms are characterised by fairly high milk yields and a low average animal density. Farm gate balance and surface balance show nitrogen surpluses fairly close to each other (theoretically they should be equal, see Section 2.2.1.), which speaks for the good accuracy in reporting the elements of the calculation. The nitrogen surpluses are rather high, i.e. in the range of 120-130 kg/ha. The utilisation percentage tells how well the nutrients are captured in the production process. The cattle balance surpluses in kg/LU are somewhat lower than the ones reported in Mälkiä (1997), i.e. 96 kg/LU vs. 114 kg/LU.

It is obvious that the utilisation of nutrients is better on the arable land than in the cattle house. Lankoski (1996) estimated nitrogen utilisation to be about 60-70% in the cereal farming and 30-40% in the livestock production (net surface balance). This explains the difference between the utilisation percentages between the farm gate balance and surface balance. The very low minimum nutrient surpluses and, consequently, high utilisation percentages are from organic farms.

[15] Kainuu area is marked on the map in Figure 14 on page 57.

Table 10. Description of the dairy farm data from Kainuu: Nitrogen balance.

	Mean	StdDev	Maximum	Minimum
Farm size, ha	22.9	9.3	64.3	7.3
Livestock units (LU)	18.2	5.9	36.5	6.1
Livestock units/ha	0.84	0.20	1.73	0.39
Milk yield/cow, kg	7,343	1,014	9,856	4,400
Nitrogen balance				
Farm gate surplus, kg/ha	128.4	37.9	214.7	20.0
Farm gate, utilis. %	19.4	5.4	51.6	8.7
Surface surplus, kg/ha	119.0	40.8	271.6	32.1
Surface, utilis. %	41.7	9.9	71.1	18.4
Cattle surplus, kg/ha	79.7	23.0	213.9	31.3
Cattle surplus, kg/LU	95.7	15.8	158.1	56.4
Cattle, utilis. %	26.7	3.6	35.3	17.5

The factors contributing to the build-up of nutrient balances were estimated from the data. Furthermore, the 21 farms (having a complete data) in the sample, which are also bookkeeping farms, were analysed to find out the relationship between farm economic indicators and nutrient balances.

6.2. Nitrogen surpluses

The reasons for high nutrient losses are manifold and depend on the circumstances of the farm, but some common nominators also exist. For example, in Germany Doluschitz et al. (1992) analysed the factors having an effect on the build-up of nitrogen surpluses (farm gate balance). They found out that, on average, the animal density is one of the most crucial factors in explaining the level of nitrogen surpluses. However, on individual farms the connection between the animal density and N-surplus was not that well verified.

Accordingly, analyses on the factors contributing to the build-up of nitrogen surpluses were conducted on the Finnish data. A regression model explaining the nitrogen surplus from the cattle balance by arable land area (HA) and the number of livestock units (LU) gave the following regression (see Annex 5);

$$\text{N-surplus}_{\text{cattle}} = \underset{(20.37)}{80.29} + \underset{(10.61)}{3.39}\text{LU} - \underset{(-13.60)}{2.71}\text{HA};\ R^2 = 51\%$$

All coefficients are significant at 1% risk level. The model states that adding one more livestock unit to the farm increases the nitrogen surplus by about 3.4 kg/ha. However, increasing the arable land area by one hectare decreases the

N-surplus by 2.7 kg/ha. This result reveals that reducing the livestock density by one unit is a more efficient way to reduce the nitrogen surplus than increasing the arable land area by one unit. Which method is the more cost effective one depends on the relative prices of production factors, as the following simple calculation shows.

If the farmer wants to decrease the farm's average nitrogen surplus by 10 kg/ha, he needs to increase his arable land area by 3.75 ha. If the price of land is 10,000 FIM/ha and the yearly gross margin income[16] from cultivating barley is 3,000 FIM/ha, this costs him about 4,500 FIM/year in a five-year period. Similarly, the same decrease in the nitrogen surplus is attained if the farmer reduces his livestock by 3.0 livestock units. If the farmer disposes of three cows, from which he gets FIM 1,000 each thus losing a gross margin income of 8,000 FIM/year, this costs him 7,400 FIM/year for five years. In this scenario it is cheaper to buy new land than to reduce livestock in order to reduce the nitrogen surplus

Table 11. The relationship between the farm size and nitrogen use efficiency by cattle balance.

Size of the farm		Nutrient use efficiency of nitrogen by cattle balance, % -24.3	24.3-27.1	27.1-29.4	29.4-	Total
	Count	13	12	15	8	48
below	Exp value	11.1	11.3	14.3	11.3	48
16.3 ha	Row Pct	27.1%	25.0%	31.3%	16.7%	100%
	Col Pct	29.5%	26.7%	26.3%	17.8%	25.1%
	Count	10	12	11	15	48
16.3-	Exp value	11.1	11.3	14.3	11.3	48
22.0 ha	Row Pct	20.8%	25.0%	22.9%	31.3%	100%
	Col Pct	22.7%	26.7%	19.3%	33.3%	25.1%
	Count	7	8	18	13	46
22.0-	Exp value	10.6	10.8	13.7	10.8	46
27.4 ha	Row Pct	15.2%	17.4%	39.1%	28.3%	100%
	Col Pct	15.9%	17.8%	31.6%	28.9%	24.1%
	Count	14	13	13	9	49
over	Exp value	11.3	11.5	14.6	11.5	49
27.4 ha	Row Pct	18.6%	26.5%	26.5%	18.4%	100%
	Col Pct	31.8%	28.9%	22.8%	20.0%	25.7%

Chi-square value is 8.804, its significance is 0.456 at 9 degrees of freedom.

[16] Gross margin income of a crop is obtained by subtracting total variable costs from total income. For simplicity and comparability issues the calculation does not take into account interest rate or work costs.

by 10 kg/ha. However, if the price of land is 15,000 FIM/ha, the cost to the farmer in the land buying option rises close to 8,300 FIM/year, and the situation is reversed.

One interesting question is the following: 'How do small farms compare to big farms in respect of the nutrient use efficiency?'. The farms were grouped into four size categories. Correspondingly, the nitrogen use efficiency by cattle balance (i.e. the quotient of the nitrogen fed to the cattle and the nitrogen in the milk and meat) was divided into four groups. The cross-tabulation of these categories is shown in Table 11.

The table shows that the group of the smallest farms (under 16.3 ha) contains only 16.7% of the farms in the best utilisation category. However, the group of the biggest farms also has a relatively low percentage (18.4%) in the highest efficiency category. The group with the farm size of 22.0 - 27.4 ha has clearly the highest nitrogen use efficiency. In this group, over 67% of the farms have better than the average nitrogen use efficiency. 47 farms of the 96 farms with under 22 ha of arable land have the nitrogen use efficiency of under 27.1%. In contrast, 42 farms of the 95 farms with over 22 hectares of arable land have the nitrogen use efficiency below 27.1%. Thus, there seems to be a tendency that the bigger farms have a higher nitrogen use efficiency by cattle balance.

6.3. Nutrient balances and farm economic indicators

Altogether 21 farms of the total of 191 farms in the data were bookkeeping farms in 1994. These farms keep very detailed records on the farm's monetary flows, but this data collection mostly lacks the data on quantities. (Necessary data for calculating the nutrient balances will be collected from bookkeeping farms starting from 1997.)

The data of nutrient balances were compared to some key economic variables, and analyses were conducted for quantifying the relationship between these factors. The data from these 21 farms are summarised in Table 12. These farms are fairly similar to the rest of the sample, on average. However, the nitrogen surpluses seem to be slightly higher than in the base data.

In general, the relationship between the farm economic indicators and the nutrient balance calculations in the data is relatively scarce. Nevertheless, some observations can be highlighted from the data. Pearson correlations[17] were conducted to assess the relationship between the nutrient use efficiency and farm economic indicators. The main findings are reported in Table 13. The column 'input costs' includes the costs for fodder and fertilising, costs for fuel

[17] Pearson correlation is a measure of linear association between two variables, i.e. $\Sigma((x_i-x_{avg})(y_i-y_{avg}))/sqr(\Sigma(x_i-x_{avg})^2\Sigma(y_i-y_{avg})^2)$

Table 12. Description of the bookkeeping dairy farm data from Kainuu: Nitrogen balance.

	Mean	StdDev	Minimum	Maximum
Farm size, ha	22.0	8.1	10.3	40.7
Livestock units (LU)	17.6	5.8	8.3	30.1
Livestock units/ha	0.82	0.17	0.41	1.23
Milk yield/cow, kg	7,265	988	5,000	9,054
Nitrogen balance, kg/ha				
Farm gate surplus, kg/ha	134.1	47.0	25.0	214.7
Farm gate, utilis. %	20.0	6.5	10.6	37.6
Surface surplus, kg/ha	128.5	54.4	32.0	271.6
Surface, utilis. %	41.3	10.3	22.1	62.6
Cattle surplus, kg/ha	79.9	21.9	32.6	132.5
Cattle surplus, kg/LU	96.6	15.9	75.2	133.4
Cattle, utilis. %	27.4	4.1	19.8	35.3

and energy, and costs for pesticides. 'Other costs' include the costs for maintaining ditches, insurance costs, and rent costs.

The results suggest a clear link between the cost structure and the nitrogen surpluses. The correlations are positive (e.g. between 0.559 and 0.645 in the case of total costs) meaning that high costs per hectare, and specifically high input costs per hectare (correlations being well over 0.7), result in high nitrogen surpluses and, consequently, high nutrient emissions to the environment. The statistical evidence is well pronounced. In other words, the lower production costs coincide with efficient nitrogen use. This finding corresponds to the

Table 13. Correlations between cost factors of farm production and nitrogen surpluses.

Nitrogen balance, kg/ha	Agricultural production cost, FIM/ha			
	animal purchase	input costs	other costs	total costs
Farm gate surplus, kg/ha		0.764**		0.559**
Farm gate, utilis. %				
Surface surplus, kg/ha		0.766*		0.644**
Surface, utilis. %		-0.559**		-0.442*
Cattle surplus, kg/ha	0.436*	0.724**		0.645**
Cattle, utilis. %				

* Correlation is significant at the 0.05 level (2-tailed).

** Correlation is significant at the 0.01 level (2-tailed).

Helander's (1996) results from pig production. When scrutinising the input factors more closely, the main component contributing to a high correlation was found to be the cost of fertilisers. The correlations were in the range of 0.64 to 0.69, depending on the nitrogen surplus category.

Although the utilisation percentage figures do not often have a significant correlation with the cost components, there is a negative relationship between these factors. The only case in which the correlation is significant at 5% level is the utilisation percentage for surface balance in relation to the input cost and the total costs.

Of course, the cost factors are just one side of the coin, and we also have to look at the income factors to gain a better picture of the matter. Again, Pearson correlations (for the nitrogen surplus per hectare vs. farm income per hectare) were calculated; and the results are shown in Table 14.

In general, the higher the agricultural income is the higher are also the nutrient surpluses. The negative correlation of utilisation percentage for surface balance with respect to total income indicates the same phenomena. Income from crop production has significant correlation only with respect to surplus of surface balance. All three balances give significant correlations with respect to farm income form animal husbandry, and they are in the range of 0.521 to 0.596. Hence, high total agricultural income is related to weak nutrient use efficiency, but this finding is less revealing than the cost inspection above.

The data from Kainuu dairy farms did not allow for calculating the production costs by output. Thus, we lack the proper data for a deeper analysis of the farm economic indicators with respect to the nutrient surpluses. Therefore, a modelling approach is developed to clarify this relationship, and to assess policy measures utilising the information from the NBCs.

Table 14. Correlations between income factors of farm production and nitrogen surpluses.

Nitrogen surplus	Agricultural income in FIM/ha from crop production	animal husbandry	total income
Farm gate surplus, kg/ha		0.521*	0.561**
Farm gate, utilis. %			
Surface surplus, kg/ha	0.584**	0.567**	0.641**
Surface, utilis. %			-0.478*
Cattle surplus, kg/ha		0.596**	0.593**
Cattle, utilis. %			

* Correlation is significant at the 0.05 level (2-tailed).
** Correlation is significant at the 0.01 level (2-tailed).

6.4. Environmental policy instruments at farm level

6.4.1. Modelling

Tietenberg (1992) wisely warns that modelling is simplifying reality and, because of this, the selective nature of models may not yield completely right conclusions. Therefore, models should always be viewed with some scepticism. Agri-environmental models are usually of three types: i.e. 1) optimisation models, 2) econometric models, or 3) simulation models. The choice of the modelling technique depends mainly on the available data. At the moment, no appropriate Finnish data on nutrient balances is available for econometric analysis. Therefore the approach used here is a combination of the modelling types 1 and 3.

At farm level, a farmer reacts to the measures which the policymaker has set in order to achieve a certain quality of the environment (i.e. the principal-agent framework). As the water quality issues are the most prevailing ones, let us inspect the consequences in the agricultural sector that follow from the proposition the environmental administration (SYKE 1995) has on reducing emissions to watercourses in Finland. The question to be answered is: "How do the NBCs compare with other policy measures in pollution abatement policy by the cost-efficiency criterion?".

The objective expressed by SYKE is to reduce the total Finnish nitrogen load to watercourses from the base level of 63,000 tonnes/year to 45,000 tonnes/year, i.e. by about 30%. For agricultural sector the goal for arable farming is to reduce the nitrogen load from the average load of 30,000 tonnes/year to 15,000 tonnes/year by the year 2005, i.e. 50%. Let us use the 30 percent reduction objective for agriculture in reducing nitrogen leaching to the watercourses as a yardstick for the policy assessment.

The cost-efficiency criterion is used for comparing different policy measures with each other in the context of the above-mentioned objective. A control measure is more efficient than another one if it yields the desired environmental quality at lower costs or if it yields at the same costs a better quality of the environment (Uimonen 1989). The cost-efficiency criterion can be expressed by marginal abatement (or average abatement) cost formula, i.e. the change in profit with respect to the change in nutrient runoffs. More formally:

$$MAC = (\delta\pi / \delta\phi) / (\delta\lambda / \delta\phi) = \delta\pi / \delta\lambda \qquad (13)$$

where π is profit,
ϕ is an economic policy instrument, and
λ is the nutrient runoff.

Three policy measures are assessed here, i.e. the application of: 1) tax on nitrogen in fertilisers, 2) levy on the nitrogen surplus (based on nutrient balance calculation), and 3) output tax. The approach is similar to Sumelius (1994) and Lankoski (1996), with some exceptions. Sumelius was more interested to test different crop response functions, and Lankoski studied only options 1 and 2 and with respect to barley. In this respect the cost-efficiency assessment for silage is studied, and the stochastic variables from climatic variability are incorporated into the modelling. Yield levels and nutrient balances are allowed to vary when simulating the weather response to the policy instrument ranking. The leakage of the nitrogen from manure is also examined here.

6.4.2. Production function

It is an extensive task to model a whole farm with its multiple inputs and outputs, and incorporate environmental factors into it. As the purpose of this research is to focus on assessing the feasibility of using the information from the NBCs in policy simulation, a more general model is needed. Because environmental problems in dairy farming stem from the use of manure, the role of manure as a production factor is inspected more thoroughly.

A crop response function (i.e. the effect of nitrogen input on silage yield) was estimated from the data of the bookkeeping farms from Kainuu. The derived yield function was:

$$y_s = \underset{(.937)}{1727} + \underset{(.820)}{22.01}*N - \underset{(-.393)}{0.03764}*N^2$$

where N is the amount of nitrogen input. The sample size was only 17 observations producing a rather unsatisfactory model with low a coefficient of determination (adjusted R^2=13%). The coefficients are not statistically significant at 5% risk level, yet they are of a logical sign (see Annex 5). Compared to the Heikkilä's (1980) function on a neighbouring area, i.e.

$$y_s = 3164 + 21.60*N - 0.0341*N^2$$

the estimated function is very similar, except for the constant term. Even though in the function by Heikkilä the constant is almost double that of the estimated function, this does not bring about a very big difference in the yield level at the common fertilising range. The estimated function yields a little lower biologically optimal level of nitrogen fertilisation than Heikkilä's function (i.e. 292 kg/ha vs. 317 kg/ha). Because of its better statistical properties, Heikkilä's crop response function is used in the modelling.

6.4.3. Leakage function

As there is no other appropriate leakage function available at present, a leakage function estimated by Simmelsgaard (1991) for the Danish circumstances is used here, and applied to Finnish conditions. The general form of the function is:

$$\ln(y / y_n) = b_0 + bN \tag{14}$$

where N = relative nitrogen fertilisation with respect to the normal intensity (1N) for fertilising the crop, the range of 0.5<=N<=1.0
y = leakage at fertiliser intensity level N
y_n = nitrogen leakage at average nitrogen use (1N)
b_0 = the constant
b = a parameter

The function measures changes in the leakage solely as a function of the fertiliser intensity level. Of course, fertilising level is not the only factor contributing to the leakage, as Simmelsgaard points out (the other main factors being the soil type, climate, and fertilising practices.). A function like this can be used to estimate average changes in the leakage as a consequence of a change in the fertilising practice over a number of years (not on a yearly basis), as Sumelius (1994) emphasises. Following Sumelius, this leakage function can be presented graphically (Figure 16). The normal fertiliser intensity level (1N) is here 90 kg/ha[18]. The parameter b has a value of 0.7, as in Sumelius' research

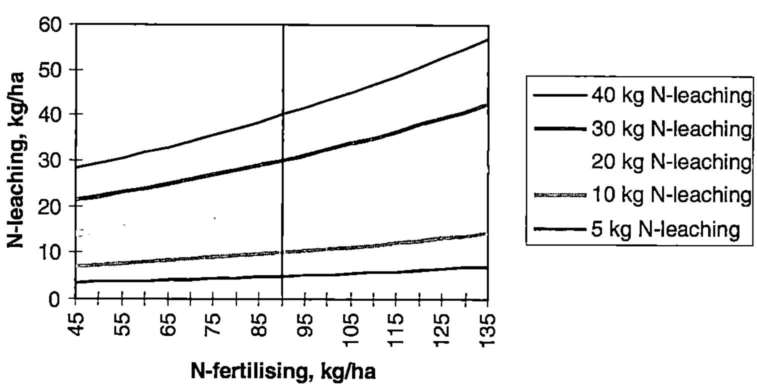

Figure 16. Approximated N-leakages at different fertilisation intensity levels.

[18] According to the FAEP the base fertilising level for fodder cereals is 90 kg/ha (MMM 1994).

and that of Lankoski (1996), hence yielding the function as $y=y_n e^{0.7(N-1)}$. Altering the value b has but a negligible effect on the leakage. The value for y_n changes from 5 kg/ha to 40 kg/ha, i.e. at the average level of 90 kg/ha (drawn as a vertical line in the figure) of N-fertiliser the N-leakage is between 5-40 kg/ha.

The figure indicates that e.g. on the level of an average nitrogen leakage of 20 kg/ha the leakage varies from 14.1 to 28.4 kg/ha, depending on the fertilisation level. From the figure above as well as from the leakage function itself it can concluded that the leakage function exhibits increasing returns to scale. The convexity of the leakage function is evident from the figure. The mathematical proof of this can be realised through the base assumption of $f(tx)>tf(x)$, when $t>1$ (see e.g. Varian (1990)). As we now have a semi-logarithmic function, $ln((y+t)/y_n)=ln(y+t)-ln(y_n)$ we get $ln(y+t)-ln(y_n)>tln(y/y_n)$, $y>0.5$. (In addition, if we assume the production function to be concave the emission function must be convex (Siebert 1995).) In other words, if the fertiliser intensity rises, a proportionally greater increase occurs in the leakage.

Simmelsgaard also estimated a leakage function for nitrogen in manure. He assumed that leakage from manure fertilising is 2.8 times greater than from mineral fertilising, thus presenting the following function:

$$y = y_n(4.6e^{0.7(N-1)} - 1.8) \tag{15}$$

Again, N represents the quotient between the real amount of manure applied to the normal level of manure application, and y_n is the leakage in normal level of manure application. Using Simmelsgaard's assumption of 40% nitrogen utilisation of the total nitrogen in the manure for plants we can calculate the leakage between 2 to 20 kg/ha. Under these assumptions we can draw a similar graph for nitrogen leakages from manure (Figure 17).

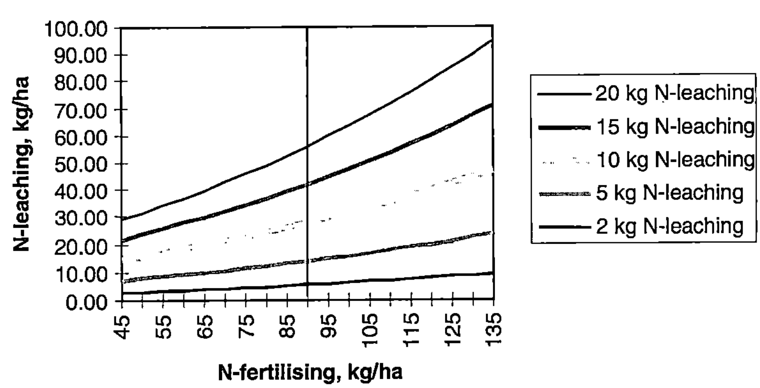

Figure 17. Approximated N-leakages at different manure fertilising levels.

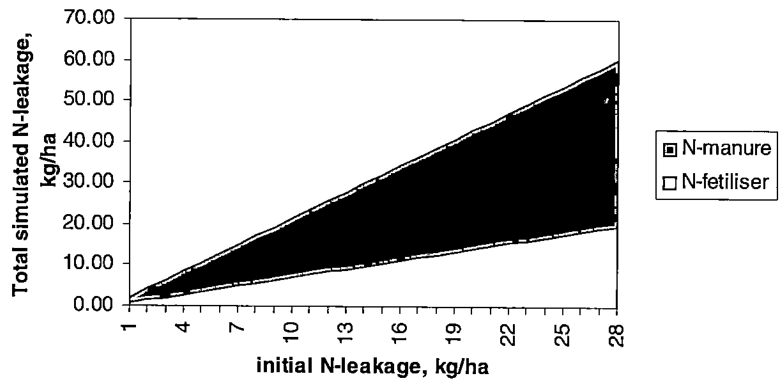

Figure 18. Leakage of nitrogen when 45 kg/ha of nitrogen is applied from both fertilisers and manure (totalling 90 kg/ha).

As can be seen in the figure, the slopes of the curves are much steeper than in the previous case. Here on a average nitrogen leakage of 10 kg/ha the leakage varies between 14.4 to 47.3 kg/ha, again depending on the fertilisation level. This indicates that the nitrogen in manure is more susceptible to leak to watercourses than fertiliser nitrogen. We can also combine these two functions. In a situation where both a fertiliser and manure are used corresponding to 45 kg of nitrogen per hectare, thus resulting in a total nitrogen application of 90 kg/ha, we get the following situation (Figure 18).

The figure also shows quite clearly that using commercial fertilisers reduces the risk of leakage. The leakage of nitrogen increases here, on average, twice (2.05 times) as fast when using manure than when a fertiliser is used.

6.4.4. Cost-efficiency scenarios

Analyses of the cost-efficiency of the policy instruments were conducted on an Excel spreadsheet. The input-output quantities of silage production on the 17 Kainuu dairy farms were used. In the base model, a farmer optimises the profit when encountering three different policy options, i.e. *a tax on a nitrogen fertiliser, a levy on the nitrogen surplus (i.e. an effluent tax)*, or *a tax on the silage price*.

The farm uses first all its manure at a fixed rate, thus changing the use of purchased commercial fertilisers. Nutrient balances were calculated by means of the surface balance method. The model does not allow for the option of transporting or selling manure elsewhere, or a technical change to take place. The model simply compares the economic and environmental implications under different policy schemes. Table 15 shows the results in the basic situation

Table 15. The effect of different policy instruments on silage production on Kainuu dairy farms.

abatement	level of instrument, FIM/kg			average abatement cost, FIM/kg of N			change in profit, FIM/ha		
	input tax*	NBCs levy	output tax**	input tax	NBCs levy	output tax	input tax	NBCs levy	output tax
5%	1.07	1.54	0.08	14.5	33.9	71.8	-102	-249	-451
10%	1.33	7.28	0.26	26.5	67.0	98.6	-472	-1,100	-1,627
15%	1.61	14.89	0.39	31.5	89.0	100.3	-821	-2,156	-2,400
20%	1.91	22.76	0.48	33.0	107.4	98.1	-1,153	-3,460	-3,051
25%	2.25	37.85	0.54	33.0	126.4	93.2	-1,470	-5,098	-3,620
30%	2.68	48.51	0.57	31.9	145.1	88.1	-1,770	-6,700	-4,204

* relative to the original price of nitrogen which, on average, was 7.47 FIM/kg.
** relative to the original price of silage, which was 0.30 FIM/kg.

where 5-30 percent abatement levels are met. The values of the parameters used in the model are presented in more detail in Annex 3.

As an example, a seven per cent tax on fertilisers, 1.54 FIM/kg levy on emissions, or eight per cent tax on silage will all reduce nitrogen emissions by five per cent. The average abatement costs (AAC) for these policy instruments range from 14.5 to 71.8 FIM/kg per abated kilo of nitrogen, and the loss of profit ranges between FIM 102 and FIM 451 per hectare.

The general conclusion from the table is that, of the three instruments, a tax on nitrogen is more cost-effective than either a levy on nitrogen surplus or a tax on the output. This can be seen either from the lower AAC or from the lower loss of profit at the corresponding abatement level. To achieve the goal of a 30% reduction[19] in the nitrogen runoffs, as proposed earlier, the change in the profit is about FIM 4,930 lower in taxing the inputs than in the case of a levy based on the nitrogen surplus, and about FIM 2,430 lower in the case of the output tax.

At lower levels of abatement, i.e. up to 15%, the levy system is more cost-effective than the output tax system. However, e.g. at a 30% level the output tax reduces profit about FIM 2,500 less than the levy system. The levy on nitrogen surplus has to be very high to yield more than 25% abatement levels – in fact, so high that the profit diminishes to negative at the range of 20-25% abatement in many cases.

[19] The results of the 30% level of abatement must, however, be considered only as trend-setting, because at individual farm level the optimum could not be found in a few cases, which lowers the average values of the level in question presented in the tables of this section.

The average abatement costs are also smallest in the input tax option. This means that reaching the desired pollution abatement is the least expensive under the input tax option. In addition, the farms having a low nitrogen surplus had a smaller than average difference of profit decrease between the two instruments.

The AAC for a nutrient surplus levy increases steadily as the abatement level increases, which is a logical result. The initial 5% abatement of runoffs is cheaper to conduct than to abate e.g. 30% of them. However, in the case of a tax on inputs the AAC first increases, but at the 25% abatement level it starts to decrease. The AAC for the output tax behaves similarly. The same phenomenon was also encountered in Lankoski's (1996) study. The reason for this stems from the price relationship between the inputs and outputs. At a certain abatement level (price level of fertilisers) it is not worthwhile for a farmer to increase inputs but reducing of them increases the profit, i.e. the production elasticity becomes smaller than one. More formal proof of this is presented in Annex 4.

However, the question whether it is valid to levy all nitrogen surplus when the farmer cannot very much control the amount of nitrogen that is lost from the manure during storing and spreading can be raised. A case in which a threshold value of nitrogen surplus was allowed was also analysed. The results showed that this allowance of nitrogen surplus needs to be about 80-110 kg/ha (i.e. from 60% to 85% of the surplus) to yield the same economic effects on farm profit as the tax system. These are very high figures, considering that the average nitrogen surplus of these 17 farms was 130 kg/ha. However, if the net surface balance and its average 30% of non-available nitrogen is used, a smaller 30% to 45% allowance is needed.

A sensitivity analysis in respect of silage price was conducted accordingly. Table 16 shows the results if the price of silage is 10% higher (i.e. 0.33 FIM/kg) than in the base model. The results are similar to the base scenario. Naturally, the tax and levy rates have to be higher in order to achieve the same results as previously, which, in turn, has an even higher effect on the profit. Again, a tax on nitrogen fertiliser stands out as the most cost-effective policy measure of these three alternatives. An output tax, in this scenario, becomes more cost-effective compared to a nutrient surplus levy already at the 15% abatement level.

Furthermore, the case where the silage price decreases 10% was also analysed. The situation is quite similar to the base scenario in respect of the policy ranking by the cost-efficiency criterion. The levels of taxes and the levy are a little lower, and so are the average abatement costs and profit loss, compared to the base scenario.

Finally, the stochastic changes of yields were allowed to take place. The average yields of silage during the past 11 years in Kainuu region were used to estimate the variation coefficient for the silage yield. From 1986 to 1996 the yields of silage varied from 12,600 kg/ha to 20,600 kg/ha. (Omitting any trends

Table 16. The effect of different policy instruments on silage production on Kainuu dairy farms; 10% higher silage price.

abatement	level of instrument, FIM/kg			average abatement cost, FIM/kg of N			change in profit, FIM/ha		
	input tax*	NBCs levy	output tax**	input tax	NBCs levy	output tax	input tax	NBCs levy	output tax
5%	1.17	3.18	0.14	22.6	60.3	102.5	-219	-603	-966
10%	1.55	11.35	0.35	35.6	105.6	133.2	-673	-2,077	-2,418
15%	1.98	22.39	0.49	38.7	134.7	123.8	-1,090	-3,979	-3,337
20%	2.46	37.71	0.58	38.8	164.1	110.6	-1,449	-6,495	-3,954
25%	3.03	46.46	0.63	37.0	155.7	108.6	-1,713	-8,873	-4,641
30%	3.22	65.62	0.69	36.4	176.2	97.7	-1,874	-11,976	-5,156

* relative to the original price of nitrogen which, on average, was 7.47 FIM/kg

* relative to the original price of silage, which was 0.30 FIM/kg

in the yield, the variation coefficient, which measures the variability in the silage yield, was calculated to be 11.9%.) Stochastic silage yields were computed according to the normal probability density function, $N(\mu,\sigma^2)$, the average yield being 16,746 kg/ha, and the standard deviation for the sample being 1,993 kg/ha. As noted in Section 5.1., the correlation between the nitrogen surplus and the cereal yield was found to be about 90%. Therefore, it was assumed that the variability in the silage yields has an additional ±5% stochastic effect on the nitrogen surpluses of silage. In addition, the stochastics had an effect on the average leaching coefficient indicated by the average nitrogen surplus.

Two scenarios were run, i.e. the effects of a 'bad year' and the effects of a 'good year' on the cost-efficiency. The farm sample was reduced to six farms. 'Bad year' means a situation where the yields are low and leakages (nitrogen surpluses) are high, and 'good year' means a situation where the yields are high and leakages low. The results of the bad-year scenario are shown in Table 17.

The effect of unpropitious weather conditions on the input tax is negligible since the only changing factor is the nutrient surplus (Table 15 vs. Table 17). The response is of a minor magnitude with respect to the actual level of input tax, the AAC, and the loss of profit. As discussed in Section 4.1., the farmer acts upon his expectations on normal weather conditions and fertilises accordingly. The unfavourable growing conditions have an influence on the yield, but since the financial incentive is upon the inputs he uses, the economic implications are minor compared to the base situation.

The policy option on levying the nitrogen surplus, however, is seriously influenced by weather. At the initial 5% abatement level the levy almost doubles. The AAC and the loss of profit are also much higher than in the base

Table 17. The effect of different policy instruments on silage production on Kainuu dairy farms; weather response simulation (bad year).

abate-ment	level of instrument, FIM/kg			average abatement cost, FIM/kg of N			change in profit, FIM/ha		
	input tax*	NBCs levy	output tax**	input tax	NBCs levy	output tax	input tax	NBCs levy	output tax
5%	1.12	3.01	0.11	20.8	56.6	93.2	-156	-429	-689
10%	1.45	12.09	0.31	37.6	112.2	138.2	-548	-1,668	-1,984
15%	1.81	24.16	0.44	41.7	145.1	132.3	-909	-3,233	-2,833
20%	2.21	40.62	0.54	42.2	176.6	120.0	-1,222	-5,256	-3,414
25%	2.66	59.96	0.61	40.5	201.3	106.9	-1,460	-7,455	-3,789
30%	3.00	66.10	0.67	38.8	176.7	97.9	-1,608	-8,507	-3,836

* relative to the original price of nitrogen which, on average, was 7.47 FIM/kg
* relative to the original price of silage, which was 0.30 FIM/kg

scenario. In this setting, the levy system is more cost-effective than the output tax system only below the 10% abatement level. Thus, the ranking of the policy instruments by the cost-efficiency criterion can change, if unfavourable weather in terms of the growing conditions prevails. Similarly, the effects of prosperous growing conditions are shown in Table 18.

Comparing the two tables above reveals that the effect of weather conditions are the most fundamental ones in respect of the NBCs levy. The levels of input tax and output tax are the same in both the tables, whereas the nutrient surplus

Table 18. The effect of different policy instruments on silage production on Kainuu dairy farms; weather response simulation (good year).

abate-ment	level of instrument, FIM/kg			average abatement cost, FIM/kg of N			change in profit, FIM/ha		
	input tax*	NBCs levy	output tax**	input tax	NBCs levy	output tax	input tax	NBCs levy	output tax
5%	1.12	2.05	0.11	14.22	38.6	72.0	-156	-428	-779
10%	1.45	8.25	0.31	25.63	76.6	107.3	-548	-1,668	-2,254
15%	1.81	16.49	0.44	28.48	99.0	103.0	-909	-3,233	-3,225
20%	2.21	27.73	0.54	28.80	120.5	93.6	-1,222	-5,256	-3,897
25%	2.66	43.33	0.61	27.63	144.7	83.6	-1,460	-7,914	-4,335
30%	3.00	55.55	0.67	26.50	148.1	76.6	-1,608	-9,219	-4,551

* relative to the original price of nitrogen which, on average, was 7.47 FIM/kg
* relative to the original price of silage, which was 0.30 FIM/kg

levy decreases by 30%. In addition, in the case of the input tax the change in profit remains the same regardless of the weather effect. The loss of profit in the nutrient surplus scheme is in essence the same in both tables. Naturally, the loss of profit in the output tax scheme is higher, because the yield levels are higher, thus resulting in higher tax payments.

Average abatement costs are uniformly lower in the good-year scenario than in the bad-year scenario. The AACs are about 1/3 lower in a good year than in a bad year in the input tax and NBCs levy options. With the output tax option the difference is slightly smaller.

In general, the ranking of these three instruments does not very much change. In terms of cost-efficiency criterion the input tax ranks the best. At low levels of abatement the nutrient surplus levy is more cost-efficient than output tax, but at higher levels of abatement the situation is vice versa. Therefore, we have to conclude that the second-best solution, i.e. an input tax, in this scenario setting turns out to be better than the first-best policy, i.e. an emission levy based on the nutrient balance, according to the cost-efficiency criterion.

6.5. Which policy instrument to use at a farm level?

The analyses conducted in the previous section examine the choice of a policy instrument solely from the perspective of cost-efficiency. In addition, policy-makers must use other criteria when choosing 'the right' policy instrument. As discussed in Section 4.5. there are many different criteria for the evaluation of policy instruments. The selection of the criteria, in turn, affects the choice of the policy instrument. The framework presented by Braden and Segerson (1993) offers a base structure for assessing different criteria aspects (on page 55). Their expression 'ability to target' is here re-formulated as 'site- and time-specificity.' Their other two criteria are enforceability and correlation to water quality. Following their approach, and including both the cost-efficiency criterion and the information dimension criterion, we get Table 19. The cost-efficiency ranking is entered to the table from the results of the modelling scenarios. The information dimension is more of a normative assessment, but it is, in most part, based on the findings of Lankoski (1996) and Tamminen (1997).

The evaluation of these criteria is – to some extent – a normative task. Economists, and other social scientists, simply lack the proper tools to rank many of these characteristics quantitatively. Therefore the rankings presented here should not be taken as absolute ratings, but rather as a general level of an assessment. In the following, each of the three policy instruments scrutinised in the previous section is reviewed in terms of the criteria presented in the table below.

Input tax: Even though there seems to be a rather high correlation between the nitrogen inputs and their emissions, stochastic factors, such as weather,

Table 19. Evaluation of policy instruments.

Policy instrument of principal	Rating with respect to site- and time-specificity	enforce-ability	correlation with water quality	cost-efficiency	information dimension
Input tax	L	H/L	M	H	M
Emission tax (N-surplus)	H	M	M	M	H
Output tax	L	H/M	L	M	L

L means low, M means medium, and H means high rating.

often interfere with this relationship and result in relatively high variations in the emission output. Furthermore, taxes applied to quantities cannot be very site- or time-specific. This means that the tax is paid by fertiliser unit regardless of where, when, and for which crop the fertiliser unit is used. Enforcing a tax on compound fertilisers is an easy task for the principal, since the tax is added to the price of the fertiliser. However, in setting a tax on manure the situation becomes more complex, because the amount of manure produced on the farm needs to be assessed somehow. This is often a difficult task. The cost-efficiency of a input tax is the best of these three methods. In addition, the information content of the instrument is relatively good, because when facing a tax on input the farmer becomes aware of the connections involved in the use of inputs; and he also realises the threat his action poses to the environment. However, this link may remain unclear, in particular, if the quantity of the input purchased is not very closely associated with the amount of pollution emissions generated.

Emission tax: Because it is based on the nutrient surplus, the emission tax is a very site- and time-specific policy instrument. The nutrient balance captures the effects of farm level emissions; and if calculated on a parcel level gives rather detailed information on the real emission outputs in the production system. Even though the nutrient balances are usually calculated at farm level and, consequently, the NBCs information does not necessarily reveal the actual losses of nutrients to the environment, the information dimension is the highest among these three policy instruments. The correlation between emission tax and the water quality is obscured by the sometimes unclear link between the nutrient surplus and the real nutrient run-offs to watercourses. Cost-efficiency of this policy instrument seems to rank second among these three instruments, especially at lower levels of pollution abatement. However, the administrative costs (enforceability) of setting up a system to collect data on nutrient balances for taxing purpose can be extensive. And, as discussed in Section 2.5., a levy system may intrinsically lead to adverse selection by tempting the farmers to 'underestimate' the inputs they use and 'overestimate' the outputs that are produced, thus leading to ostensibly low nutrient surpluses, but, in fact, to environmentally unpropitious outcome.

Output tax: The third policy instrument assessed here, the supply oriented output tax, ranks highest in terms of enforceability, but this is somewhat ambiguous. The outputs a farm sells can be easily counted, but the outputs the farm uses internally, e.g. silage, are more difficult to assess quantitatively. (The situation is similar to the case above when assessing the amount of manure for input tax purposes.) The output-based approach can not be easily targeted to sensitive areas or times, which means low site- and time-specificity. Furthermore, the output levels are not necessarily related to the water quality problems; as it is actually the way the outputs are produced that has the most significant impact on the watercourses. Thus, the rankings of the policy instrument, with respect to both site- and time-specificity and the correlation to water quality, remain low. The cost-efficiency of the instrument is the weakest of these three instruments; except in the case of over 20% abatement level, where it ranks better than the emission tax. The information dimension of the output tax is low, because of its low correlation to the actual water quality.

As can be seen in table p above, no single policy instrument rates the best in terms of all criteria. Therefore, the principal has to weigh these rankings when selecting the proper policy instrument to tackle the agricultural nutrient runoff problem. Cost-benefit analysis may be one option for the principal to perform.

The criteria presented above are, by no means, the only relevant ones. Just as the welfare concept Pareto-optimality in itself says nothing about the *fairness* or distribution effects of the solution, the criteria above also lack the discussion of the fairness of the policy instruments. The fairness of the policy instrument depends on the weather response to production and on other exogenous variables. As the farmer cannot control these factors, a policy instrument based on the nutrient surplus becomes unjust, because uncontrollable factors may result in penalising his production activity regardless of his actions. This is, in essence, the same phenomenon touched earlier in Section 2.3., where it was argued that the weather effect on nutrient surplus may screen the true impacts of environmental policies, if the impacts are measured by the NBCs.

7. Conclusions and discussion

The last two previous chapters above showed first how important a role the weather plays in the build-up of the nutrient surpluses, and, secondly, what this means in the policy formulation. As noted in Chapter 5, different crops have varying levels of nutrient surpluses (and nutrient utilisation). For example, rye has, on average, two- to four-fold higher nitrogen surpluses compared to barley and oats, whereas potatoes have clearly higher phosphorous surpluses compared to rye, barley, or oats. Therefore, the choice of a crop the farmer cultivates has a dramatic effect on the nutrient surpluses. However, we need to remember that the nutrient surpluses are also natural phenomena, i.e. the nutrient emissions are to some extent unavoidable, and reducing the emissions to zero is impossible.

At farm level, the options for adjusting the production with respect to the weather effects are limited. If a sanction system, e.g. a tax, solely based on the nutrient surplus is set, the 'naturally' most polluting crops are penalised more heavily than the crops with more efficient nutrient utilisation processes. This would have an effect on the choice of crops the farmer wishes to cultivate.

The NBCs seem to be like a double edged sword; i.e. they make it possible to assess the farm specific nutrient emissions, but they are obstructed by various factors lowering the accuracy and usability of them. However, once these problems are perceived they can be solved; and, in many cases, it is eventually the end use of the NBCs that justifies the way and the precision of the method the nutrient balances are calculated with.

In Chapter 6 an optimisation model was build up to assess dairy farms' silage production employing profit maximising approach. The model was coupled with a leaching function, and adjusted with the farm specific nitrogen surpluses. The cost-efficiency of three policy instruments, i.e. a tax on a nitrogen fertiliser, a levy on the nitrogen surplus (i.e. an effluent tax), or a tax on the silage price, was assessed. The results of the model suggest that in terms of the cost-efficiency criterion the best policy option to reduce agricultural nitrogen runoffs in silage production is setting a tax on fertiliser inputs. This holds true if the price of the output changes, and also if the stochastic impacts of the weather on the yield and on nutrient utilisation are taken into consideration. The tax on output (silage) is in general the least cost-effective of these three policy instruments. However, at high abatement levels of nutrient runoffs, i.e. over 20%, the tax on output becomes more cost-effective than the tax on nitrogen surplus.

These findings slightly contradict those of Fontein et al. (1994) who state that a levy on the surplus of nitrogen is a more cost-efficient way to reduce the output of nitrogen (emissions) than a levy on feed in pig production. Yet, Lankoski's (1996) results, in modelling grain production, support the findings of the present study by noting that the cost-efficiency of different instruments is sensitive to the changes in the abatement levels, as well as to initial leakage levels. It is obvious that the results of different models depend on the modelling structure, and thus they are not fully comparable.

The variation due to weather factors in yields and nutrient surpluses had a rather small effect on the overall cost-efficiency ranking. However, stochastic impacts may have an effect on the ranking between the output tax option and nutrient surplus levy. The weather impact on the choice of a policy instrument seems to be of rather minor importance. In extreme weather conditions, though, policy impact assessment utilising the NBCs may become distorted.

The modelling of silage production indicated that setting a tax on an input is a more cost-efficient way to abate nutrient runoffs than a tax based on nitrogen surplus or a tax on silage output. Cost-efficiency is but one criterion, and the policymaker then has to assess other criteria in the final policy choice, also. The criteria which are selected for the final policy assessment play an important role in guiding the decision-making process.

If the objective is to enhance water quality by reducing agricultural nutrient runoffs, the solutions should be very site- and time-specific. Usually, localised measures are needed to reach the desired environmental response – first, in the agricultural production practice, and, secondly, in the nutrient emissions. Unfortunately, highly differentiated regional or local programmes lead to high transaction and administration costs. In addition, policies hindered by information problems, such as the measurability problem of nonpoint emissions, give rise to adverse selection from the farmers' part, which needs to be accounted for in the policy formulation.

In the light of the discussion on environmental policy measures, it must be proposed that a functional environmental policy design needs to respect the information advantage the farmers posses on the environmental position of their farms. Some economists are already talking about the third phase in the environmental control policies. This means that the conventional command-and-control policies (i.e. standards or regulations) or economic instruments are being replaced by information policies.

Ample literature concerning the control of nonpoint pollution has emerged during the recent years. Many of these findings advocate some type of incentive schemes or farm co-operation in reducing agricultural nutrient runoffs. Quite often these models rely on the information qualities of the policy measures (e.g. Xepapadeas 1997, Romstad 1997). Farmers' awareness of agri-environmental linkages is improved and they can opt for the most suitable methods to reduce the pressure their production imposes on the environment.

Although this study did not attempt to quantify the effect of the information dimension, there is evidence (e.g. Tamminen 1997) that 'upgrading' farmers' knowledge on the environmental issues has made them more apt to carry out environmental investments. Bosch et al. (1995) noted that farmers in Nebraska, who had received test information of nitrogen from soil fertility tests, were more eager to use that information in nitrogen application decisions than farmers with no soil testing information. Berentsen and Giesen (1997) also emphasise the role of information measures in the nutrient runoff abatement policies. Consequently, more emphasis needs to be addressed to the information content of the policy measures. But what can we say for policy recommendations?

The present Finnish Agri-Environmental Programme (FAEP) is mainly a uniform programme where the criteria are only to some extent differentiated by region (e.g. the fertilising base levels, and the green cover criterion). The new

FAEP will be introduced in 2000. Further development of the FAEP requires more attention to site-specific criteria and to the information dimension of the measures. For example, the green cover criterion is 30% for all farms in Southern Finland (areas A and B), regardless of the proximity to watercourses. This means that farms must have at least 30% of the arable land area covered by plants or plant residues outside the growing season, or an approved method of reduced tillage must be used (MMM 1994). Therefore, the farms located further away from the watercourses face the same requirement as the farms having a direct connection to rivers, lakes, or the sea. In addition, adverse selection has taken place when farmers have started to cultivate by means of reduced tillage which in its heavy form very much approaches ploughing. Compared to ploughing, reduced tillage seems to increase the risk of erosion and the drifting of soluble phosphorous to watercourses (Turtola 1997).

The present FAEP presents several criteria for farmers to fulfil in order to be eligible for a compensatory payment. This is a very regulatory approach. An information policy approach would be to introduce the environmental goals and the means for attaining the goals by providing the farmers with knowledge and know-how on the ways to improve the environmental efficiency of their farms. Nutrient balance calculations provide one way in which this efficiency can be assessed both at the farm level and at a more general level. Thus, the NBCs information serves both the farmer and the policymaker in assessing the environmental impacts of the programme. However, the limitations of the calculations must be recognised and accounted for.

Presently, information policies are used in connection with other environmental policy instruments, but the information policy approach has started to gain more importance as a policy measure in itself. The information policy approach needs much more integration between the farmer, extension services, and administration. It is a challenging approach, but overcomes some problems that obstruct the use of more direct policy instruments. Although there may be scepticism concerning the efficacy of the information policy approach, it is clear that this approach leaves more options for the farmer to choose from in the nutrient runoff abatement; and maybe the farmer knows something the policymaker does not know about the feasible ways, for his part, to contribute to a better, cleaner environment.

8. Summary

Agricultural impacts on the environment in Finland are manifold. These impacts can be both positive (e.g. landscape and biodiversity) or negative (e.g. nutrient run-offs and erosion). These are also called the positive or the negative externalities of production. Agriculture is a sector contributing the most to the nitrogen and phosphorous runoffs to watercourses, compared to the other sectors of the Finnish society. Other sectors, such as industries, have been able to reduce nutrient emissions by applying cleansing mechanisms or procedures to their production systems. This has been possible because the emissions, e.g. in industrial processes, emanate from certain distinct sources, i.e. the pollution comes from a point source origin. However, agricultural nutrient runoffs are mainly of nonpoint origin. This means that it is not possible to point out where the runoffs measured in ambient concentration originate from, once they have leached from the arable land to the watercourses. Consequently, it is not possible to point at the true polluters of the watercourses, even though the rising nutrient content is observed. This makes the agri-environmental policy design more challenging.

This research studied the possibilities of reducing agricultural nutrient emissions to the environment by means of different policy instruments. These policy instruments can be categorised as economic instruments, direct regulations, and information policy. Much of the past research emphasis has been on studying the optimal solutions to environmental problems by using economic instruments or standards in the agri-environmental policy design. However, the role of information policies as a control instrument in itself is rising; and the nutrient balance calculations (the NBCs) can also be seen in that light.

In general, the NBCs imply the difference between the nutrients entering and the nutrients leaving an agricultural system, typically resulting in a surplus of nutrients. The two most widely used methods to calculate the NBCs are the farm gate balance and the surface balance. The NBCs are used for different purposes, e.g. for farm production planning or at regional and national level for assessing the impact of agriculture on the environment, as well as the effect of agri-environmental policies on agricultural nutrient runoffs. Thus, interest in assessing the agricultural impact on the environment with the NBCs is growing.

The nutrient balance calculations can be seen as a way to inspect farm pollution (nutrient runoffs) from the point source perspective. Thus, the traditional nonpoint aspect of agricultural pollution is avoided, and conventional environmental control policies can be directed to the polluting discharges. As charming as this seems, we should not make hasty assumptions on the feasibility of the NBCs in environmental control policies before scrutinising the problems which hinder the use of the NBCs as well.

One major obstacle in using the nutrient balance calculations in agri-environmental policy is that the NBCs do not convey the information about where the nutrients leave to from the agricultural system. The NBCs just state and quantify the situation but do not reveal whether the nutrients accumulate to the soil, leach to watercourses, or volatile into the air. Therefore, if such a policy measure based on the nutrient balance calculation is set up, the specific environmental effects, e.g. with respect to the water quality, may remain unclear.

Perhaps even more importantly, the weather plays an important role in the agricultural production process in the northern latitudes. Crop yields, and consequently the nutrient surpluses, vary considerably in Finland because of the northern climate. For example, the year 1987 was very poor for crop growth in Finland and the nutrient uptake was 30% lower than in the previous year, thus resulting in high nutrient surpluses (on average 97 kg/ha).

Furthermore, there are difficulties related to the accuracy of the NBCs. These include e.g. the accuracy of the coefficients used to assess the amount of nutrients in the animal manure and in the nutrient uptake by crops.

In addition, the purpose for which the NBCs are used is reflected in the accuracy of the calculations. If farmers feel the NBCs give them new and relevant information for improving the effectiveness of nutrient use, they will provide accurate information for the NBCs. However, if they feel that the NBCs are used in penalising their farms, the information aspect is shadowed by the strategic behaviour of reporting the NBCs as low as possible.

The sensitivity of crop response to weather conditions leads to similar responses with respect to the nutrient surplus. An analysis of the variation of nutrient surpluses in different crops showed that Finnish nitrogen surpluses may vary considerably from year to year. Potatoes and oats, in particular, are very sensitive to night frost, and if such a thing happens that easily leads to high nutrient surpluses. The temperature, especially in August, is significant for the build-up of nutrient surpluses, The higher the degree-days in August is the lower is the nutrient surplus, and the better the nutrients are utilised. The data also suggested that proper precipitation is significant for the build-up of the nutrient surpluses. Excessive rain in May and August, in particular, result in high nutrient surpluses. Thus, cold summer and wet May and August usually lead to high nutrient losses. However, phosphorous surpluses are much more stable than nitrogen surpluses with respect to weather factors.

Farm economic indicators were compared to the nitrogen surpluses of Kainuu dairy farms. The results suggest a clear link between the cost structure and the nitrogen surpluses. The correlations are positive (i.e. between 0.56 and 0.65 in the case of total costs) meaning that high costs per hectare result in high nitrogen surpluses and, consequently, high nutrient emissions to the environment. In other words, the lower production costs coincide with efficient nitrogen use. The main component contributing to a high correlation was found to be the cost of fertilisers. The correlations were in the range of 0.64 to 0.69, depending on the nitrogen surplus category. Therefore, farms having low fertiliser cost also have low nutrient surpluses. In addition, there seems to be a tendency that larger farms are, on average, more efficient in using the nutrients measured by the cattle balance.

In addition to the statistical analyses a profit maximising optimisation model was build up. Farm level comparison of the cost-efficiency of input tax, emission levy (based on the NBCs) and output tax showed that in silage production where farm uses both commercial fertilisers and manure, the tax on nitrogen is, in general, the most cost-efficient one in abating nitrogen leakage. An emission levy exhibited higher cost-efficiency than output tax at low abatement levels.

The effect of weather conditions on the cost-efficiency ranking showed that if the weather is unfavourable for crop growth the emission tax is more cost-effective than output tax only at 10% abatement level. Input tax is by far the most cost-efficient policy instrument of these three even under stochastic weather effects. If growing conditions turn out to be better than normal, the ranking does not change. However, the relative efficiency of all three policy instruments improves, i.e. the average abatement costs decrease.

The final choice of a policy instrument is a matter of many perspectives, cost-efficiency being one of them. In addition, criteria such as how easily the instrument is enforced, and how time- and site-specific the environmental responses are, play an important role in the final policy ranking. Furthermore, the information dimension also has to be considered in the policy choice. The better the policy instrument communicates information both to the farmer and the policymaker, the more willing the farmer (the agent) is to provide as precise as possible data on the true impact his production has on the environment, and the better the policymaker (the principal) is conscious of the results of the policy it has established. With this respect, the NBCs provide the best information dimension.

The NBCs can be viewed as a way to simplify the use of the command-and-control policy measures. Instead of setting up multiple regulations aiming at improving the environmental effects of farming practises, calculating the nutrient surpluses gives a simple assessment on the impact a farm has on the environment. Then, it is up to the farmer to decide what to do to ameliorate the nutrient use efficiency at his farm.

The NBCs are already used in agri-environmental policy, e.g. The Netherlands is moving to a system where farms face a financial sanction based on the farm's nutrient surplus of phosphate. The Dutch approach is understandable because of the scale of the excess-nutrient problem. In Finland, however, levying the surpluses is a too strong of a procedure. Levying emissions does not seem to be very feasible policy instrument because of its low cost-efficiency.

The nutrient balance calculations can give important information to farmers and administration on the impacts agriculture has on the environment. The accuracy and feasibility of this information depends on many different factors. These factors must be taken into consideration in calculating the nutrient balances, and before using the information of the NBCs at farm level, or for agri-environmental policy formulation.

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SELOSTUS

Ravinnetaseet maatalouden ympäristöpolitiikassa

Reijo Pirttijärvi

Maataloustuotannon ympäristövaikutukset ovat moninaisia. Nämä vaikutukset voidaan luokitella joko positiivisiksi, kuten maaseutumaiseman kokeminen ja biodiversiteetin lisääntyminen, tai negatiivisiksi, kuten ravinnehuuhtoumat ja eroosio. Tässä yhteydessä taloustieteilijät puhuvat positiivisista ja negatiivisista ulkoisvaikutuksista. Maatalous on tätä nykyä suurin vesistöjen ravinnekuormitusta aiheuttava yksittäinen sektori. Muut yhteiskunnan sektorit ovat kyenneet vähentämään ravinnekuormitustaan asentamalla päästöjään pienentämään erilaisia puhdistusmekanismeja. Tämä on ollut mahdollista koska esimerkiksi teollisuuden päästöt ovat luonteeltaan pistekuormitusta, jonka kontrolli on tällä tapaa mahdollista. Maatalouden ravinnepäästöt ovat kuitenkin pääosin hajakuormitusta, jonka mittaaminen ja kontrollointi on paljon hankalampaa. Nämä hajakuormituksen mittaamiseen ja kuormittajan tunnistamiseen liittyvät vaikeudet ovatkin johtaneet siihen, että periaatteessa tehokkaimpia päästöihin kohdistuvia ympäristöpoliittista toimenpiteitä ei ole voitu toteuttaa.

Ympäristöongelmiin on yleensä puututtu joko 1) hintaohjauksen (esim. verojen) tai 2) määräohjauksen (esim. ympäristönormien asettaminen) välineillä. Eräät taloustieteilijät ovat kuitenkin alkaneet puhua ympäristöohjauskeinojen kolmannesta aallosta, jolla tarkoitetaan 3) tiedollista ohjausta. Tiedollinen ohjaus (eli informaatio-ohjaus) on aiemmin mielletty kahta edellistä tukevaksi ja täydentäväksi toimenpiteeksi, mutta se on alkanut nyttemmin saada jalansijaa myös itsenäisenä ohjauskeinonaan. Tiedollisen ohjauksen tarkoituksena on vaikuttaa ihmisten asenteisiin ja käyttäytymiseen välittämällä heille kohdennettua ympäristön tilaa koskevaa tietoa, ja näin saada aikaan muutosta heidän käyttäytymisessään ympäristön suhteen.

Perinteisesti maatalouden kuormituksen on nähty olevan pääosin hajakuormitusta, mutta ravinnetaselaskelmat tarjoavat mahdollisuuden tarkastella maatalouden ympäristökuormitusta pistekuormituksen näkökulmasta. Ravinnetaselaskelmilla mitataan tuotantoprosessiin käytetyissä panoksissa tulevien ja lopputuotteisiin sitoutuneiden ravinnemäärien erotusta. Tyypillisesti tämä erotus osoittaa ravinneylijäämien syntyä eli tuotantoprosessiin on tullut enemmän ravinteita kuin sieltä on lähtenyt. Taselaskelman lopputulos ilmoitetaan yleensä peltohehtaaria kohden (kg/ha) tai tuotos-panos –suhteena (hyötyprosentti).

Tämä tutkimus tarkasteli maatalouden ravinnetaseiden käyttöä ympäristökuormituksen vähentämisessä. Ravinnetaseiden avulla saadaan käsitys siitä, missä määrin maatila (tai yleisemmin laajempikin tuotantokokonaisuus) kykenee hyödyntämään tilalle tulevia ravinteita ja missä määrin ne kulkeutuvat ohi varsinaisen tuotantoprosessin.

Ravinnetaselaskelmien käyttö on yleistymässä tuotannon suunnittelussa. Tilan ravinnevirtoja kuvaavat laskelmat voivatkin antaa tilanjohdolle arvokasta tietoa tuotannon suunnitteluun. Myös hallinnon piirissä on herännyt kiinnostus ravinnetaselaskelmia kohtaan. Viranomaiset toivovat saavansa laskelmista tietoa ympäristöpolitiikan vaikutuksista, esimerkiksi maatalouden ympäristötukiohjelmaa arvioitaessa.

Tässä tutkimuksessa tarkasteltiin ravinnetaselaskelmien metodologiaa, ravinnetaseisiin liittyvää tutkimusta sekä taselaskelmiin liittyviä ongelmia. Taselaskelmia on olemassa lähinnä kolme eri päätyyppiä: 1) ravinteiden porttitase, 2) peltotase ja 3) karjatase. Ravinnevirtojen tarkastelupisteet ovat näissä joko tilan portilla (panosten ostot – tuotteiden myynnit), peltopinnalla (kasviravinteet – sato) tai karjarakennuksessa (eläinten rehut – eläintuotteet).

Ravinnetaseisiin liittyy kuitenkin kaksi perusongelmaa, jotka vaikeuttavat ravinnetaseinformaation käyttöä ympäristöpolitiikan työkaluna. Ensinnäkin ravinnetaselaskelma, joka yleensä osoittaa tilalta muodostuvan ravinneylijäämiä, ei kerro sitä minne ravinteet joutuvat. Ravinnetaselaskelma vain toteaa ravinneylijäämien olemassaolon, mutta ei ota kantaa siihen joutuvatko ravinteet vesistöön, haihtuvatko ne ilmaan vain jäävätkö ne maaperään. Näin ollen, mikäli asetetaan ravinneylijäämiin kohdistuva ympäristöpoliittinen toimenpide (esim. maksu tai vero), ei voida olla varmoja siitä missä määrin maatalouden ravinnepäästöt vesistöihin todella pienenevät.

Toiseksi, säätekijät vaikuttavat Suomen oloissa merkittävästi ravinneylijäämien syntyyn. Satotasot (ja siten ravinneylijäämät) vaihtelevat Suomessa pohjoisen ilmaston vuoksi huomattavasti. Esimerkiksi vuoden 1987 kasvukauden sääolot olivat erityisen epäedulliset, jolloin kasvien typen hyödyntämisen taso oli 30 % pienempi kuin edellisenä vuonna. Ravinneylijäämät olivat tuona vuonna poikkeuksellisen korkeita (keskimäärin 97 kg/ha).

Ravinnetaselaskemisen tarkkuuteen vaikuttaa se, kuinka tarkkoja kertoimia on käytössä. Esimerkiksi peltotasetta laskettaessa lannan ravinnemäärien oikealla arvioinnilla on keskeinen merkitys laskelman lopputuloksen kannalta. Myös sadossa poistuvien ravinnemäärien arviointi vaikuttaa laskelmaan keskeisesti.

Se, mihin ravinnetaselaskelmia käytetään, heijastuu myös taselaskelmien tarkkuuteen ja luotettavuuteen. Jos viljelijät kokevat saavansa taselaskelmista hyödyllistä tietoa tilansa tuotannon suunnitteluun ja seurantaan, he pyrkivät tekemään laskelmat mahdollisimman luotettavasti ja tarkasti. Jos viljelijät pelkäävät ravinnetaseinformaatiota käytettävän heidän tuotantonsa kontrollointiin (esim. ravinnetasevero), viljelijöiden strateginen käyttäytyminen häivyttää taselaskelmien informaatioulottuvuuden, kun viljelijät pyrkivät ilmoittamaan laskennalliset ravinneylijäämät mahdollisimman pieninä.

Varianssianalyysi eri kasvien (ohra, kaura, ruis ja peruna) suhteen osoitti, että ravinneylijäämät vaihtelevat 11 vuoden ajanjaksolla melkoisesti erityisesti perunan ja kauran suhteen, jotka ovat hallanarkoja viljelykasveja. Säätekijöiden vaikutus fosforiylijäämien syntyyn on kuitenkin typpiylijäämien syntyä vähäisempää. Tutkimus osoitti, että erityisesti toukokuun sademäärillä ja elokuun lämpötiloilla on keskeinen merkitys ravinneylijäämien syntyyn. Säätekijöillä on kuitenkin kaksijakoinen vaikutus kasvien ravinteiden hyödyntämisessä. Toisaalta liian märkä ja toisaalta liian kuiva toukokuu saavat aikaan saman suuntaisen vaikutuksen ravinteiden hyödyntämisessä (korkea ravinneylijäämä ja alhainen hyötyprosentti). Niin ikään kuuman tai kylmän sään vallitessa kasvien ravinteiden hyödyntämiskyky heikkenee. Lämpötilasummalla on sademäärää merkittävämpi vaikutus ravinteiden hyödyntämisessä. Toisin sanoen tutkitut neljä kasvia ovat herkempiä reagoimaan ravinteiden hyödyntämiseen lämpötilasta johtuvien tekijöiden vaikutuksesta kuin sademäärien muutosten vaikutuksesta.

Tutkittaessa Kainuun maitotiloilla tilan taloudellisten indikaattorien ja typpiravinneylijäämien suhdetta aineistosta löytyi joitakin tilastollisesti merkitseviä riippuvuuksia. Korkeat tuotantokustannukset (mk/ha) korreloivat voimakkaasti korkeiden ravinneylijäämien (kg/ha) kanssa (korrelaatio r = 0.56 - 0.65). Erityisesti lannoitekustannus korreloi varsin vahvasti ravinneylijäämän kanssa (r = 0.64 - 0.69). Näin ollen tilat, joilla on alhainen lannoitekustannus, omaavat myös alhaisen ravinneylijäämän typen osalta. Lisäksi kävi ilmi, että kooltaan suuremmat tilat näyttävät olevan hieman tehokkaampia ravinteiden hyödyntäjiä karjataseen hyötyprosentilla mitattuna.

Tilastollisten analyysien ohella tutkimuksessa tarkasteltiin optimointimallien avulla tilannetta, jossa tila käyttää sekä ostolannoitetta että tilan omaa karjanlantaa säilörehun tuottamiseen, ja tilan tuotantoon kohdistuu typpipäästöjen vähennykseen tähtäävä taloudellinen ohjauskeino. Optimointimalleissa tutkittiin kolmen ohjauskeinon (panosvero, päästövero eli ravinneylijäämävero ja tuotevero) kustannustehokkuutta pyrittäessä pienentämään maatilan typpipäästöjä 5-30 prosenttia.

Tulosten mukaan typpipanokseen kohdistuva vero oli näistä kolmesta kustannustehokkain kaikilla päästövähennystasoilla. Toisin sanoen pyrittäessä alle 30 prosentin typpipäästöjen vähentämiseen sekä typen ravinnetaselaskelmaan perustuva päästövero että säilörehun määrään kohdistuva vero aiheuttavat korkeammat tulonmenetykset kuin typpipanokseen kohdistuva maksu. Säilörehun määrään kohdistuva lopputuotevero on pääsääntöisesti päästöveroa tehottomampi keino, mutta on yli 20 prosentin päästövähennystä tavoiteltaessa tätä tehokkaampi.

Lisäksi tarkasteltiin säästä johtuvien satoon, ja siten myös ravinnetaseisiin vaikuttavien satunnaistekijöiden vaikutusta suhteelliseen kustannustehokkuuteen. Kasvukauden huono sää pienentää satomääriä ja nostaa siten ravinneylijäämiä. Säätekijöillä ei kuitenkaan ollut kovin suurta vaikutusta eri ohjauskeinojen suhteelliseen kustannustehokkuuteen. Huonon kasvukauden sattuessa tuoteveron kustannustehokkuus paranee hieman suhteessa päästöveroon, mutta se on silti alle 10 prosentin päästövähennystavoitteella päästöveroa huonompi. Hyvän kasvukauden sattuessa kaikkien ohjauskeinojen kustannustehokkuus paranee.

Kustannustehokkuus ei kuitenkaan ole ainoa kriteeri, jolla ympäristöpolitiikan ohjauskeinoja voidaan arvioida. Muiksi keskeisiksi kriteereiksi nousevat ohjauskeinon kyky saavuttaa sille asetettuja tavoitteita ajallisesti ja paikallisesti, ohjauskeinon hallinnollinen toteutettavuus, ohjauskeinon korrelaatio veden laadun suhteen sekä ohjauskeinon informaatioulottuvuus. Mikään tutkimuksessa tarkastellusta kolmesta ohjauskeinosta ei kaikkien näiden kriteerien suhteen nouse yksiselitteisesti muita paremmaksi. Näin ollen päätöksentekijän tulee punnita tarkkaan eri kriteerien painoarvoja ohjauskeinoa valittaessa.

Tutkimuksessa pyrittiin erityisesti painottamaan informaation merkitystä politiikkatoimenpiteen valinnassa. Maatalouden hajakuormituksen kontrolliin on esitetty viime aikoina monia lupaavia ohjauskeinoratkaisuja. Monessa tapauksessa informaation luonteella on keskeinen merkitys ohjauskeinon toimivuuden kannalta. Ohjauskeinon tulee ottaa huomioon se, että viljelijällä on yleensä tarkempi tieto tilansa ympäristövaikutuksista, kun taas säätelijällä on parempi tieto tarvittavista päästövähennyksistä. Mitä paremmin ohjauskeino välittää tietoa sekä viljelijälle että säätelijälle sitä tarkemmin viljelijä laskee tilakohtaisen ravinnetaseen ja sitä paremmin säätelijä saa mahdollisimman tarkan tiedon asetettujen politiikkatoimenpiteiden vaikuttavuudesta. Tässä suhteessa ravinnetaseeseen perustuva ohjauskeino on kaikkein informatiivisin.

Ravinnetaselaskelmat voidaan myös nähdä keinona yksinkertaistaa suoraan säätelyyn pohjaavaa hallinnollista ohjausperinnettä. Monimutkaisten viljelymenetelmiä koskevien säädösten (kuten ympäristötukijärjestelmä) asemesta maatalouden ympäristötavoitteiden toteutumista voitaisiin tarkastella ravinnetaselaskelmien avulla. Ravinnetaseita käytetään jo nykyään ympäristöpolitiikan työkaluna; esimerkiksi Hollannissa ollaan ottamassa käyttöön fosfaattiylijäämiin pohjaava tilakohtainen maksujärjestelmä. Hollannin järjestelmä on ymmärrettävä maan ravinneylijäämäongelman laajuuden näkökulmasta käsin. Suomeen tämäntyyppinen ratkaisu on liian voimakas, eikä se tämän tutkimuksen valossa näytä Suomeen sovellettuna olevan ratkaisuna kovin kustannustehokaskaan.

Alueellisten tai maakohtaisten ravinnetaseiden laskeminen tasaa jossain määrin tilatasolla tapahtuneet pellonkäyttöön liittyvät vaihtelut, ja siten voidaan tehdä arvioita ympäristöpoliittisten toimenpiteiden vaikuttavuudesta. Säätekijöistä johtuvat stokastiset ravinneylijäämien vaihtelut jäävät silti jäljelle. Näin ollen eri vuosien lukuarvot eivät ole suoraan verrannollisia keskenään, mutta ne antavat keskiarvotasolla kuvan kehityssuunnasta.

Ravinnetaseiden käyttö ympäristöpolitiikan yleistyökaluna on ongelmallista, sillä tilakohtaiset tekijät vaikuttavat ratkaisevalla tavalla ravinneylijäämien syntyyn. Tilan viljelykasvi, viljelykierto, maaperän ominaisuudet ja sääolot muuttavat vuosittain samankin tilan ravinneylijäämiä varsin paljon. Näin ollen eri vuosien ravinneylijäämien vertailu ei ole kovin mielekästä, ja vielä vähemmän mielekästä on vertailla eri tiloja saati sitten eri tuotantosuuntien tiloja keskenään. Parhaiten ravinnetaseet soveltuvat tilakohtaiseen ravinnevirtojen seurantaan yhdistettynä muuhun tuotannon suunnitteluun ja seurantaan. Tällöin taselaskelma tuottaa viljelijälle tietoa ravinteiden käyttöön liittyvistä ongelmakohdista. Ravinnetaselaskelmat toimivat näin ollen tiedollisen ohjauksen työvälineenä. Todettuaan ravinteiden käytössä ongelmakohtia on viljelijän sitten itsensä päätettävissä, mitä tilalla tulee tehdä päästöjen pienentämiseksi. Tämä on haastava lähestymistapa, ja se jättää paljon vastuuta tutkimukselle ja neuvonnalle välittää viljelijälle oikeaa tietoa tilalle soveltuvista ja käyttökelpoisista ratkaisuvaihtoehdoista.

Ravinnetaselaskelmat voivat välittää tärkeää tietoa sekä viljelijälle että hallinnolle maatalouden ympäristövaikutuksista. Tämän tiedon tarkkuus ja käyttökelpoisuus riippuu monista tekijöistä. Nämä tekijät tulee ottaa huomioon taselaskelmia tehtäessä ja käytettäessä laskelmista saatua tietoa tilan tuotannon suunnitteluun ja ympäristöpolitiikan työkaluna politiikkatoimenpiteiden vaikutusten arvioinnissa.

Annex 1. The production of nutrients by animal type and crop type in Finland.

Nutrient supply from organic manure (kg/year) by species in Finland.

Animal	Manure nutrient coefficients kg/year		
	N	P	P_2O_5
Milking cow	100	18	41
Sucker cow	55	9	21
Heifer	37	7	16
Calves (< 8 months)	25	3.5	8
Bovine animal >1 year	40	7	16
Sow with piglets	40	9	23
Pigs (by fattening place)	11	2.5	6
Horse	65	10	23
Pony	45	7	16
Sheep, goat	17	3.5	8
Laying hen	0.5	0.2	0.5
Broiler	0.3	0.05	0.11

Source: Own calculations.

Nutrient uptake by cultivated crops in Finland.

Plant	Nutrients in plant weight, kg/100 kg		
	N	P	P_2O_5
Winter wheat	20	3.4	7.8
Spring wheat	21	3.4	7.8
Rye	18	3.4	7.8
Barley	17	3.5	8.0
Oats	18	3.5	8.0
Mixed cereal	20	3.5	8.0
Pea	34	3.8	8.7
Potatoes	3.5	0.5	1.1
Sugar beet	2.0	0.35	0.8
Dry hay	19	2.4	5.5
Silage	5.5	0.6	1.3
Clover seed	15	2.2	5.0
Timothy seed	15	2.2	5.0
Turnip rape	37	8.6	19.7
Rape	37	9.0	20.6
Straw	6	1.0	2.3

Source: Tuori et al. 1996.

Annex 2.

Weitzman (1974) presents in his classic article that if benefit uncertainty and cost uncertainty are simultaneously present, and if costs are not independently distributed the comparative advantage of price instrument over a quantity instrument is given by:

$$(16) \qquad \Delta_{pq} \approx \sigma^2_C B''/2C''^2 + \sigma^2_C B''/2C'' ;$$

where Δ_{pq} = the net welfare advantage of the price instrument, relative to the quantity instrument;
B'' = the slope of marginal benefit function, the second derivative of the total benefit function, B;
C'' = the slope of the marginal cost function, the second derivative of the total cost function, C;
σ^2_C = the variance of costs;

and the ≈ sign is used to represent an accurate local approximation (Stavins 1996). Thus, the curvatures of abatement cost and benefit functions determine whether incentive or regulatory instruments are more robust when applied to local circumstances (Braden and Segerson 1993).

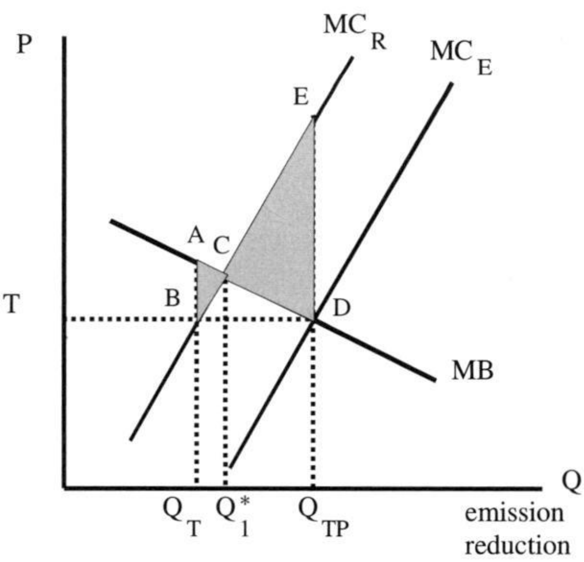

Figure 19. Cost uncertainty and the choice of policy instrument. Source: Stavins 1996.

Figure 19 shows a situation where the expected marginal costs (MC_E) of emission reduction differ from the realised marginal costs (MC_R). T is the price instrument, i.e. a tax, set to achieve the expected social optimum (D) in emission reduction, and Q_{TP} is the amount of tradable permits (quality instrument). However, as the real marginal costs are greater than what expected (i.e. MC shifts to left), the ex post optimum turns out to be at Q_1^*. In this example, it is clear that the social loss with the tax option at Q_T is much smaller (shaded triangle ABC) than with the tradable permit option at Q_{TP} (shaded triangle CDE). Thus, instrument superiority is a relationship between the expected and realised cost functions and the relative slopes of MC and MB, as formulated in equation *(16)*. Therefore, if the policymaker lacks the knowledge of the costs of emission reduction, the tax system is likely to lead to smaller welfare loss than setting standards, especially in case of steeply sloped marginal cost curves.

Annex 3. Variables and parameter values used in the silage production model.

Price levels are from 1994.

price of silage = p_q = 0.30 FIM/kg
price of nitrogen in fertilisers = w_f = 7.47 FIM/kg, [6.58,10.28] kg/ha
price of nitrogen in manure = w_m = 4.13 FIM/kg, [3.25,6.95] kg/ha
average leakage = Y = 130.0 kg/ha; average nitrogen surplus of the 17 farms
average fertilising for silage = A = 200 kg N/ha
yield function for silage Q =f(X); $b + 21.6X_{f,m} - 0.0341X_{f,m}^2$,

where b = an adjustment parameter from each farm (difference to 3164 from the Heikkilä's yield function), and

$X_{f,m}$ = the amount of fertiliser applied (fertiliser and manure)

leakage function for nitrogen; $L(N) = d + Ye^{0.7(Nf - 1)} + Y(4.6\, e^{0.7(Nm - 1)} - 1.8)$,

where d = an adjustment parameter from each farm (i.e. the difference of the farm specific nitrogen surplus to the average nitrogen surplus, Y)

N_f = X_f/A, the amount of fertiliser nitrogen used with respect to the average use of nitrogen

N_m = X_m/A, the amount of manure nitrogen used with respect to the average use of nitrogen

for fertiliser tax (t_1) the profit π is;

$$\pi(X_{f,m}) = p_q(b + 21.6X_{f,m} - 0.0341X_{f,m}^2) - (t_1 + 1)\, \Sigma w_{f,m} X_{f,m}$$

the effect of changing the level of fertilising is the derivative of the function above, i.e.;

$$d\pi / d X_{f,m} = p_q(21.6 - 0.0682X_{f,m}) - (t_1 + 1)\Sigma w_{f,m}$$

for levy on nitrogen surplus (t_2) the profit π is;

$$\pi(X_{f,m}) = p_q(b + 21.6X_{f,m} - 0.0341X_{f,m}^2) - \Sigma w_{f,m} X_{f,m} - t_2(d + Ye^{0.7(Nf - 1)} + Y(4.6e^{0.7(Nm - 1)} - 1.8))$$

thus the effect on profit when changing the fertiliser intensity is;

$$d\pi / dX_{f,m} = p_q(21.6 - 0.0682X_{f,m}) - \Sigma w_{f,m} - t_2(0.7N_f e^{0.7(Nf - 1)} + 4.6N_m e^{0.7(Nm - 1)})$$

for output tax (t_3) the profit π is;

$$\pi(X_{f,m}) = (p_q - t_3)(b + 21.6X_{f,m} - 0.0341X_{f,m}{}^2)$$

and the effect on profit when changing the fertiliser intensity is;

$$d\pi / dX_{f,m} = (p_q - t_3)(21.6 - 0.0682X_{f,m})$$

correspondingly the effect on leaching when changing the fertiliser intensity is;

$$dL / dX_{f,m} = 0.7N_f e^{0.7(Nf - 1)} + 4.6N_m e^{0.7(Nm - 1)}$$

weather probability function $W \approx N(\mu,\sigma^2)$,
average yield = 16,746 kg/ha, standard deviation for the sample = 1,993 kg/ha.

Annex 4. Some mathematical proofs.

Let us inspect the effect of price relationships in the case of output tax. For simplicity we can assume that the farm uses only one input, nitrogen, in silage production. Regardless of its origin, nitrogen is assumed to be the similar in quality and the price of it is w.

$Q = f(x)$ = quantity of silage production
t = tax on silage
p = price of silage
X^* = amount of nitrogen in optimal solution
w = price of nitrogen

the profit function can be written as;

$$\pi^* = (1-t)pf(X^*) - wX^*$$

differentiated with respect to X*, we get;

$$\delta\pi^*/\delta X^* = (1-t)p\delta f(X^*)/\delta X^* - w$$

setting this to zero and rearranging;

$$(1-t)p\delta f(X^*)/\delta X^* = w$$

and, in multiplying both sides with X^*/Q^* where $Q^* = f(X^*)$ we get;

$$(p\delta f(X^*)/\delta X^*)(X^*/Q^*)) = wX^*/(p(1-t)Q^*)$$

in here we note that the left hand side equals to the elasticity of production, *E*, which shows decreasing returns to scale if the right hand side is smaller than 1, i.e.;

$$\frac{wX^*}{p(1-t)Q^*} < 1$$

In general this holds true, as if we use the average parameter values for w, X*, p and Q, and set *t* to zero, the *E* is about 0.18. This means, that it does not pay off for farmer to use more fertilisers to get higher yield as the price relationships are so 'obscured'. It is only after the tax on silage price rises to about 0.82 when *E* turns to over one.

Similar inequalities can be presented for input tax and nitrogen surplus levy. For input tax, h, the formula is;

$$\frac{(h+1)wX^*}{pQ^*} < 1$$

In here, applying the average parameter values and setting h to zero, the E is again 0.18. In order for E to be over one h needs to be about 4.6.

Annex 5. Regression analyses.

Analysis from Section 6.2.

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.716[a]	.513	.507	16.1084

a. Predictors: (Constant), field area, livestock units

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	80.285	3.941		20.374	.000
	livestock units	3.391	.320	.865	10.612	.000
	field area	-2.710	.199	-1.108	-13.600	.000

a. Dependent Variable: cattle balance, kg/ha

Analysis from Section 6.4.2.

Dependent variable.. YIELD Method.. QUADRATIC

Listwise Deletion of Missing Data

Multiple R	.48656
R Square	.23675
Adjusted R Square	.12771
Standard Error	992.51693

Analysis of Variance:

	DF	Sum of Squares	Mean Square
Regression	2	4277751.1	2138875.5
Residuals	14	13791257.9	985089.8
F = 2.17125		Signif F = .1509	

---------------------- Variables in the Equation -----------------

Variable	B	SE	B	BetaT	Sig T
NITROGEN	22.010193	26.849767	.899846	.820	.4261
NITROGEN**2	-.037640	.095661	-.431918	-.393	.6999
(Constant)	1727.257378	1842.527007		.937	.3644

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