Economic impacts of transferable quotas in pesticide regulation

data, model and scenarios

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Economic impacts of transferable quotas in pesticide regulation

- data, model and scenarios

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Copenhagen 2002

Abstract

The report describes the design and data base of a novel model concept for performing integrated economic-environmental quantitative assessments of agri-environmental policy instruments. The core of the concept is an econometrically estimated model of the Danish agricultural sector (ESMERALDA) based on economic and structural data from farm accounts and supplemented with farm specific data on nitrogen and pesticide use estimated from survey and recorded data.

The model concept is used to analyse the effects of an aggregate reduction in the use of herbicides in line with the target level of the Pesticide Action Plan by either uniform (farm-level) reductions or by using a system of individually transferable reduction requirements. Regardless of the mode of regulation, herbicide reductions of 62 per cent are followed by slight increases in the use of other pesticides and imply reductions of 39 per cent in total pesticide use. However, the geographical distribution of pesticide use is more unevenly distributed when using transferable reduction requirements, implying a 'polarisation' effect that leads to pesticide use being concentrated in some areas. Herbicide reduction implies increases in the nitrogen surplus, most significantly in the case of a uniform reduction. Even though the acreage of cereals increases and the acreage of seed and root crops decreases, the aggregate land use is not affected significantly by the regulations, and only 6 per cent of the reduction in pesticide use is attributable to shifts in land use.

Considerable efficiency gains – some 600 million DKK – in the form of reduced costs of compliance can be achieved by the introduction of transferable reduction requirements (total costs 1.4 billion DKK) as opposed to a uniform regulation (total costs 2 billion DKK). However, as the implementation of transferable reduction requirements yields an uneven geographical distribution of herbicide use compared to the case of regulating individual farms, the results suggest that the setting of national environmental targets should be accompanied by spatial distribution objectives.

Contents

Preface

The objective of this report is to analyse the economic and environmental effects of introducing transferability into Danish pesticide regulation. Furthermore, an objective of this report is to describe and demonstrate the application of a novel model concept that facilitates integrated economic-environmental quantitative assessments of agri-environmental policy instruments targeting e.g. the use of nitrogen and pesticides, including regional and distributional effects within the agricultural sector.

The report is a part of the activities in the AMOR 4 programme under the auspices of the AMOR research centre, financed by the Strategic Environmental Research Programme II. The overall objective of the research centre is to conduct integrated social and scientific environmental research focusing on developing and testing models for analysing the concept of sustainability.

This report was prepared and edited by senior researcher Jørgen Dejgaard Jensen, researcher Hild Rygnestad and research assistants Martin Andersen, Henrik Huusom and Stine H. Jørgensen along with senior researcher Tove Christensen and research director Søren E. Frandsen from the Agricultural Policy Research Division of FOI.

Danish Research Institute of Food Economics (FOI), December 2002

Ole P. Kristensen

1. Introduction

1.1. Background

In Denmark as well as in many other countries it is considered socially desirable to reduce nutrient and pesticide pollution from agriculture. Throughout recent years, a number of analyses – qualitative as well as quantitative – have been directed towards various aspects of environmental impacts from agriculture and the economic impacts of regulation on agriculture at the farm and/or sector level. The applied methodologies vary substantially from farm level mathematical programming models based on field experiment data to econometric farm sector models and more general economic (applied general equilibrium) models. These economic models often have more or less closely linked environmental satellite models. Examples of such research related to nitrogen problems are Walter-Jørgensen (1998), Vetter et al. (1995), Schou et al. (1996, 2000), Becker & Kleinhanss (1995), Linddal (1998), Vatn et al. (1996) and Jacobsen et al (1997), whereas Ørum (1999), Jacobsen (1999) and Jacobsen et al. (1999) address the issue of pesticides.

In general, the above studies address either the nitrogen or the pesticide issues as isolated problems. However, it is important to analyse interactions between different environmental problems. For instance, regulation of nitrogen use influences the use of other fertilisers as well as pesticides, and restrictions on the use of pesticides have additional effects on the use of fertilisers. In order to make environmental regulation more efficient, such interactions should be taken into account.

One general problem related to quantitative analyses of these issues is a lack of sufficiently detailed data for describing the interaction between agricultural economic behaviour and the use of nutrients or pesticides. Often, data representing the overall economic behaviour on farms are relatively aggregated with respect to nutrients and pesticides. In other cases, data focus on nutrients and/or pesticides at a detailed level, but neglect the aspect of overall economic optimisation.

Another general problem is the lack of behavioural parameter estimates describing the interactions between the use of different nutrients or pesticides and other inputs and outputs. Again, parameter estimates either address the issues at a too aggregated level, or they have a partial economic interpretation ignoring the interactions with other agricultural inputs.

The optimal design and implementation of environmental regulation in agriculture has been focused upon in the last decade, along with a discussion of the relative merits of market-based instruments and traditional regulatory measures. The use of economic instruments in environmental policy making is often argued to expand the scope for efficiency gains. Standard textbook analysis imply that in the presence of agents with different cost structures, the introduction of a system of transferable pollution quotas will increase the cost-efficiency of a reduction policy – i.e. achieve given environmental objectives at lower costs – compared to direct control measures, cf. section 4.1. Still, pesticide regulation in Denmark has concentrated mainly on implementing direct controls and information measures to reduce the environmental and public health risks associated with pesticide use.

1.2. Objectives

The objective of this report is to quantify the potential efficiency gains that arise when a flexible market based incentive policy in the form of transferable quotas is implemented in pesticide regulation. Furthermore, the report aims to quantify a selection of environmental effects and the cross-effects between nitrogen and pesticide policies.

Specifically, the environmental and economic effects of two scenarios differing in the mode of pesticide regulation are analysed. Both scenarios achieve the same reduction in herbicide use, but whereas the reduction of the first scenario is implemented by farm-level reductions, the reduction in the second scenario is achieved by using a system of individually transferable reduction requirements.

More generally, the objective of this report is to improve the knowledge base and hence the possibilities for integrated economic-environmental quantitative assessments of agri-environmental policy instruments, focusing on nutrients and pesticides in agriculture. This involves improving the quantitative description of nutrient and pesticide use in Danish agriculture. Such descriptions include disaggregated information about use of specific nutrients and pesticides on different farm types, as well as behavioural parameters for the response in the use of these inputs to changes, e.g. in price relations.

1.3. Methodological overview

The overall approach in this report is to bring data from different sources together on a common platform, and attach economic behavioural parameters to this data. Hence, the task has been to select a common data platform, to collect supplementary data, to determine behavioural parameters and finally to make these supplementary data and behavioural parameters consistent as illustrated in the middle section of figure 1.1.

Figure 1.1. Methodological framework

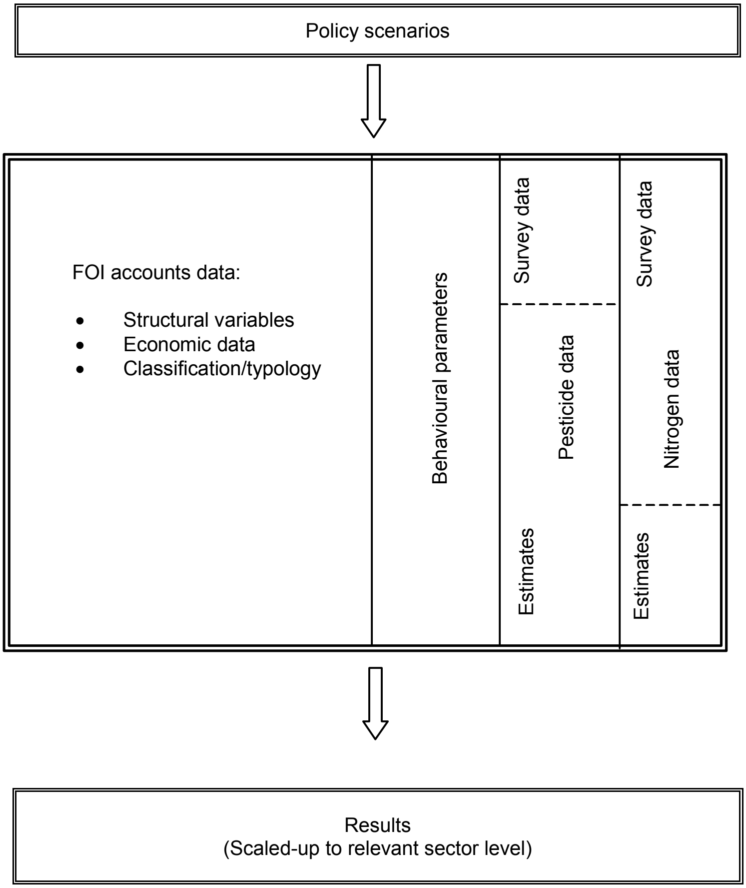

Figure 1.1 illustrates the methodological framework with which policy scenarios can be analysed and related to the Danish agricultural sector. Using the Agricultural Accounts Statistics database of the Danish Research Institute of Food Economics (FOI) – *accounts database* or *FOI database* – behavioural parameters and supplementary data on pesticide and nitrogen use as a basis for the econometric agricultural sector

model ESMERALDA, changes in agricultural behaviour (production, input use, incomes etc.) can be simulated, e.g. in the case of price changes. In the supplemented account statistics, as well as in the sector model, the results can be scaled-up to the relevant sector level, i.e. national, regional, local or farm types[1].

The Agricultural Accounts Statistics database of FOI has been selected as the platform for the methodological framework. The database consists of agricultural accounts information from 1500-2000 farms per year, aiming at a representative description of the Danish agricultural sector. Such information includes data on land use, livestock, output values specified by products, input costs divided into a number of main components, income, use of labour and capital. The database underlies the annual publication Agricultural Accounts Statistics, (see for example SJFI[2], 2000).

One limitation of the accounts database is that the use of fertilisers and pesticides is described on an aggregated basis. The use of fertilisers is only described in terms of the total cost of commercial fertilisers, whereas the use of pesticides is described only in terms of the total pesticide cost at the farm level. Hence, the accounts data may be too aggregated for conducting environmentally related analyses. By merging databases, supplementary data have been established for all farms in the FOI accounts database, here referred to as the *FOI farms*. As indicated in figure 1.1 the supplementary data on nitrogen and pesticide use are collected for a subset of the FOI farms, referred to as the *survey farms*.

Data on nitrogen use is available for the 1997/98 harvest year from the Danish fertiliser accounts database. It is possible to identify 1382 survey farms whose information is used to estimate nitrogen use data for the remaining FOI farms. The estimation procedure is described in chapter 2. Furthermore, an environmental indicator of potential nitrogen loss is developed, estimating nitrogen surplus using field level balance sheets.

At an earlier stage of the AMOR-project, a questionnaire was sent to all FOI farms addressing in details the farms' use of pesticides in the harvest year 1996/97 (Schou, 1998). Of these a total of 448 survey farms are used to estimate pesticide use for the remaining FOI farms. As is the case with nitrogen data, the estimation procedure is described in chapter 2.

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[1] To aid non-Danish readers, a regional map of Denmark is shown in appendix 4.

[2] Prior to January 2002, The Danish Research Institute of Food Economics (FOI) was titled The Danish Institute of Agricultural and Fisheries Economics (SJFI).

Finally, behavioural parameters, i.e. price elasticities, are estimated econometrically for eight different farm types (part-time farms and full-time crop, cattle and pig farms on sandy and loamy soil). Specifically, these include disaggregated price elasticities for individual fertiliser (nitrogen, phosphorus and potassium) and pesticide components (herbicides, fungicides, insecticides and growth regulators). Each of the FOI farms are classified into one of the eight categories and the corresponding set of elasticities is attached to the respective farm. The behavioural model is described in more detail in chapter 3.

1.4. Outline of the report

Chapter 2 describes the construction of supplementary data on nitrogen and pesticide use for the analytical framework, whereas the estimation of behavioural parameters related to the agricultural use of fertilisers and pesticides is described in chapter 3. The framework is applied in chapter 4 to two scenarios of herbicide reduction (in line with the Pesticide Action Plan's goal); in scenario 1 a cap on individual farms' herbicide use is enforced, and in scenario 2 the same aggregate reduction is achieved by a system of individually transferable reduction requirements. Finally in chapter 5, the main findings and conclusions of the report are summarised and the relevance and future perspectives of the model approach is discussed.

2. Data

To enable economic modelling as well as compilation of national statistics of environmental parameters, the FOI accounts database is supplemented with farm level quantities of selected input factor use. The supplemented data contain farm level information on the use of two input factors:

- Nitrogen input data consists of quantities of applied nitrogen from commercial fertilisers and organic nitrogen other than manure (e.g. compost). The data are extracted from the national database of farms' fertiliser accounts (Landbrugets EDB Center, 1999).
- Pesticide input data consists of application rates of active ingredients[3] and standard application doses[4] of the four groups of pesticides: herbicides, fungicides, insecticides and growth regulators. Data on individual farms' pesticide use are extracted from a survey by Schou (1998).

Figure 2.1 illustrates how data on these input factors are generalised in two steps. Firstly, input data are available only for a subset of the FOI farms – the survey farms. These are used to estimate input factor use on the remaining FOI farms by an approximation procedure, cf. section 2.1. Secondly, national statistics are compiled using standard FOI scaling-up methods in which each FOI farm has an allocated weight indicating how many farms it represents on a national level [5]. See SJFI (2000) for a description of how the accounts database is used for compiling national statistics. In Huusom (2001), the approximation procedure yields national results for pesticide use that range from 15 per cent below to 7 per cent above the national sales figures depending on the choice of indicator.

This chapter first describes the general features of the approximation procedure, followed by two sections dealing with nitrogen and pesticides, respectively.

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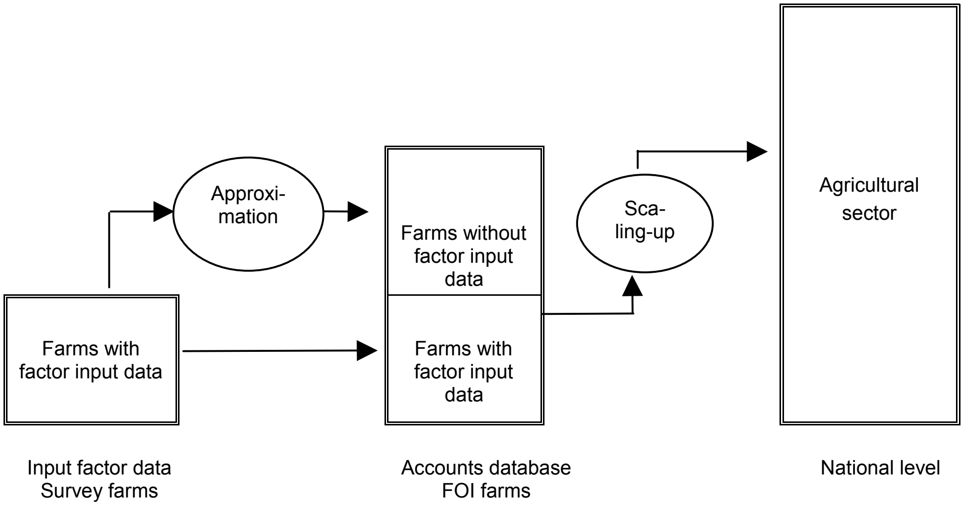
3 Active ingredients (AI) refers to the amount of biologically active compounds in the formulated product used.

4 Standard treatment dose (SD) refers to the recommended quantity of the formulated product to be used in one treatment of one hectare of a given crop. The application of a total of one SD to one hectare equals a treatment frequency index (TFI) of 1.

5 The ESE (economic size unit) weights are used commonly to scale-up the FOI data to match national statistics on the value of agricultural production. Other weightings systems that may be applied include area weights to match the agricultural area or regional weights to match regional statistics.

Figure 2.1. The principles of generalising limited farm level input factor data



2.1. Approximation

Data on input factor use are determined for all FOI farms by using the approximation procedure outlined in box 2.1. Input use on some farms is determined directly from survey data, while the others are estimated as a weighted average of input use on a selection of survey farms. By applying this procedure, it is assumed that farms, which are similar with respect to selected structural variables, behave similarly with respect to input use.

Box 2.1. Main steps in the approximation procedure

1. Establish structural data for all FOI farms from the accounts database.
2. Establish structural and factor input use data for survey farms by merging databases.
3. Using 2, select structural variables and determine their variable-weights.
4. Using 1, select 10 survey farms and their farm-weights for each remaining FOI farm.
5. Adjust the farm-weights to match real and estimated farm area.
6. Use the adjusted farm-weights to estimate input factor use on each FOI farm.

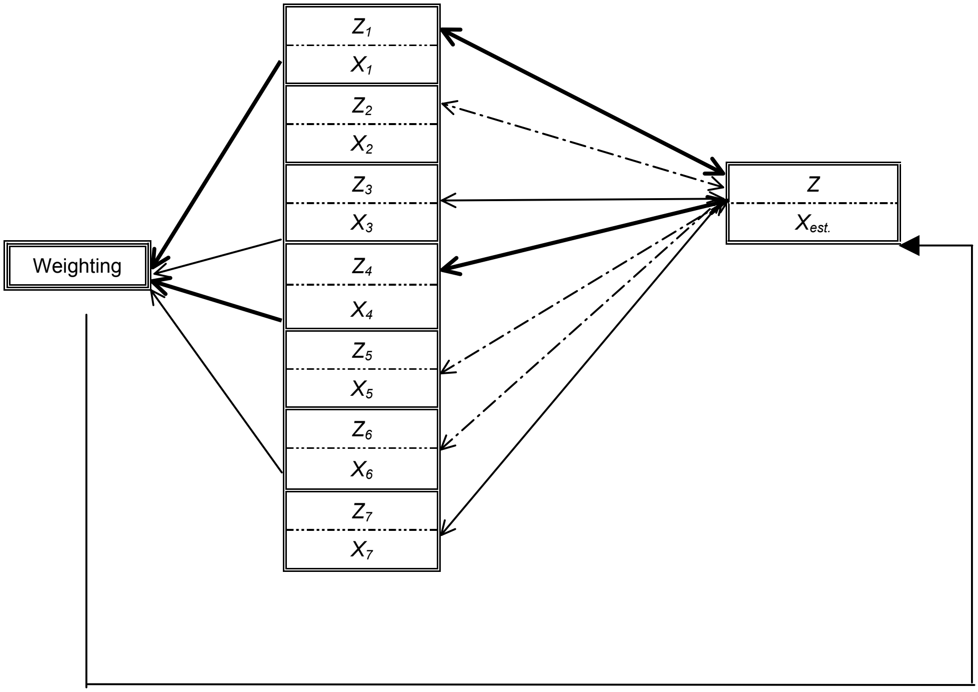
Structural data are accessible directly from the FOI database (step 1), while as discussed in chapters 2.2 and 2.3, merging these with input data requires more resources (step 2).

The approximation procedure centres on comparing the structural variables of survey farms with those of other FOI farms. These variables are for example: size of farm area; number of livestock; and production value. The variables are chosen based on their correlation with input use data (step 3). As some structural variables basically express the same feature, for example related to farm size, it is important to exclude similar variables to achieve a more broadly based approximation. The correlation coefficients are also used to determine relative weights between the selected structural variables. Thus, these variable-weights reflect the relative importance of each structural variable in describing input factor use. Note that, to reduce the computational time of running the approximation procedure, it is necessary to limit the number of chosen variables.

In step 4 an aggregate measure of similarity is used to find the ten most applicable survey farms to form a weighted average for each FOI farm. The chosen ten survey farms represent the smallest sum of squared differences with respect to selected structural variables. In addition, these ten farms are allocated a farm-weight such that a farm with low sum of squared differences get a high weight and vice versa. This is illustrated in figure 2.2 where one farm's input use is estimated as a weighted average of four survey farms. In the approximation procedure a number of ten farms is chosen to obtain robust estimates of input use. The choice of number of farms is subject to sensitivity analyses. Note that the variable-weights indicate the importance of different structural variables, while farm-weights indicate the importance of different farms.

Because the use of commercial nitrogen and pesticides relate to each farm's agricultural area, it is viewed as particularly useful to obtain an accurate approximation of total farm area. As such, the farm-weights are adjusted so that the estimated area on each farm matches that observed in the FOI data (step 5). Concluding the approximation procedure, it is now possible to estimate input use for each FOI farm using survey farm data (step 6). See appendix 1 for further details and Huusom (2001) and Rygnestad et al. (2000) for applications of the approximation procedure.

Figure 2.2. Estimation of input data for one FOI farm as a weighted average of four farms' input data



Survey farms FOI farm

Notes: 1. Z - vector of structural variables; X - vector of input factors.
2. Bold lines indicate a high degree of similarity (i.e. a small sum of squared differences) and thus a high farm weight (survey farms 1 and 4).
3. Fully drawn lines indicate a lower degree of similarity (i.e. a larger sum of squared differences) and thus a low farm weight (survey farms 3 and 7).
4. Dotted lines indicate a very low degree of similarity (i.e. a very large sum of squared differences) and such farms are excluded from the weighted average (survey farms 2,5 and 6).

2.2. Nitrogen

Supplementing the FOI accounts data with data on nitrogen use has a two-fold aim: Firstly, it enables existing modelling tools to incorporate quantities of nitrogen more explicitly than previously. Thereby it is possible to predict how changes in the economy (prices etc.) affect agricultural production as well as input factor use. This can be seen as a first step towards modelling the environmental effects of varying agricultural policies. Secondly, the extra data make it possible to construct soil surface balances and surplus for nitrogen use at the farm level. This is a step further towards analysing environmental effects as it is used as an environmental indicator for potential nitrate loss from each farm (Rygnestad & Schou, 1999).

The approximation procedure

In order to achieve this, it is necessary to merge the FOI data with the fertiliser account data. The fertiliser account data are extracted for the 1997/98 harvest year and contain information for 43,806 farms particularly on nitrogen contents in their applied commercial fertiliser and in other organic fertiliser than manure, but also on farm area, livestock numbers and trade in manure [6]. Manure use is estimated from FOI data on livestock numbers together with standard production coefficients (Plantedirektoratet, 1997). Merging is based on both CPR and SE numbers of the farm owners [7].

Fertiliser accounts were identified for 82 per cent of the 1956 FOI farms in the corresponding financial year. Upon closer inspection the differences in records of livestock numbers in the two databases are so large on 21 of these farms, that the estimated production of animal manure is smaller than the observed sale of manure. This is taken to imply a problem of farm unit definition, and these farms are added to the group with no nitrogen fertiliser data. Thus to estimate nitrogen use on other FOI farms, 1382 survey farms are used in the approximation procedure. Note that nitrogen data are not estimated for FOI farms that have no land. In addition, it is assumed that organic farms do not apply commercial nitrogen fertilisers.

In some cases, because data sources are collated at different times and because their farm unit identification varies, comparable data from both sources do not match perfectly and need adjustment. Farm area is chosen as the adjustment factor because the ultimate aim is to construct soil surface nitrogen balances for each farm. For differences in agricultural area, the nitrogen data was scaled to match the FOI data. As such the nitrogen application intensity, kg N/ha, is used rather than the total nitrogen applied.

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[6] Other information is also contained such as nitrogen content of applied animal manure. However, the quality of this data is commonly regarded as less precise. Because of few observations and as this study is based on only one year of data from the fertiliser accounts, estimates are not obtained for manure trading on all farms.

[7] CPR and SE numbers are registered personal and business identification numbers depending on which one has been supplied. Starting from the 1998/99 accounting year, all farms are identified with a CVR number (registered business numbers).

The nitrogen data for 1382 survey farms form the basis for the approximation procedure. To select which structural variables should be used in the procedure, correlation coefficients are calculated between nitrogen use and several structural variables [8]. To reduce the computational time, only the six variables with the highest correlation coefficients are selected. Their relative weights are also based on the correlation coefficients [9]. The results lead to the use of the following variables according to decreasing weights: total farm area; standard gross margin from crop production, gross output in DKK; livestock units; share of area under crops with fertiliser rates between 100 and 150 kg N/ha; and share of area under crops with fertiliser rates between 150 and 200 kg N/ha.

Finally, soil surface balances are constructed according to the method in Rygnestad & Schou (1999). By deducting the amount of nitrogen removed from the soil from the applied amount, the remaining nitrogen surplus is an indicator of *potential* nitrogen loss. It is seen as a measure of potential loss because the surplus contains not only losses to soil and air but also changes in the pool of nitrogen in the soil.

To establish consistent data material for the analysis it is necessary to estimate nitrogen data for the 1996/97 harvest year - corresponding to available pesticide data. As such it is assumed that farmer behaviour with regard to nitrogen use is comparable between the 1996/97 and 1997/98 harvest years. Thus, to estimate nitrogen data on 1956 FOI farms 1382 survey farms are used, from the two years respectively. The following results are thus shown for the 1996/97 harvest year. See Appendix 2 for numerical results when the approximation procedure is run for the initial year, 1997/98.

Results

Table 2.1 shows some of the data obtained in the current work. In particular it includes the different components used to construct soil surface balances for each FOI farm. With each FOI farm having an estimated nitrogen balance sheet, the basis situa-

[8] To avoid scaling problems, the structural data were normalised to lie between 0 and 1.

[9] The initial variable-weights are adjusted in a procedure whereby the input use of each survey farm is estimated as a weighted sum of that on ten other survey farms. The adjustment minimise the sum of squared differences between the observed and estimated total input for all survey farms. This entails using the approximation procedure, whereas future development could include obtaining maximum likelihood estimators for the weights. It could also be considered to use squared correlation coefficients as the basis for the determination of the weights. However, for the sake of robustness in the aggregation, (straight?) correlation coefficients have been applied here.

tion is established for scenario analyses of potential nitrogen loss. The last column of the table shows how scaled-up results can indicate the basis situation at a national level.

Looking at average values for three different farm types, the results show a higher use of commercial nitrogen per hectare on crop farms, while total nitrogen use is highest on cattle farms, due to their large areas of nitrogen intensive fodder crops and the use of manure. Because of the relatively high level of removed nitrogen with crops on cattle farms, estimated nitrogen surplus per hectare is lower than that on pig farms.

Table 2.1. Estimated soil surface nitrogen balance for the agricultural sector, 1996/97 harvest year

	Crop farms kg N/ha	Cattle farms kg N/ha	Pig farms kg N/ha	National [3] 1000 tons N
Nitrogen input through:	171	286	258	611
Commercial nitrogen	123	104	94	290
Animal manure	11	154	128	230
Organic nitrogen other than manure	3	1	2	5
Seed	3	3	3	7
Biological fixation	15	8	14	33
Atmospheric deposition	15	15	15	40
Asymbiotic fixation (micro organisms)	2	2	2	5
Removed nitrogen by:	99	149	103	309
Cash crops [4]	90	37	95	195
Fodder crops	9	112	8	114
Nitrogen surplus: Input – Removed	72	137	155	302

Notes: 1. Total agricultural area = 2.7 mill. ha;
2. Excludes trade in manure between farms;
3. Results are scaled-up using the FOI ESE-weights (economic size unit weights);
4. Cash crops include: cereals, pulses, rape, root- and non-food crops for sale. Own production of cereals for fodder is also included here.

Because the modelling section later is centred on eight farm types, their average nitrogen surplus and areas are shown in table 2.2. The results show that full-time pig farms on sandy soil have the highest surplus per hectare (167 kg N/ha), followed by cattle, crop and part-time farms. In addition, nitrogen surplus per hectare is generally higher on sandy soils (see also Dalgaard, 1998).

Table 2.2. Estimated nitrogen surplus (soil surface) on eight farm types 1996/97 harvest year

	Nitrogen surplus 1000 tons N	Area 1000 ha	Nitrogen surplus kg N/ha
Crop farms, loamy soil	21	319	65
Crop farms, sandy soil	22	273	81
Cattle farms, loamy soil	19	124	156
Cattle farms, sandy soil	95	661	144
Pig farms, loamy soil	30	220	138
Pig farms, sandy soil	63	378	167
Part-time farms, loamy soil	15	241	64
Part-time farms, sandy soil	35	436	80
Total	302	2652	114

Note: Excludes trade in manure between farms.

Evaluating the results

The approximation procedure used to estimate nitrogen use can be evaluated in several ways. By definition it is not possible to evaluate the ability to approximate nitrogen use directly, so figure 2.3 shows estimated versus observed commercial fertiliser costs for the FOI farms[10]. The approximation is considered reasonable as most farms lie on or near the 45°-line in the graph. However, there appears to be a tendency for underestimation on farms with larger costs.

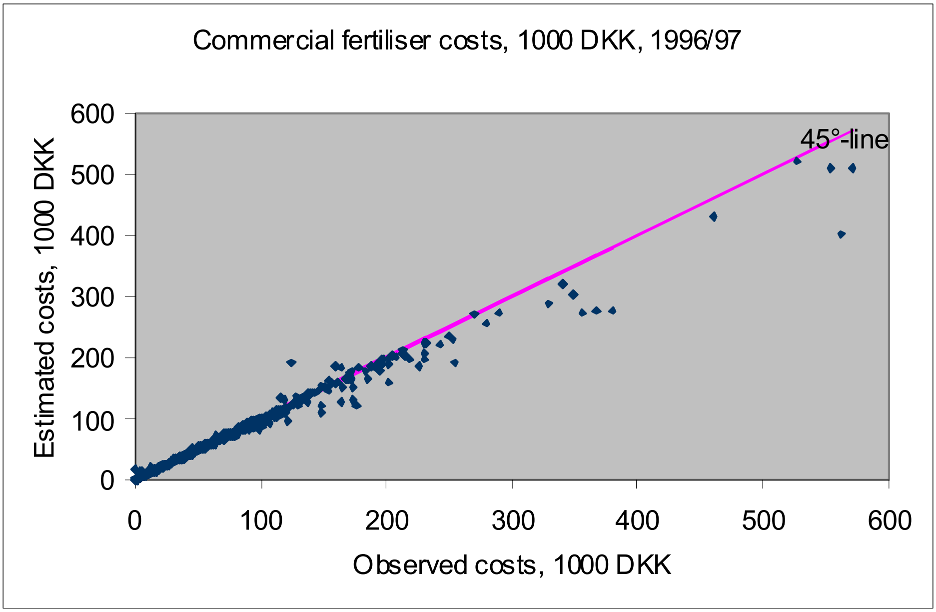
A second way of evaluating the approximation is to compare the scaled-up estimate for total use of nitrogen from commercial fertilisers in Denmark (291 thousand tonnes) with official statistics. In this case the estimate, using ESE weights for each FOI farm multiplied by their nitrogen use, seems reasonable as it is found to be only 1 pct. higher than the national sales statistics of 288 thousand tonnes (Danmarks Statistik, 1999).

Compared to farm level analyses by Landbrugets Rådgivningscenter (1998) the current field level nitrogen balances lie in the lower range of Danish study farms. This is probably because of differences in the farm structures between the two studies. The estimated total nitrogen content in animal manure is 230 thousand tonnes, which is only 1 pct. lower than that estimated in Kyllingsbæk et al. (2000). Compared to their results, the current estimate of the total nitrogen surplus (302 thousand tonnes) is 7 pct. lower, probably due to a 10 pct. overestimate of nitrogen removed with crops. In

[10] Commercial fertiliser costs not only include nitrogen use, but also phosphorous and potassium use.

total, the approximated data of applied commercial nitrogen appears to be within the range of other studies, while the final nitrogen surplus indicator might be slightly underestimated.

Figure 2.3. Estimated versus observed commercial fertiliser costs, 1996/97 harvest year



2.3. Pesticides

Supplementing the FOI accounts data with data on pesticide use enables existing modelling tools to incorporate groups and quantities of pesticides more explicitly and in more detail than previously. Specifically, data from the pesticide survey database and the FOI accounts database are combined to estimate individual farms' crop-specific pesticide application rates, making it possible to model changes in pesticide use resulting from changes in crop composition. Also, by scaling up data from the supplemented FOI accounts database to the national level, it is possible to obtain an estimate of the aggregated pesticide use in Denmark as well as a detailed description of the structure of pesticide use in the entire agricultural sector.

The approximation procedure

Pesticide data are extracted from a survey conducted by Schou (1998) based on some 600 completed questionnaires specifying details on treated areas and applied doses for each pesticide treatment for the harvest year 1996/1997 on participating farms. For each farm, the quantity of pesticides used is computed from survey data and total pesticide use is expressed in active ingredients and standard treatment doses specified by pesticide group: herbicides (H); fungicides (F); insecticides (I) and growth regulators (V). Pesticide data are then linked to structural data for 448 of the 1956 FOI farms on the basis of identical FOI identification numbers [11]. The survey data are used to estimate farm level pesticide use by an approximation procedure for the remaining 1508 FOI farms for which no pesticide data are available. Note that the 35 organic farms and the 21 farms with no cultivated area in the FOI accounts database 1997/98 are excluded from the approximation procedure as they are assumed not to use pesticides.

Data from the survey farms are first used to select the structural variables that the approximation procedure is based on. For reasons of computational ease, only seven structural variables are selected. The selection is based on a high degree of correlation between the structural variables and aggregate pesticide use of individual farms. The correlation coefficients lead to the selection of the following structural variables: pesticide costs; total farm area; standard gross margin from crops; proportion of loamy soil to total farm area; wheat area; seed area; pulses (only used for approximating AI) and sugar beets (only used for approximating SD). Additionally, the correlation coefficients are also used as a basis for determining variable-weights reflecting the relative significance of different variables in describing aggregate pesticide use [12].

After determining the variable-weights, the approximation procedure is executed to produce estimates of pesticide use of the 1508 FOI farms. For each of these farms, pesticide use is estimated as a weighted average of the pesticide use of ten farms from the pesticide survey. In some cases, the estimates do not reflect the observed areas very well, although aggregate pesticide use is highly correlated with the cultivated

[11] The Agricultural Accounts Statistics 1997/98 is the relevant year for comparing structural data to pesticide data for the harvest year 1996/97.

[12] The initial variable-weights are adjusted in a procedure whereby the input use of each survey farm is estimated as a weighted sum of that on ten other survey farms. The adjustment minimise the sum of squared differences between the observed and estimated total input for all survey farms. This entails using the approximation procedure, whereas future development could include obtaining maximum likelihood estimators for the weights.

area of individual farms. To ensure this, farm-weights are adjusted to reflect the observed agricultural areas of the farms.

The approximation procedure is executed both for active ingredients and for standard treatment doses. By combining observed data from survey farms with estimated data for 1508 farms, all FOI farms have data on total amount of active ingredients as well as on the total number of standard applications doses used. In addition, pesticide use is specified by pesticide group.

As previously stated, there is a modelling requirement to determine crop-specific pesticide application rates. However, because structural data on different crop areas diverge between FOI and survey data for the same farm, some assumptions are made. The FOI data on areas under different crops are taken to be the base, whereas crop-specific pesticide application rates are based on survey data. When a farm's application rates for a given crop is not reported, the average for that crop across all survey farms is applied. In this way, crop-specific pesticide application rates for all (and not only observed) crops are estimated for all survey farms. Next, crop-specific pesticide application rates for the remainder of the FOI farms are calculated as weighted averages of the corresponding application rates of ten survey farms as described above. Finally, aggregate pesticide use of the individual FOI farms is calculated by multiplying for each farm the crop-specific pesticide application rate by the area of the individual crop and by summation over all crops. In order to produce national estimates of pesticide use, farm-specific aggregate pesticide use is scaled up using the ESE weighting procedure described previously.

Results

Three types of results are derived from the approximation procedure. Firstly, estimates of crop-specific application rates are produced for each FOI farm. Secondly, by multiplying the crop-specific application rates with FOI data on individual farms' crop areas, estimates of aggregate pesticide use on all farms in the FOI accounts database are obtained. For each farm, these estimates are given for the aggregate pesticide use broken up by pesticide groups and two indicators: active ingredients and standard treatment doses. In the results, no distinction is made between observed and estimated pesticide data. Thirdly, farm specific results are then scaled up using standard FOI scaling-up factors to produce results on a national level that form the basis for a description of pesticide use in the Danish agricultural sector.

The first type of result is a list of crop-specific pesticide application rates for each farm. This makes it possible to model changes in pesticide use as a function of varying crop areas. Table 2.3 displays examples for three farms. For illustrative purposes only data for winter barley are shown.

Table 2.3. Examples of data listing crop-specific pesticide application rates distributed between groups

Crop	Winter barley							
	AI (kg/ha)				SD (number/ha)			
Identification	H_{AI}	F_{AI}	I_{AI}	V_{AI}	H_{SD}	F_{SD}	I_{SD}	V_{SD}
Farm 1	1.072	0.223	0.001	0.023	1.420	0.645	0.007	0.046
Farm 2	0.955	0.180	0.000	0.012	0.921	0.412	0.003	0.024
…	…	…	…	…	…	…	…	…
Farm 1956	2.106	0.287	0.000	0.010	5.860	2.185	0.005	0.035

Notes: herbicides (H); fungicides (F); insecticides (I) and growth regulators (V). AI and SD denote active ingredients and standard treatment doses, respectively.

The list of crop-specific application rates thus comprises 1956 farms and for each farm application rates are listed for the 17 different crops. For each crop, application rates are listed as shown in table 2.3.

Below is a brief presentation of the estimates of the aggregate pesticide use in Denmark and the average pesticide application rates of individual farm types.

Total pesticide use in Danish agriculture is estimated to be 3.1 thousand tonnes active ingredients in the harvest year 1996/97, corresponding to an average pesticide application rate of 1.2 kg active ingredients per hectare of agricultural land. These figures are 15 per cent lower than the national estimate of pesticide use based on annual sales statistics published by the Danish EPA (Miljøstyrelsen, 1998) adjusted for differences in the accounting period. These differences can partly be ascribed to stock changes and general wastage. Herbicides constitute by far the major proportion of the pesticide use (74%), followed by fungicides (19%), growth regulators (5%) and insecticides (1%).

Using the number of standard treatment doses as a measure of total pesticide use, an estimated 5.7 million standard treatment doses are applied in the harvest year 1996/97. This figure is 5 per cent higher than the EPA estimates, and the difference is attributed to the use of different assumptions in the different calculations. The number of standard treatment doses per hectare is 2.15 when considering the total agricultural area. To calculate the Treatment Frequency Index (TFI), only the area in rotation is considered and the TFI is calculated by dividing the total number of standard treatment doses by the agricultural area minus permanent grassland and set-aside. The TFI is calculated to be almost 2.5 for 1996/97, 7 per cent higher than the EPA estimate for reasons described above. Again, herbicides (65%) and fungicides (21%) constitute the major proportions, whereas insecticides and growth regulators now constitute 10 and 3 per cent of total pesticide use, respectively. For both indicators, the estimated distribution by pesticide groups corresponds very well with the national statistics based on pesticide sales.

In tables 2.4 and 2.5, pesticide application rates are shown for eight farm types as measured by kg active ingredients per hectare and TFI, respectively.

Table 2.4. Pesticide use in terms of active ingredients and application rates distributed by farm types, 1996/97.

Farm types	AI application rate (kg/ha)				
	H	F	I	V	All groups
Crop farms, loamy soil	1.41	0.26	0.04	0.11	1.83
Crop farms, sandy soil	0.99	0.60	0.02	0.09	1.70
Cattle farms, loamy soil	0.89	0.10	0.02	0.04	1.05
Cattle farms, sandy soil	0.81	0.12	0.02	0.05	1.00
Pig farms, loamy soil	0.99	0.23	0.02	0.07	1.31
Pig farms, sandy soil	0.81	0.26	0.01	0.05	1.14
Part-time farms, loamy soil	0.74	0.16	0.01	0.03	0.94
Part-time farms, sandy soil	0.55	0.16	0.01	0.04	0.75
All farms	**0.87**	**0.23**	**0.02**	**0.06**	**1.17**
Distribution by pesticide groups	74%	19%	1%	5%	100%

Notes: herbicides (H); fungicides (F); insecticides (I) and growth regulators (V).
AI denotes active ingredients.
Due to rounding, columns may not add up to totals.
Calculations are based on an agricultural area of 2.7 million hectares.

Table 2.4 shows that crop farms have the highest application rates (1.7-1.8 kg per hectare) and these are almost twice as high the application rates of cattle farms (around 1 kg per hectare). Pig farms have intermediate application rates ranging from 1.1 to 1.3 kg per hectare. Furthermore, the table shows that soil type considerably affects pesticide application rates in that farms on sandy soil have lower application rates than farms on loamy soil. Finally, table 2.4 shows that full-time farms have higher application rates than part-time farms.

Similar results can be seen in table 2.5, which shows the total pesticide use in terms of standard dosage. The Treatment Frequency Index (TFI) is calculated to be 2.5 from a total pesticide use of 5.7 million standard treatment doses on a total rotation area of 2.3 million hectares.

Table 2.5. Pesticide use in terms of standard dosage and Treatment Frequency Index (TFI), 1996/97

Farm types	Application rates by pesticide groups (SD/ha) H	F	I	V	Total (TFI)
Crop farms, loamy soil	2.51	0.81	0.52	0.15	3.98
Crop farms, sandy soil	1.80	0.98	0.23	0.12	3.14
Cattle farms, loamy soil	1.78	0.35	0.29	0.07	2.49
Cattle farms, sandy soil	1.51	0.24	0.22	0.07	2.04
Pig farms, loamy soil	1.74	0.69	0.26	0.08	2.77
Pig farms, sandy soil	1.45	0.59	0.20	0.07	2.32
Part-time farms, loamy soil	1.31	0.47	0.19	0.04	2.02
Part-time farms, sandy soil	1.21	0.38	0.14	0.06	1.79
All farms	**1.63**	**0.53**	**0.25**	**0.08**	**2.49**
Distribution by pesticide groups	65%	21%	10%	3%	100%

Notes: herbicides (H); fungicides (F); insecticides (I) and growth regulators (V).
SD denotes standard treatment doses.
Calculations are based on 2.3 million hectares of agricultural land in rotation.

Again, crop farms have the highest application rates (3.1 – 4.0 SD per hectare), cattle farms have lower application rates (2 – 2.5 SD per hectare), and pig farms have intermediate application rates with TFI ranging from 2.3 –2.8, depending on soil type. Similar to the results from table 2.5, farms on loamy soil have higher application rates than farms on sandy soils and full-time farms have higher application rates than part-time farms. Also analogous to the results from table 2.4, herbicides and fungicides constitute the major groups (65 and 21 per cent, respectively) of total pesticide use. Contrary to the results shown in table 2.5, insecticides now constitute a larger proportion (10 %) of total pesticides use than growth regulators (3 %).

Although tables 2.4 and 2.5 in general show equivalent results, the use of different pesticide products by different farm types can explain the differences in the ranking of pesticide groups in the two tables. This implies that variations in pesticide use are not symmetric when measured by active ingredients and standard treatment doses, respectively.

A comprehensive description of aggregate pesticide use on crop, farm, regional and national level is given in Huusom (2001) along with an evaluation of the applied method and results.

3. Modelling economic behaviour related to fertilisers and pesticides

The quantitative relationships established in the previous chapter, between agricultural production and economic data on the one hand and physical quantities of nitrogen and pesticides on the other, can be applied as part of an analytical framework for evaluating economic and environmental effects of various policy actions. However, the relationships are static and their straightforward use is restricted to a rather limited range of policy instruments. For policy scenarios involving changes in e.g. prices, output- or input-related subsidies, restrictions on specific variables etc., account must be taken for the behavioural adjustments, and hence the changes in the level and composition of production, the use of inputs etc. caused by the considered policy changes. For such purposes, an econometric model (ESMERALDA) describing farmers' behaviour is applied. The model conforms to the data structure in FOI's agricultural accounts statistics database, and is hence compatible with the expanded data framework outlined in the previous chapter.

This chapter gives a brief introduction to the ESMERALDA model, and a more detailed description of the estimation of disaggregated behavioural parameters relating to nutrients and pesticides.

3.1. ESMERALDA – an econometric sector model for Danish agriculture

The ESMERALDA model describes production, input demands, land allocation, livestock density and various economic and environmentally relevant variables on representative Danish farms, and subsequently in the Danish agricultural sector at relevant levels of aggregation. See Jensen et al. (2001) for a detailed description of ESMERALDA. These variables are assumed to be functions of the economic conditions facing the farms (including agricultural prices, economic support schemes, quantitative regulations etc.), based on an underlying assumption that farmers exhibit economic optimisation behaviour. In the model, this behaviour is expressed in terms of a number of behavioural parameters – e.g. price elasticities. Such parameters have been estimated econometrically using anonymous farm account data from 1000-2000 Danish farms per year in the period 1973-74 to 1997/98, distinguishing eight different farm categories: part-time farms and full-time crop, cattle and pig farms on loamy and sandy soils, respectively. To each farm in the model, the most relevant of these eight sets of parameters is attached.

The model covers 15 lines of agricultural production (which is a slight aggregation of the sectors covered in the FOI accounts database): 7 cash crops (spring barley, winter barley, wheat, pulses, rape, potatoes and sugar beets), 3 roughage crops (fodder beets, green fodder in rotation and permanent grass), 2 cattle sectors (dairy and beef cattle), pigs, poultry and fallow. Of these, 11 outputs are assumed to be marketed in the model, whereas the outputs from roughage and fallow are not. Fallow is not supposed to yield an output, and the production of roughage is assumed to serve as on-farm input in cattle production. Along with the 11 commercial outputs, the model determines demands for 12 variable inputs in the short run (energy, labour, 3 nutrients from commercial fertilisers, 4 pesticide types, contract operations, purchased roughage and purchased concentrate feeds), and changes in activity levels (land allocation and livestock density) and input of capital in the long run.

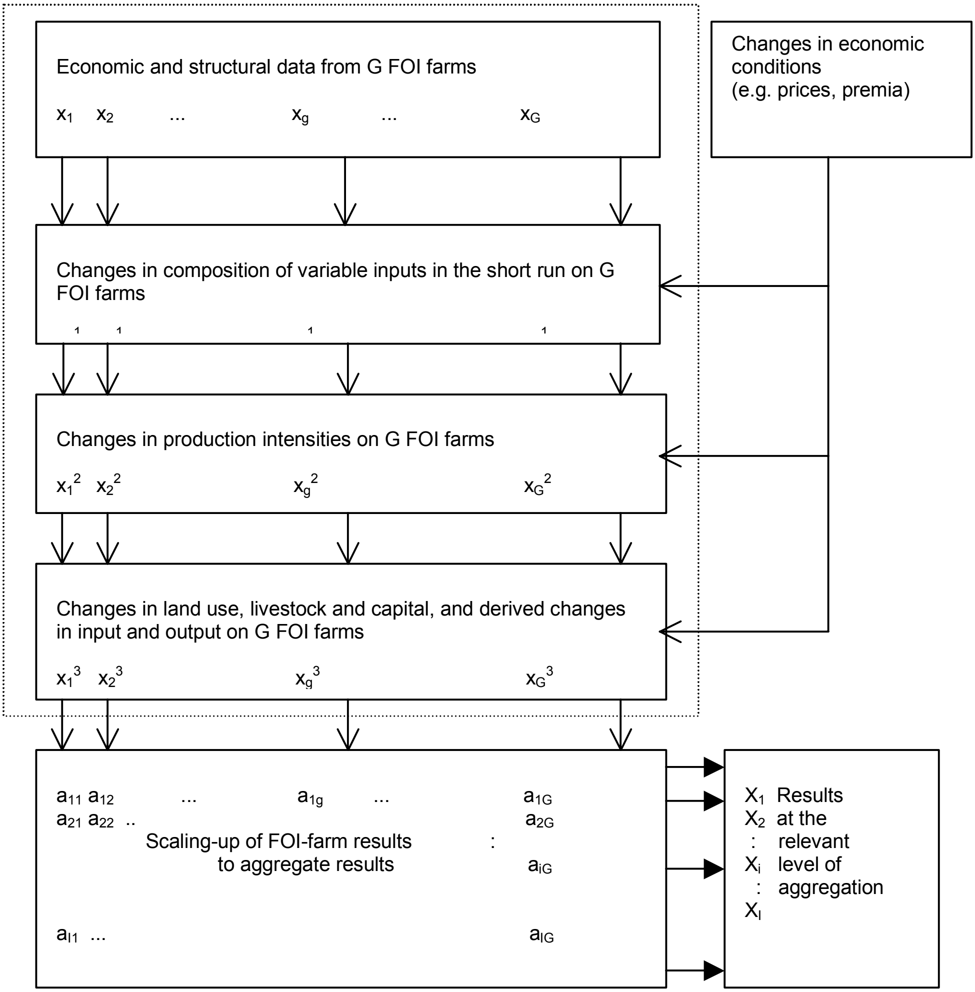
The main principle in the model is to determine economic behaviour on a number of representative Danish farms, and subsequently scale these farm level results up to the relevant level or type of aggregation. The economic behaviour at the farm level includes determination of input composition, production intensity in individual lines of production as well as activity levels (numbers of hectares or animals) in each line of production. Examples of relevant scaling-up schemes could be the construction of geographic (national or regional/local) or typologic (main production, farm size etc.) aggregates. The overall structure of the model is illustrated in figure 3.1.

The contents of the dotted box in the figure represent assumed behaviour at the farm level on each of the model's (approximately 2000) representative FOI farms, as responses to changes in economic conditions – represented by the upper right module in the diagram. Farm level adjustment takes place in three stages:

1) cost minimising adjustments in the composition of variable inputs,
2) short-run profit maximising adjustments in the yield levels in individual subsectors (keeping land allocation and livestock numbers constant), and derived adjustments in input applications
3) adjustments in activity levels (land allocation, livestock numbers) and capital, and derived changes in output and input quantities

In each stage, the behavioural adjustments (e.g. price changes) are determined by the estimated behavioural parameters.

Figure 3.1. Stages in agricultural adjustments in ESMERALDA



The part below the dotted box in figure 3.1 represents the scaling-up of individual model farms to the relevant level of aggregation. The scaling-up is carried out by means of an aggregation matrix, which contains a weight for each model farm according to how many farms it represents at a national level. Hence, the aggregation matrix represents the farm structure related to the considered grouping of farms. The aggregation matrix is assumed to be independent of the economic conditions specified in the upper right module. This assumption may be considered as a restrictive one.

However, a study by Wiborg & Rasmussen (1997) indicates, that developments in the Danish farm structure seems to have been fairly unaffected by observed changes in prices and regulations.

3.2. Price elasticity estimates for fertiliser and pesticide components

There exist only very few empirical studies of behavioural parameters for agricultural use of fertilisers and pesticides at a sufficiently detailed level, and at the same time taking into account the interrelations with other agricultural inputs, e.g. labour, energy, capital, feeds etc. A major reason for this rather limited amount of studies is the lack of data appropriate for such estimation. In the current study, an effort has been made to extract maximum behavioural information from available data, and combine this with information derived from theoretical assumptions, in order to estimate behavioural parameters related to the use of nutrients and pesticides. The estimations have been carried out in connection with the overall estimations of the ESMERALDA model, and the estimated elasticities are linked to (an hence consistent with) the general set of elasticities in the model.

The econometric methodology distinguishes an input substitution effect (i.e. for given yield levels, a change in relative input prices will lead to changes in the input composition) on the one hand, and a yield effect (changes in the optimal yield level due to changed price relations) on the other. In appendix 3, a brief description of the applied methodology is given, whereas a more detailed description is available in Jørgensen et al. (2000). Behavioural parameters have been estimated for eight farm categories: part-time farms and full-time crop, cattle and pig farms, on loamy and sandy soil, respectively.

Input substitution parameters

Input substitution parameters represent the changes in the composition of variable inputs due to changes in relative input prices for a given output level. For fertilisers, this substitution effect represents the substitution between mineral fertilisers and nutrients in animal manure. In the case of pesticides, the substitution effect represents substitution between chemical and mechanical pest abatement. In both cases, input substitution effects may also reflect the possibility of compensating lower fertiliser or pesticide applications with increased intensity of e.g. labour for harvesting, field maintenance etc. for maintaining a given yield level. For the group of full-time crop farms on loamy soil, table 3.1 gives the set of price elasticities representing input substitution effects due to changes in disaggregated fertiliser and pesticide prices.

Table 3.1. Output compensated elasticities between fertiliser and pesticide component prices and input use, crop farms on loamy soil, 1995

	Energy	Labour	Nitrogen	Phosphorus	Potassium	Herbicides	Fungicides	Insecticides	Service	Feeds
Price:										
Nitrogen	0.087	0.087	-1.245	0.477	0.477	0.087	0.087	0.087	0.087	0.132
Phosphorus	-0.006	0.225	0.155	0.000	-0.467	-0.294	-0.294	-0.294	0.028	-1.172
Potassium	0.317	-0.262	0.158	-0.476	-0.010	0.601	0.601	0.601	0.029	0.067
Herbicides	-0.211	0.089	0.109	0.109	0.109	-1.500	2.768	2.768	0.085	-0.329
Fungicides	0.014	-0.130	0.070	0.070	0.070	0.451	-0.057	-4.015	0.014	1.072
Insecticides	0.015	0.015	0.138	0.138	0.138	0.478	-4.249	-0.500	0.015	0.168

The price elasticities in the table represent the percentage change in the respective inputs due to a one per cent change in the considered input price. For instance, a one per cent increase in the price of nitrogen fertilisers leads to a 1.245 per cent decrease in the use of commercial nitrogen fertilisers and a 0.087 per cent increase in the use of energy, labour, pesticides and services. Hence, commercial nitrogen fertilisers can be considered as a substitute for all other inputs, possibly due to the possibility for substituting nitrogen in commercial fertilisers with nitrogen in animal manure in terms of a higher rate of utilisation – however at the cost of more labour, energy etc. Although the use of pesticides does not influence this utilisation rate, there seems to be substitution between nitrogen and pesticides. Hence, if the use of nitrogen fertiliser decreases, increased application of pesticides is necessary, if the output level is to be maintained. Correspondingly, if the prices of pesticides increase, the use of commercial fertilisers also increases. Moreover, an increase in the price of herbicides leads to an increase in the use of labour and services, which could be due to substitution between chemical and mechanical weed control.

As indicated above, the price elasticity estimates in table 3.1 represent an 'output compensated effect'. Thus, the elasticities should be interpreted as the cost minimising input adjustments due to a percentage change in a given input price, if the output – in terms of activity and yield levels in all agricultural sub-sectors – should be kept at its original level.

Yield-related parameters

Changes in e.g. fertiliser or pesticide prices affect the optimal yield level, due to decreasing marginal yields. Hence, if the price of an input increases, it may be profitable to reduce the application of the considered input and thus the planned yield level. Estimated cross-price elasticities between disaggregated fertiliser and pesticide prices and crop yield levels are given in table 3.2.

Table 3.2. Yield response elasticities, crop farms on loamy soil, 1995

	Spring barley	Wheat	Pulses	Potatoes	Sugar beets
Nitrogen price	-0.046	0.000	-	-0.100	-0.508
Phosphorus price	-0.032	-0.321	-0.132	-0.043	0.000
Potassium price	0.000	-0.081	0.000	-0.034	-0.876
Herbicide price	0.000	-0.288	-0.219	-0.091	0.000
Fungicide price	-0.009	0.000	-0.035	-0.027	0.000
Insecticide price	-0.009	-0.042	-0.028	-0.014	0.000
Growth regulator price		-0.006	-	-	-

Note: These elasticities are calculated for mean values of farms, where the crop is present.

Hence, a one per cent increase in the price of nitrogen leads to a decrease in the yield level for spring barley at 0.046 per cent, whereas no significant effect could be estimated in the other cereals sectors. The prices of insecticides and growth regulators have relatively low impacts on yield levels, whereas the price of herbicides has some impact on yields in wheat, pulses and potatoes.

In general, the yield responses reflected in table 3.2 lead to lower input demands – including lower demands for fertilisers and pesticides – when the prices of these inputs increase. Hence, a price change of an input leads to changed yield levels in different sub-sectors, which in turn leads to changed input demands. These effects are summarised in table 3.3, where yield-determined own- and cross-price elasticities are given for crop farms on loamy soil.

For example, a 1 per cent increase in the price of nitrogen leads to crop yield reductions, which again lead to a 0.186 per cent decrease in the demand for nitrogen fertiliser, a 0.015 per cent decrease in the demand for herbicides etc. All elasticities in table 3.3 are non-positive, implying that an increase in any of the input prices cannot lead to higher yield levels in any crop sector, and lower yield levels in general lead to lower input demands.

Table 3.3. Own- and cross-price elasticities for fertilisers and pesticides due to yield effects, crop farms on loamy soil, 1995

	Nitrogen	Phosphorus	Potassium	Herbicides	Fungicides	Insecticides	Growth regulators
Price:							
Nitrogen	-0,186	-0,038	-0,804	-0,015	-0,025	-0,025	0,000
Phosphorus	-0,022	-0,557	-0,137	-0,173	-0,022	-0,152	-0,131
Potassium	-0,282	-0,137	-1,397	-0,044	-0,005	-0,038	-0,033
Herbicides	-0,028	-0,510	-0,131	-0,171	-0,028	-0,145	-0,117
Fungicides	-0,008	-0,013	-0,004	-0,007	-0,008	-0,008	0,000
Insecticides	-0,006	-0,078	-0,019	-0,025	-0,006	-0,023	-0,017
Growth reg.	0,000	-0,010	-0,002	-0,003	0,000	-0,002	-0,002

Tables 3.1 and 3.3 contain two sets of own- and cross-price elasticities between individual nutrients and pesticides, representing an input substitution and a yield effect, respectively. In principle, these two effects may add to the total effect of fertiliser or pesticide price changes. However, the elasticity matrices are not strictly additive because changes in input composition may in itself lead to changes in the profit maximising yield level, and these feed back effects are not consistent with those represented by the pure yield effect

Similar sets of price elasticities have been estimated for the 7 other farm categories. It should be noted that the above elasticities represent 'short-run' effects of price changes, i.e. activity levels in different agricultural sub sectors underlying these elasticity estimates are assumed to be fixed.

It should be noted that the economic model framework builds on the assumption that farmers seek to maximise profits. Hence, profits are assumed to be maximised at the outset and adjustments in production intensity, input composition, land allocation etc. take place as a result of changed relative profitabilities within the range of individual agricultural activities, e.g. due to new regulations or changed market conditions. The assumption of profit maximisation may be considered restrictive in the sense that other rational optimisation criteria may exist. For instance, some part-time farmers may seek to minimise labour input if their main source of income is from off-farm employment, some farmers may maximise utility from undertaking nature-friendly production even if this implies reduction in profits compared to conventional farming, etc. Still the optimisation criterion of the model concept is considered to be valid in describing the behaviour of the agents in the agricultural sector.

In the next chapter, the model concept is applied to two scenarios of herbicide reduction.

4. Economic impacts of tranferable and non-tranferable herbicide quotas

The aim of this chapter is to quantify the economic and environmental effects as well as the cross-achievements between different policy measures. Two illustrative scenarios are used to integrate the econometric behavioural model (presented in chapter 3) with data on nitrogen and pesticide use (from chapter 2).

Section 4.1 contains a brief, non-technical description of the principles behind individually transferable quotas in pesticide regulation. Section 4.2 introduces the scenarios of implementing individual versus transferable quotas on pesticide use in order to reach the goal of the Danish Pesticide Action Plan. The actual modelling of the scenarios is found in section 4.3. Finally, the results of the analyses are presented and discussed in section 4.4 and 4.5, respectively.

4.1. Transferable quotas in pesticide regulation

The Danish pesticide policy goal is formulated in terms of pesticide reductions. Hence, transferable quotas in the present report are modelled as a system of tradeable reduction requirements. This is equivalent to a scenario where the policy goal is formulated in terms of maximum allowable pesticide use that is achieved through a system of tradable permits for using pesticides.

A scheme of tradeable permits entails that the regulator determines the acceptable aggregate quantity of pesticide use as well as the initial allocation of permits between farmers. Because of differences in the farmers' cost structures, there exists an economic potential for trading permits for using pesticides. Trading results in an equilibrium price at which farmers that incur high costs from a reduction in pesticide use will prefer to buy permits rather than reduce pesticide use. Conversely, farmers that find a reduction in pesticide use relatively cheap may profit from reducing pesticide use more than required by the initial allocation and thus be able to sell permits.

A system of tradeable permits has several benefits that make it an efficient instrument in pollution control. Firstly, a system of tradeable permits is cost effective (which means that the total costs of compliance are minimised), as farmers with low costs of reducing pesticide use are encouraged to do so first, while farmers with high costs of reducing pesticide use are allowed to purchase additional pesticide use permits. Thus, transferability ensures a decentralised and flexible adaptation to environmental regulation. Secondly, in a system of tradeable permits it is possible to address the goal of pesticide reduction directly, whereas reductions obtained through a tax or subsidy depend on the actual substitution possibilities. Thirdly, distributional aspects can be addressed in the actual implementation of the system through the initial allocation of permits (for example, the initial permits could be auctioned away or distributed on the basis of historical pesticide use). Finally, tradeable permits can, compared to non-tradeable permits potentially encourage technological innovation by providing a financial incentive for the individual farmers to reduce pesticide use more than required by the initial allocation of permits.

Graphical illustration

To illustrate the principles of using transferable quotas in pesticide regulation, consider two farmers, A and B. The farmers' demand for pesticides are represented in figure 4.1 by the curves, a and b. The curves indicate each farmer's marginal willingness to pay for the use of pesticide. Moving from right to left, the curves can be interpreted as the marginal costs of reducing pesticide use. The slopes of the curves indicate that it is relatively cheap to reduce pesticide use a little, while it is relatively expensive to reduce pesticide use significantly.

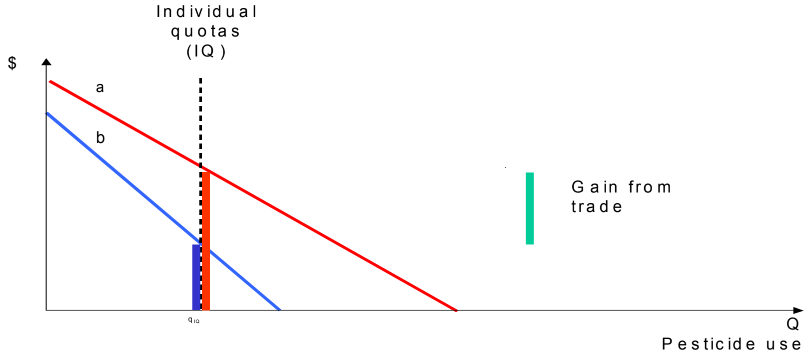
Administrative regulation can be modelled as a unified cap on all farmers' pesticide use, corresponding to individual quotas on pesticide use for all farmers. In figure 4.1, the line IQ illustrates an individual permit to use q_{IQ} pesticides. The regulation imposes an economic loss corresponding to the triangles delineated by IQ, the horizontal axis, and the lines a and b for farmers A and B, respectively. From figure 4.1 it can be seen that the economic loss of farmer A is larger than the loss of farmer B.

If the farmers are allowed to trade a permit to use one extra unit of pesticide amongst each other, farmer A will increase his pesticide use by one unit and farmer B will reduce his pesticide use by one unit[13]. The aggregate use of pesticides is unchanged by this trade, and farmer A achieves a gain corresponding to the high column while farmer B only foregoes a gain corresponding to the low column. The difference between the two amounts – shown as the column to the right – corresponds to the trading potential, i.e. the gain that arises from trading reduction requirements. The aggre-

[13] In this presentation of the subject, strategic behaviour and information asymmetries related to the bargaining situation is left out of consideration for ease of exposition.

gate use of pesticides is thus similar to the conventional administrative regulation, but the costs are lower.

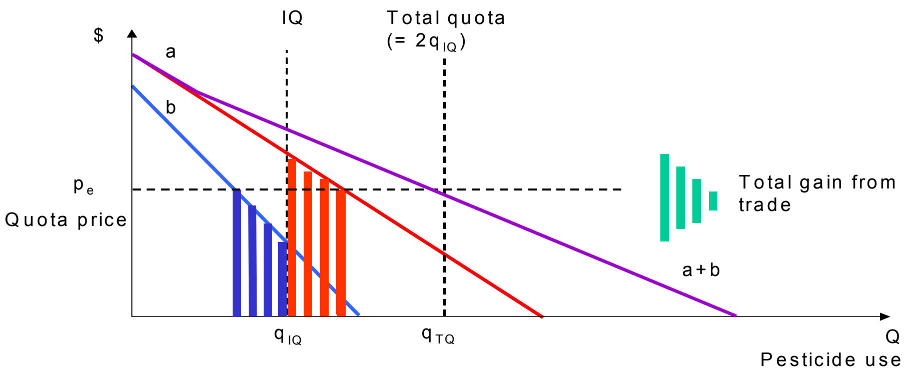
Figure 4.1. Conventional administrative regulation of pesticide use modelled by individual (non-transferable) quotas



In figure 4.2, the same reasoning is repeated for additional units of pesticide use up to the point where farmer A's potential marginal gain from trade is equal to farmer B's potential marginal loss, i.e. to the point where the two "marginal" columns have the same size. At this level, the trading potential is exhausted and trading will cease. The resulting total gain from trade is illustrated by the triangle to the right in figure 4.2. In sum, when there are differences in the cost structures between farmers then transferable quotas can bring about efficiency gains compared to a conventional administrative regulation.

Figure 4.2 can also be used to illustrate the relationship between tradable permits and a tax. A tax equal to the equilibrium price of permits p_e would result in the same overall reduction in pesticide use. The tax is determined as the intersection between the aggregate demand curve a+b and the aggregate supply curve, i.e. the total quota that in a 2-farm model is equal to twice the size of the individual quota. The main difference between a tax and tradeable permits is that a tax is determined centrally while the equilibrium price is determined decentrally through trading in the market. This implies that in order to reach a certain goal of pesticide use, determination of a tax requires information about the pesticide application behaviour of the farmers while a scheme of transferable permits only requires determination of the total amount of pesticides that can be used.

Figure 4.2. Trading continues until the trading potential is exhausted and an equilibrium price is found



4.2. Two scenarios for reducing herbicide use

As stated in the introduction to this chapter, two alternative scenarios for pesticide regulation are analysed in the following. In both scenarios, pesticide use is to be reduced in line with the goals of the Pesticide Action Plan (Miljø- og Energiministeriet, 1998). Using 1981-85 as the reference period, the plan stipulated a 50 per cent reduction in national pesticide use by 1997 as measured by total amount of active ingredients as well as by the Treatment Frequency Index (TFI). The goal of reducing applied active ingredients by 50 per cent by 1997 has more or less been achieved, partly through a ban on certain pesticides. However, a major contribution to the reduction in active ingredients stems from the introduction of herbicides with low quantities of active ingredients per standard treatment dose. For this reason, the amount of active ingredients is not considered very appropriate as an environmental indicator, and instead the Treatment Frequency Index (TFI) is used to describe application behaviour on different farms.

The target TFI of the Pesticide Action Plan is shown in table 4.1, including national estimates based on total pesticide sales for 1997. For example, the total TFI should have been reduced from 2.67 to 1.34 by 1997, however the index in 1997 was 82 pct. above the goal. Furthermore, the table shows that the main reduction needs to come from reducing herbicide use. According to the table, herbicide TFI should be reduced to 0.64.

Table 4.1. The Pesticide Action Plan – baseline, reduction target and observed values for 1997

	Treatment Frequency Index (TFI)		
Pesticide groups	1981-85	Target for 1997	Observed in 1997
Herbicides	1.27	0.64	1.65
Fungicides	0.81	0.41	0.46
Insecticides	0.45	0.23	0.30
Growth regulators	0.14	0.07	0.05
Total	2.67	1.34	2.45

Source: Miljøstyrelsen (1998, table 3.2).

The analysis is restricted to one group of pesticides – herbicides - in order to simplify the modelling of the two scenarios. The reasons for choosing herbicides in the analysis are that most of the applied pesticides are herbicides, and this group still lies furthest from the Pesticide Action Plan target.

Policy effects are expected to comprise reduced pesticide use through changes in production levels, a range of farm and sector economic effects, changes in land use and effects on nitrogen use and –surplus.

Although the two scenarios aim at the same target TFI at the national level, they differ in one crucial respect. In scenario 1, the target TFI constitutes an upper limit to herbicide application on each individual *farm*. In scenario 2, the target TFI is considered an upper limit at the *sector* level. The reductions in scenario 2 are implemented via a scheme of transferable quotas, as explained in section 4.1.

Box 4.1 summarizes the design of the scenarios.

Box 4.1.	**Scenarios for reducing the Treatment Frequency Index for herbicides, TFI_{Herb}**	
	Scenario 1 Individual quotas for herbicide use	Scenario 2 Transferable quotas for herbicide use
Reduction requirement	Treatment Frequency Index for herbicides reduced to the level stipulated by the Pesticide Action Plan	
Regulation level	Target TFI achieved at the *farm* level	Target TFI achieved at the *sector* level

4.3. Modelling the scenarios

Technically, the regulation defined in the two scenarios is implemented in the model framework by determining an implicit tax rate on herbicides, which ensures compliance with the relevant regulation for each farm in the model. The implicit tax rate represents the quota rent or 'shadow price' related to the imposed restriction on herbicide use, and it is assumed that farmers will allocate resources according to this implicit price change. However, in contrast to an explicit tax on herbicides, the quota does not generate tax revenues – hence the 'implicit tax revenue' remains within the farm in this scenario. The implicit tax rate which results in the targeted TFI is calculated in an iterative procedure, taking into account any adjustments in e.g. activity composition.

In scenario 1, an implicit tax rate is calculated for each individual model farm[14] in order to ensure compliance with the reduction requirement at the farm level. These individual tax rates represent the extent to which the requirement is binding, which in turn represents the farm's deviation from the target and the farm's possibility for adjustments. The more binding the reduction requirement on the farm, the higher the implicit price of herbicides will be and hence the larger the incentives will be to substitute herbicides for other inputs or adjust the production composition.

In scenario 2, a uniform implicit tax rate for all farms is calculated that induces the same aggregate reduction in herbicide use as in scenario 1. This tax rate can be considered an equilibrium price of trading the reduction requirement, cf. section 4.1. Hence, a farmer with an individual implicit tax rate above the uniform tax rate will profit from paying other farmers to execute his reduction requirements, whereas

[14] Except for farms that already comply with the regulation.

farmers with an individual implicit tax rate below the uniform rate may be willing to accept additional reduction requirements at a payment equal to the uniform tax rate.

4.4. Results

In the following, results from the scenarios are presented. The results include the effects of the regulation on pesticide use, land use, and cross effects on fertilizer use, as illustrated by changes in nitrogen surplus. Furthermore, the costs of the regulations are calculated for farm types and for the sector as a whole.

4.4.1. Effects on pesticide use

The cap on TFI_{Herb} on the individual farms in scenario 1 yields a national TFI_{Herb} of 0.62 which is slightly below the target TFI_{Herb} of 0.64.

In table 4.2, the changes in pesticide use in the two scenarios are shown along with the base situation. The table shows that a 62 per cent reduction in herbicide use leads to a 39 per cent reduction in total pesticide use in both scenarios. In absolute terms, this translates into a reduction of the TFI from 2.49 in the base situation to a TFI of 1.53 and 1.52 in scenarios 1 and 2, respectively.

Table 4.2. Pesticide use (TFI) in base situation and scenarios 1 and 2

Pesticide group	Base TFI	Scenario 1 TFI	Scenario 1 Change (%)	Scenario 2 TFI	Scenario 2 Change (%)
Herbicides	1.63	0.62	-62	0.62	-62
Fungicides	0.53	0.54	1	0.54	2
Insecticides	0.25	0.26	6	0.25	2
Growth regulators	0.08	0.11	33	0.11	31
All pesticides	2.49	1.53	-39	1.52	-39

Source: ESMERALDA.

Because of input substitution, the reduction in herbicide use is partly offset by slight increases in the use of other pesticide types. The increases are insignificant in absolute terms – corresponding to a TFI of around 0.03. For growth regulators where the relative increases are more than 30 per cent due to the low initial level.

In sum, the restriction in the use of herbicides leads to an overall reduction of total pesticide use of the same relative magnitude in the two scenarios. Also, the use of pesticides other than herbicides increases slightly in both scenarios.

The use of herbicides by farm types in the base situation and the two scenarios is shown in table 4.3.

Tabel 4.3. Herbicide use in base situation and scenarios 1 and 2 distributed by farm types (TFI_{Herb})

	Base situation	Scenario 1		Scenario 2	
	TFI_{Herb}	TFI_{Herb}	Change (%)	TFI_{Herb}	Change (%)
Crop farms, loam	2.51	0.62	-75	1.11	-56
Crop farms, sand	1.80	0.64	-64	0.70	-61
Cattle farms, loam	1.78	0.63	-65	0.62	-65
Cattle farms, sand	1.51	0.60	-60	0.56	-63
Pig farms, loam	1.74	0.62	-64	0.66	-62
Pig farms, sand	1.45	0.62	-57	0.52	-64
Part-time farms, loam	1.31	0.62	-53	0.51	-61
Part-time farms, sand	1.21	0.63	-48	0.42	-65
All farms	1.63	0.62	-62	0.62	-62

Source: ESMERALDA.

There are substantial differences in the use of herbicides among farm types between the two scenarios. In scenario 1, the TFIs for herbicides for different farm types are all around 0.64 because the cap acts as a cap on the use of herbicides is implemented at the farm level. The large differences in relative reductions indicate large differences in the costs of compliance to the reduction requirements. In relative terms, the reductions vary from 48 to 75 per cent with the smallest relative reductions on part-time farms on sandy soils and the largest on crop farms on loamy soils. Conversely in scenario 2, TFIs for herbicides for different farm types are seen to vary within a range from 0.42 to 1.11, whereas relative reductions cover a comparatively narrow range from 56 to 65 per cent with the smallest relative reductions on crop farms on loamy soils. In sum, the relative impacts of the regulation in scenario 2 seem more evenly distributed among farm types compared to scenario 1, whereas the resulting absolute herbicide TFIs are more homogenous in scenario 1.

Turning to the effects of herbicide reductions on total pesticide use on different farm types, table 4.4 presents the pesticide TFI's in the base situation as well as in the two scenarios.

Table 4.4. Pesticide use (TFI) in base situation and scenarios 1 and 2 distributed by farm types

	Base situation	Scenario 1		Scenario 2	
	TFI	TFI	Change (%)	TFI	Change (%)
Crop farms, loam	3.98	2.43	-39	2.86	-28
Crop farms, sand	3.14	1.77	-44	1.83	-42
Cattle farms, loam	2.49	1.66	-33	1.61	-35
Cattle farms, sand	2.04	1.31	-36	1.23	-40
Pig farms, loam	2.77	1.68	-39	1.71	-38
Pig farms, sand	2.32	1.31	-44	1.21	-48
Part-time farms, loam	2.02	1.35	-33	1.37	-32
Part-time farms, sand	1.79	1.18	-34	0.96	-46
All farms	2.49	1.53	-39	1.52	-39

Source: ESMERALDA.

When considering total pesticide use – and not just herbicides – the distribution of the relative reductions by farm types differ significantly from table 4.3, as the relative reductions in scenario 2 are seen to be in a wider range than in scenario 1. This is taken to imply that the possibilities of input substitution – e.g. substituting herbicides with other types of pesticides – depends on the type of farm, e.g. crop farms on loamy soils are able to reduce the impacts of the regulation on total pesticide use.

The general pattern of pesticide use in the scenarios is much like the pattern in the base situation; full time farms tend to use more pesticides than part-time farms, farms on loamy soils generally have higher application rates than farms on sandy soils, and usually crop farms are the most pesticide intensive farm type. However, contrary to the base situation, cattle and pig farms have similar pesticide uses in the two regulations.

The results presented in table 4.3 indicate that in the case of transferable reduction requirements it is profitable for farms with high levels of herbicide use in the base situation to pay farms with a low initial herbicide use to take over some of their reduction requirement, e.g. crop farms will pay part-time farms on sandy soils to implement the reduction required by the crop farms.

Geographical differences in pesticide use

The geographical distribution of pesticide use will vary according to the regulation, and scenario 1 yields a more uniform reduction in herbicide and pesticide use than scenario 2, cf. table 4.5.

Table 4.5. Regional distribution of herbicide and total pesticide use in base situation, scenario 1 and scenario 2.

	Herbicides, TFI_{Herb}			Pesticides, TFI		
	Base	Scenario 1	Scenario 2	Base	Scenario 1	Scenario 2
Capital region	1.57	0.62	0.58	2.52	1.67	1.64
West Zeeland	1.81	0.63	0.73	2.78	1.68	1.79
Storstrøm	2.45	0.62	0.95	3.70	2.05	2.38
Fyn	1.90	0.64	0.75	2.98	1.97	2.08
South Jutland	1.45	0.60	0.57	2.26	1.43	1.37
Ribe	1.44	0.63	0.59	2.03	1.22	1.14
Vejle	1.53	0.61	0.61	2.29	1.39	1.44
Ringkøbing	1.35	0.61	0.49	2.05	1.16	1.03
Århus	1.52	0.62	0.56	2.41	1.51	1.45
Viborg	1.41	0.62	0.51	2.09	1.38	1.28
North Jutland	1.55	0.62	0.54	2.31	1.41	1.30
Denmark	1.63	0.62	0.62	2.49	1.53	1.52

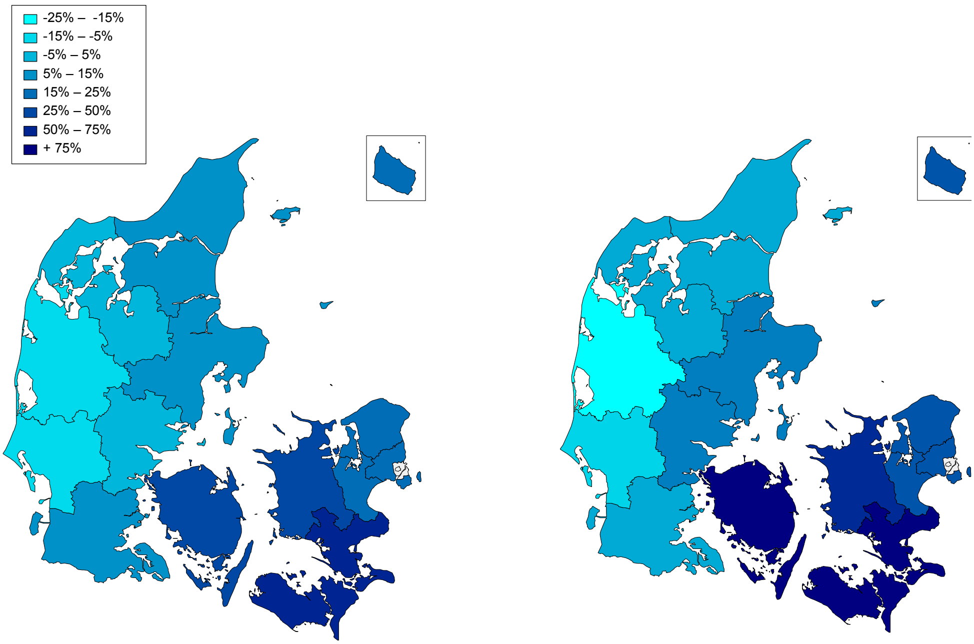
Source: ESMERALDA.
Note: The Capital region includes the island of Bornholm and the counties of Frederiksborg, Roskilde and Copenhagen. For geographic localisation of the regions, see Appendix 4

If only herbicides are considered, the table shows that the regional absolute values for TFI_{Herb} fall within a narrow range in scenario 1 and in a much wider range in scenario 2. This is a direct consequence of the structure of the regulations; scenario 1 is a cap on individual farms' herbicide use while scenario 2 reflects the possibilities for redistributing herbicide use among farms and regions dependent on the profitability of herbicide application of individual farms.

Considering total pesticide use, the table shows that the regions in both scenarios with the highest application rates are the same as in the base situation and vice versa. Also, more or less the same pattern of regional variations that was observed for herbicides - albeit less prominently - is seen in the regional pesticide application levels, implying that the total use of pesticides is more unevenly distributed regionally in scenario 2 than in scenario 1.

The regional distribution of pesticide use in the two scenarios is illustrated in figure 4.3. The shading of the regions reflects the degree to which the regional TFI meets the target of the Pesticide Action Plan, cf. table 4.1. Note that the results for the Capital region includes the island of Bornholm situated in the upper right-hand corner of the map.

Figure 4.3. Regional distribution of pesticide use in scenario 1 (left) and scenario 2 (right)



Source: ESMERALDA
Note: Range of TFI in intervals according to relative deviation from target level of 1.34.

From the figure, a ‘polarization’ effect in some regions can be discerned, i.e. that if these regions have comparatively low TFI’s in scenario 1, they will have even lower TFI’s in scenario 2 and vice versa. Consider for example the counties of Ringkøbing and Funen as representing regions with comparatively low and high TFIs in the base situation, respectively. In going from scenario 1 to scenario 2, the TFI of Ringkøbing is reduced from 13 to 23 per cent below the target TFI, while the TFI of Funen increases from 47 to 54 per cent above the target TFI.

4.4.2. Shifts in land use

The reductions in herbicide use in the two scenarios are the results of several adaptations to the regulation by the individual farmers cf chapter 3. A change in the crop composition (land-use displacement) can be anticipated as one way for the farmers to adapt to the reduction requirements.[15]

Restrictions on the use of herbicides affect the competition between different agricultural land uses. Hence, crops highly dependent on the use of herbicides are expected to become less competitive in comparison with crops less dependent on herbicides. These competitive effects may differ on different farm types, as the initial land use as well as the restrictiveness of the imposed herbicide reduction requirement differs among these farm types.

Table 4.6 shows the relative composition of crops in the base situation as well as in the two scenarios along with the changes relative to the base situation for the two scenarios.

In both scenarios, herbicide reduction requirement favours cereal production. Spring cereals increase by some 12 per cent, whereas winter cereals only increase by 4 – 5 per cent. These results are plausible, as pesticide application rates are lower in spring cereals than in winter cereals. Conversely, the production of seed crops and root crops is reduced significantly – from 26 to 36 per cent – in both scenarios compared to the base situation. These results are also intuitively appealing, as high pesticide application rates are found in these crops. The fact that the acreage of roughage, grass and fallow is virtually unchanged in the scenarios might be more surprising, as it would be expected that the acreage of these crops would increase because of their low pesticide application rates. However, due to quotas on milk production and beef cattle premia, the potential for expanding these productions, and hence the roughage area, is limited.

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[15] Also, reductions in the use of herbicides can to some extent be off-set by increased use of other input factors, including fertilizers and other types of pesticides (input factor substitution), or can lead to a general extensification of the cropping and as such lead to lower yields (yield adaptations). Both input substitution and yield adaptations reflect changes in the cropping intensity.

Table 4.6. Crop composition in base situation, scenario 1 and scenario 2, per cent of agricultural area

	Crop composition			Change from base	
	Base	Scen. 1	Scen. 2	Scen. 1	Scen. 2
Spring cereal	25	28	28	12	11
Winter cereal	33	34	34	4	5
Seed crops	10	7	6	-33	-53
Root crops	6	4	4	-27	-36
Roughage	14	14	14	1	1
Fallow and permanent grasslands	13	14	13	1	1
Total	100	100	100		

Source: ESMERALDA

Aggregation:
Spring cereal: Spring barley, oats
Winter cereal: Winter barley, wheat, rye, and other cereals
Seed crops: Seeds for sowing, peas, rape
Root crops: Potatoes, sugar beets, fodder beets
Roughage: Silage cereals, grass in rotation

Table 4.7 shows the changes in crop composition by farm type in the two scenarios relative to the base situation. The crops have been further aggregated compared to table 4.6 to improve the overview of the results.

For most farm types, the effects on crop composition are similar in the two scenarios. Even so, large variations between farm types can be seen to exist, especially when considering seed and root crops.

In order to quantify the effects on the use of pesticides solely resulting from shifts in land allocation – all other factors kept constant, e.g. by maintaining individual farmers' application rates – further analysis shows that only 6 per cent of the reduction of total pesticide use is attributable to shifts in crop composition. The major adjustment to herbicide reduction requirements – 94 per cent – is thus a result of changes in cropping intensity, and input substitution among different types of pesticides as well as between pesticides and other factors of production.

In sum, the aggregate land use and the implied impacts on landscape in the two scenarios are considered to be minor, although the areas with seed and root crops may be expected to decrease. Moreover, the differences in land use between the two scenarios are very small and from the table no significant differences can be discerned. This supports the result that the adjustment to the herbicide reduction is only to a small extent borne by changes in the crop composition.

Table 4.7. Changes in crop composition distributed by farm types, per cent

	Cereals		Seed crops		Root crops		Roughage, fallow and permanent grasslands	
	Scen.1	Scen.2	Scen.1	Scen.2	Scen.1	Scen.2	Scen.1	Scen.2
Crop, loam	11	10	-38	-38	-16	-12	2	0
Crop, sand	19	19	-40	-41	-75	-77	0	0
Cattle, loam	6	7	3	3	-30	-32	0	0
Cattle, sand	12	13	-71	-74	-31	-31	1	1
Pig, loam	10	10	-56	-56	-1	-1	1	1
Pig, sand	4	5	-19	-22	-13	-10	2	2
Part-time, loam	-1	0	1	0	0	-2	5	1
Part-time, sand	4	6	-26	-35	-10	-9	1	1
All farms	8	8	-33	-36	-27	-26	1	1

Source: ESMERALDA

4.4.3. Nitrogen surplus

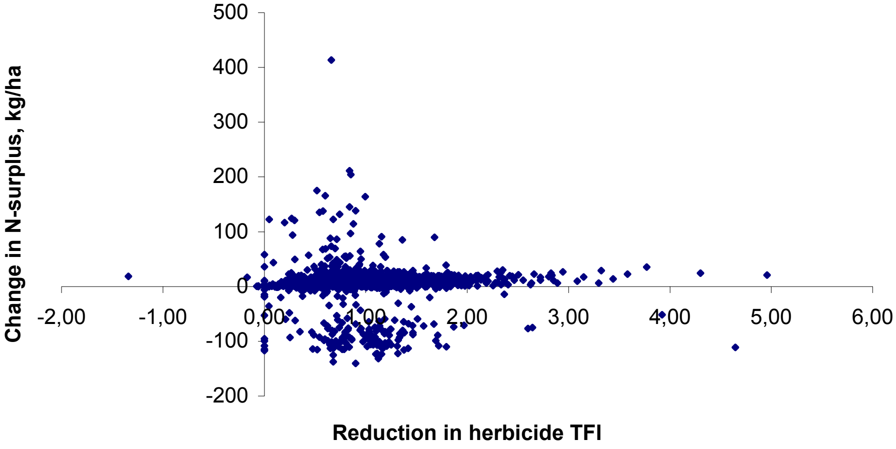
As previously mentioned, the reduction in herbicide use implies several changes in production: input substitution, yield level reduction and land reallocation. In the following, the effects on the use of nitrogen and hence changes in the nitrogen surplus induced by the requirement for herbicide reduction are considered.

The impact on nitrogen use is composed of an input substitution effect (herbicide use is to some extent replaced by nitrogen in the input composition, which in general increases the use of nitrogen), a yield level effect (reduced herbicide reduces the yield level, which in turn reduces the need for fertilisers) and a land allocation effect (changes in the crop composition affects the average use of nitrogen fertiliser). A priori, the direction of the latter effect is ambiguous.

The nitrogen surplus is affected by changes in nitrogen use and by changes in nitrogen removals, in terms of nitrogen contents in harvested crops. The main effects in this respect are a yield reduction and a land allocation effect as nitrogen applications and removals differ among crops.

The interrelations between changes in herbicide use – in terms of standard treatment doses per hectare (SD per ha) – and nitrogen surplus (kg per hectare) in scenario 1 on the model farms are illustrated in figure 4.4.

Figure 4.4. Herbicide reductions and change in nitrogen surplus, scenario 1.



Source: ESMERALDA

A general impression from the figure is that the interrelation between changes in herbicide use and nitrogen surplus is somewhat ambiguous, although the effect of herbicide reduction tends to increase the nitrogen surplus on the majority of sample farms.

In scenario 2 (not shown), a similar impression arises, although the variation between farms is somewhat more modest, because the extreme results for some farms in scenario 1 are replaced by more modest ones, due to the possibility of transferring the reduction requirements in scenario 2.

Table 4.8 shows the average nitrogen surpluses by farm types, in total 1000 tons N and in kg N per hectare. Relative to the base situation, the total nitrogen surplus increases by 13,000 and 7,000 tons in scenarios 1 and 2, respectively. This corresponds to an increase of more than 4 per cent in the case of individual reduction requirements and slightly over 2 per cent in the case of transferable reduction requirements. Furthermore, the herbicide reductions implied by the two scenarios lead to an increase in the per hectare nitrogen surplus of all farm types, except for cattle farms on sandy soil. The largest relative increases are seen on crop farms and on part-time farms on loamy soils.

Tabel 4.8.	**Estimated nitrogen surplus by farm types**					
	Base situation		Scenario 1		Scenario 2	
	Nitrogen surplus 1000 tons N	Nitrogen surplus kg N/ha	Nitrogen surplus 1000 tons N	Nitrogen surplus kg N/ha	Nitrogen surplus 1000 tons N	Nitrogen surplus kg N/ha
Crop, loam	21	65	27	83	25	77
Crop, sand	22	81	27	99	25	93
Cattle, loam	19	156	20	168	20	165
Cattle, sand	95	144	89	134	86	131
Pig, loam	30	138	32	148	32	146
Pig, sand	63	167	65	172	65	172
Part-time, loam	15	64	18	75	18	77
Part-time, sand	35	80	36	83	36	83
All farms	302	114	315	119	309	117

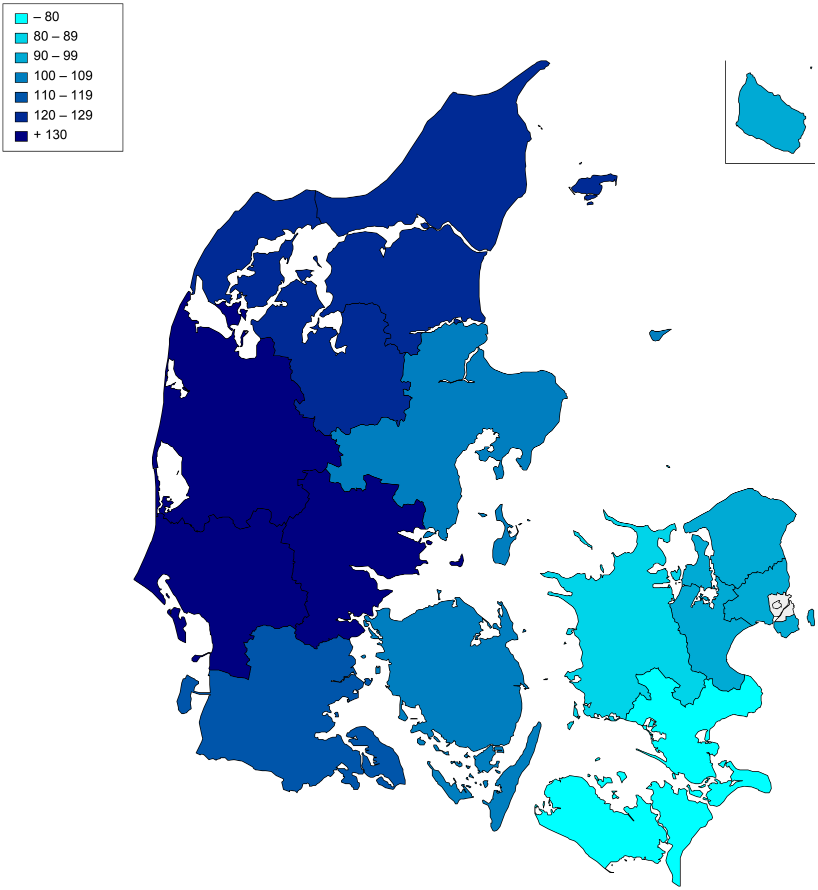
Source: ESMERALDA
Note: Excludes trade in manure between farms.

In sum, the reductions in herbicide use lead to slight increases in the total nitrogen surplus with the increase in scenario 1 being somewhat greater than in scenario 2. From an aggregate point of view, scenario 2 is thus preferable to scenario 1 if an objective of minimising total nitrogen surplus is pursued.

Figure 4.5 illustrates the regional distribution of nitrogen surpluses in the base situation as indicated by the shading explained in the legend box. Note that the results for the Capital region include the island of Bornholm in the upper right-hand corner. The figure illustrates the general observation that the highest nitrogen surpluses are observed in the Western parts of Denmark.

Table 4.9 presents the regional nitrogen surpluses in the base situation and the scenarios along with the relative changes from the base situation. Table 4.9 shows that the highest relative changes in the scenarios are observed in the Eastern parts of Denmark, and that the percentage changes in the two scenarios are very similar. Note that due to data requirements, the island of Bornholm is included in the capital region even though Bornholm especially with respect to animal husbandry differs markedly from the rest of that region.

Figure 4.5. Nitrogen surplus in base situation (kg per hectare)



Source: ESMERALDA

The table reflects the prevailing farm types in the different regions. For example, the lowest nitrogen surpluses are observed in regions dominated by crop farms, e.g. the Capital region and the counties of West Zealand and Storstrøm, and regions with high nitrogen surplus are dominated by livestock production. In total, the reduction of herbicide use is accompanied in both scenarios by slight increases in nitrogen surplus per hectare. Regionally, the largest relative increases occur in the regions that initially have the lowest nitrogen surplus, whereas the counties of Viborg and North Jutland –

that in the base situation have relatively high nitrogen surpluses – experience slight decreases. Only in one region – the county of Ribe – is the effect of the herbicide reduction with respect to changes in nitrogen surplus dependent on the choice of scenario, in that scenarios 1 and 2 imply an increase and a decrease in nitrogen surplus, respectively.

Table 4.9. Estimated nitrogen surplus by regions

	Base kg N/ha	Scenario 1 kg N/ha	Scenario 2 kg N/ha	Change from base (%) Scenario 1	Change from base (%) Scenario 2
Capital region	90	99	99	11	10
West Zealand	88	101	100	14	13
Storstrøm	76	92	89	20	16
Fyn	106	116	114	10	8
South Jutland	111	117	115	6	3
Ribe	134	136	133	2	-1
Vejle	137	143	139	4	1
Ringkøbing	134	139	136	4	1
Århus	108	110	109	2	1
Viborg	129	124	124	-3	-4
North Jutland	124	122	120	-1	-3
Denmark	114	119	117	4	2

Source: ESMERALDA.
Note: For geografical localisation of the regions see Apendix 4.

4.4.4. Economic effects

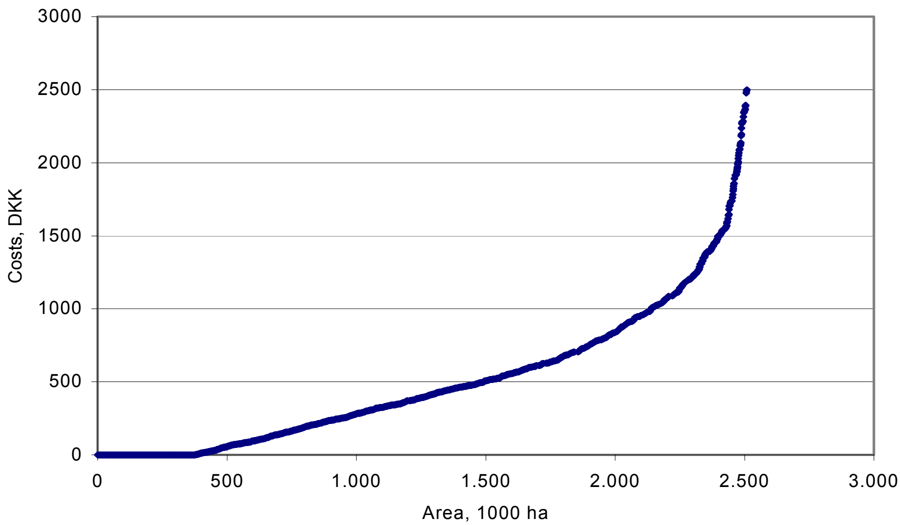
The extent to which farms adjust to the regulation, especially in scenario 1, differ between farm types – as was clear from table 4.4 – but also between individual farms within the 8 farm types. Some farms have to reduce herbicide use significantly, whereas other farms need to reduce more moderately. Some farms apply less herbicide than the allowed standards, and these farms will not be affected by the individual standards imposed by the regulation under scenario 1. Due to these variations as well as differences in the cost structures of the individual farms, the resulting costs will also differ markedly between farm types and between individual farms.

In scenario 2, a farmer can pay other farmers to implement his reduction requirements. The price is determined as the implicit tax rate necessary to reduce the aggregate herbicide TFI to 0.62. The actual design and implementation of such a scheme, however, is considered to be beyond the scope of this analysis. Thus, in the following sections the economic effects do not include any associated administration or transaction costs.

One indicator for the costs of herbicide reduction is the loss of agricultural gross margin per unit reduction in standard treatment doses (SD) at the farm level. This indicator represents the average value of the herbicide use foregone due to the regulation (a unit cost), and is thus an approximate[16] measure for the farmers' willingness-to-pay for the right to use an additional standard treatment dose. The measure can be compared across farms, and hence indicates the economic potential for trade.

Figure 4.6 presents this cost indicator for all farms in scenario 1, where reduction requirements are non-transferable. The cost indicator has been related to the size of agricultural area that these farms represent in Denmark.

Figure 4.6. Costs of herbicide reduction as measured by agricultural gross margin on different areas, DKK/standard treatment dose reduction. Scenario 1



Source: ESMERALDA

[16] The theoretically most meaningful measure would be the marginal cost in the point of expansion (the regulated level), whereas the current measure represents an average cost of the reduction from the initial to the regulated level.

For around 15 per cent of the total agricultural area, there are no costs due to the regulation, because these farms apply less herbicide than allowed. For some 45 per cent of the area, the unit cost lies between zero and 500 DKK per hectare, for 25 per cent of the area, the unit cost lies between 500 and 1000 DKK per hectare and for the remaining 15 per cent of the area, the unit cost lies above 1000 DKK per hectare. Hence, there is a substantial variation in the average cost per reduction unit across farms.

In order to estimate the full costs of the herbicide reductions outlined in the two scenarios, another cost measure than the agricultural gross margin is used below. The reason for the choice of a total cost indicator is that a substantial proportion of the input factor substitution occurs between variable factors and labour. As labour is not included in the agricultural gross margin measure, the economic consequences of these substitutions are not properly reflected in the gross margin measure.

Table 4.10 shows the distribution of per hectare total costs of the reduction requirements in the two scenarios along with the differences in costs between the scenarios. Costs in scenario 2 include net payments to other farmers for taking on reduction requirements. The differences represent the reductions in costs that arise from achieving a given level of herbicide reduction by means of transferable reduction requirements instead of restrictions on individual farms.

Table 4.10. Total costs of herbicide reduction requirements by farm types, DKK per hectare

Farm type	Total costs per hectare (DKK/ha)		Difference	
	Scenario 1	Scenario 2	DKK/ha	Per cent
Crop, loam	943	740	204	22
Crop, sand	892	770	122	14
Cattle, loam	250	29	222	89
Cattle, sand	753	273	480	64
Pig, loam	1293	1278	15	1
Pig, sand	853	821	32	4
Part-time, loam	454	380	74	16
Part-time, sand	538	280	259	48
All farms	757	535	222	29

Source: ESMERALDA
Note: Due to rounding, columns may not add up to totals

On average, a 62 per cent reduction in herbicide use equivalent to a 39 per cent decrease in total pesticide use entails total costs of 757 and 535 DKK per hectare for scenarios 1 and 2, respectively. Thus, there is a potential gain of 222 DKK per hectare or 29 per cent in relative terms from using transferable as compared with individual reduction requirements.

There are large variations between farm types as to the total costs per hectare. In both scenarios, crop and pig farms incur the highest per hectare costs, while cattle and part-time farms incur the lowest total costs per hectare. For scenario 1, this corresponds well with the magnitude of total pesticide reductions on these farm types as shown in table 4.4, i.e. farms that experience the largest reductions in pesticide use incur the highest costs. The largest differences between the two scenarios – and therefore the largest potentials for trade – are found on cattle and crop farms as well as on part-time farms on sandy soils.

Table 4.11 presents national estimates of the total costs imposed by the regulation under each scenario. The total costs are calculated as per hectare costs depicted in table 4.10 timesthe total acreage of each farm type.

Table 4.11. Total costs of reductions in herbicide use distributed by farm types, differences in costs between the scenarios and distribution of gains from transferring reduction requirements

Farm type	Total costs (mio. DKK) Scenario 1	Scenario 2	Net purchase	Net gain	Percentage buyers
Crop, loam	290	215	43	33	74
Crop, sand	238	204	4	30	44
Cattle, loam	31	4	0	28	45
Cattle, sand	500	186	-6	321	33
Pig, loam	264	261	2	2	44
Pig, sand	317	309	-11	18	25
Part-time, loam	113	96	-7	24	23
Part-time, sand	240	135	-23	128	18
All farms	1994	1409	0	585	28

Source: ESMERALDA
Note: Due to rounding, columns may not add up to totals.

According to the table, the total costs of the restrictions in herbicide use is 2 and 1.4 billion DKK in scenarios 1 and 2, respectively, corresponding to a potential gain of almost 0.6 billion DKK from introducing transferable reduction requirements and not restricting herbicide use on individual farms. In scenario 2, the costs are divided into loss of production-related profits and net purchase of herbicide use permits. For example, crop farms face a cost of paying other farmers to take on reduction requirements (positive net purchase), whereas part-time farmers receive net revenues in terms of payments from other farmers (negative net purchase). The difference in net costs between the two scenarios is given in the 'net gain' column for the respective farm types. For all farm types, the costs of reductions are lower in scenario 2 than in scenario 1, as would be expected. Apart from crop farms, pig farms on loam are net buyers of herbicide use permits, while livestock farms on sandy soil and part time farms are net sellers, on average.

As a lot of variation in production structure etc. exists between the farm types as well as within the individual groupings, not all farms within a group can be expected to be either net buyers or net sellers. The final column of the table shows the share of farms, which are net buyers within the respective farm groups in scenario 2. Farms exist within each group (e.g. 26 per cent of crop farms on loamy soil) that commit themselves to reducing herbicide use above and beyond the reductions mandated by scenario 1 and thus incur increased costs relative to that scenario. At the same time, other farms commit themselves to reducing herbicide use less than mandated by scenario 1 (e.g. 74 per cent of crop farms on loamy soil), and thus benefit from the transferable reduction requirements. The group of crop farms on loamy soil uses 43 million DKK for buying permits from other crop farms but also from other farm types. The net gain from the transferability of reduction requirements to the crop farms on loamy soils is thus 33 million DKK.

A large proportion of pig farms and part-time farms on sandy soils commit themselves to additional herbicide reductions and thus incur relatively large additional costs compared to scenario 1. For these types of farms, the gain takes the form of payments from other farms. On the other hand, a large share of crop farms on loamy soils obtain a net gain by paying other farmers to reduce herbicide application, because their marginal profitability of herbicide use exceeds this payment.

4.5. Discussion

The preceding section has illustrated the effects of a reduction in herbicide use in line with the targets of the Pesticide Action Plan, by analysing the environmental and economic consequences of two different scenarios. In scenario 1, an aggregate reduction in herbicide treatment of 62 per cent use is achieved by restricting herbicide use on individual farms, and in scenario 2 an equivalent aggregate reduction is implemented through a scheme of transferable reduction requirements. By restricting herbicide use, the aggregate use of all pesticides in both scenarios is reduced by the same proportion – 39 percent – compared to the base situation. The scenarios are thus similar with respect to their overall objectives, and the economic consequences of the two regulations may be compared.

The total costs of the regulation are roughly 1,4 billion DKK in scenario 2 and 2 billion DKK in scenario 1. This implies that the equivalent aggregate reduction in herbicide and pesticide use can be achieved some 600 million DKK cheaper by implementing a system of transferable reduction requirements compared to regulating individual farms. This corresponds to an average of 222 DKK per hectare of agricultural land[17].

Environmental effects

Even though the aggregate reductions in herbicide and pesticide use in the scenarios are equivalent, the actual effects of the two regulations on the environment are not necessarily identical. Consider the geographical distribution of pesticide use. The individual regulation results in a reduction of herbicide use that is uniformly spread across the entire country, whereas the transferable reduction requirements lead to an uneven distribution pattern, i.e. herbicide use is concentrated in some areas. Transferability of herbicide reductions can thus lead to an increase in the use of herbicides in e.g. areas with valuable drinking water reserves, and in other places where a decrease in the use of herbicides is warranted. However, no unequivocal conclusions can be drawn within this framework as to which situation is preferable from an environmental point of view.

Similarly, the reduction in herbicide use is accompanied by an increase in the use of other pesticides. If the actual distribution of these pesticides differ in the two scenarios, an unambiguous ranking with respect to environmental impacts of the scenarios compared to the base situation may not be possible. However, as the use of insecticides increases more in scenario 1 than in scenario 2 and other pesticide types increase by the same amount in the two scenarios – cf. table 4.2 – it is considered safe

[17] For comparison, the GFI of the Danish agricultural sector was 29 billion DKK in 1997 (Danmarks Statistik 1998: Landbrugsstatistik 1997), corresponding to around 11,000 DKK per hectare.

to conclude that scenario 2 on aggregate impacts less on the environment than scenario 1 in this respect. The spatial distribution patterns of pesticide types other than herbicides have not been the subject of any further specific analysis, but the general observations regarding the geographical variations mentioned in the previous paragraph of course apply here too.

Apart from the changes in the use and distribution of pesticides, the regulation implied by the scenarios also affect the use of nitrogen fertilisers and the soil surface nitrogen balance. The aggregate nitrogen surplus increases by some 4 and 2 per cent in scenarios 1 and 2, respectively. Comparing this to a 39 per cent reduction in pesticide use, the observed cross effects with reduced pesticide use and increased nitrogen surplus seem modest. However, as there is no common yardstick for comparing environmental impacts from pesticides and nitrogen, it is not possible to conclude if the scenarios actually represent an improvement in environmental performance for agriculture compared to the base situation.

If changes in the acreage of grasslands and fallow are accepted as a rough indicator of the variations in plant and wildlife habitats, i.e. that an increase in extensively farmed areas leads to an environmental improvement with respect to biodiversity, the two scenarios do not seem to offer much in this respect. According to table 4.5, the proportion of grassland and fallow is virtually identical in the base situation and in the two regulation scenarios. However, the aggregate land use does not reveal any changes in the location of extensively farmed areas that could have some effect with respect to biodiversity and nature values. From this point of view, the results may however be somewhat pessimistic, as the applied model framework predominantly focuses on the productive uses of agricultural land, with less emphasis on idling of land.

Comparison with other studies

In the following, the results of this analysis are compared with other studies describing the economic costs of a reduction in the use of pesticides in Denmark.

Ørum (1999) obtains national estimates of the costs of different scenarios for pesticide reduction by scaling up results from a micro economic – i.e. farm management level – model. Farm profit losses[18] are calculated for three scenarios of partial or complete bans on pesticide use: a 28 per cent reduction entailing a decrease of 188 million DKK in profits, a 77 per cent reduction yielding a 1,314 million DKK loss, and a complete ban on pesticides reducing profits by 2,514 million DKK. The costs determined by Ørum (1999) are significantly lower than the findings in the present study. These differences can be attributed to differences in the modelling of production technology (implying that some of the reduction in pesticide use in Ørum's study is assumed to be undertaken without costs) as well as the extent of the data base of the calculations. On this background, the results of Ørum are not considered incompatible with the findings of this study.

Jacobsen & Frandsen (1999) analyse the economy-wide effects of a complete or partial ban in the use of pesticides in Denmark using a computable General Equilibrium (CGE) model. In this study, a complete ban on pesticides entails a reduction in Gross Factor Income (GFI) of 3,834 million DKK in 1992 prices. A partial ban – corresponding to a reduction in pesticide use of 81 per cent – entails a reduction in GFI of 1,955 million DKK. These results depict the long term economic consequences of bans on pesticide use, while the assumed flexibility and possibilities for structural adaptation to a change in pesticide policies are more limited in the present study. For example, the regulation analysed in the present study solely considers herbicide reduction, whereas a general objective of reducing total pesticide use leaves the individual farmers with several options of achieving a similar reduction in aggregate pesticide use. Ceteris paribus, this higher flexibility in the analysis by Jacobsen & Frandsen should also be expected to imply lower costs. Furthermore, the farm structure is assumed locked in the present study, whereas cost-reducing adjustments in the farm structure are possibly in the analysis by Jacobsen & Frandsen analysis. Moreover, Jacobsen & Frandsen (1999) incorporate price effects, e.g. higher product prices that could alleviate the costs somewhat. However, as input prices (e.g. labour) can increase as well, the total effect of incorporating price effects are ambiguous.

In a study based on model calculations of different schemes of crop rotation estimating the loss of profits (Gross Margin II) due to conversion from conventional agriculture to pesticide free management in water catchments, Dubgaard & Mortensen (2000) found that the reductions in profits for standard cropping was around 1000 DKK per hectare on loamy soils and almost 650 DKK per hectare for sandy soils.

[18] In the study, profit is defined as the gross margin minus labour and machinery costs in the crop production.

Economic losses for cattle farms were found higher on loamy soils and lower on sandy soils. Even keeping in mind that the figures reflect the costs of a total phase-out of pesticide use, they seem comparable with the results of the present report.

As indicated in the discussion above, caution should be exercised in a direct comparison of the results of the different studies, as e.g. the definitions of cost measures are not identical, there are differences in the underlying focus and time horizons of the analyses, and furthermore the analyses represent different years. It is, however, possible to infer that the results of this study are of the approximate same magnitude as in the other studies, although they seem to be in the relatively high end of the range.

Finally, it could be noted that in a subsidy scheme for agriculture in particularly environmentally vulnerable areas[19], a compensation of 600 to 700 DKK per hectare is given to farmers who refrain from using pesticides. This is also lower than the results of the present study suggest. It should be noted, however, that this scheme primarily target areas of low productivity and that the results of the present report reflect average productive soils.

In sum, the cost results of the present study seem to be quite high compared to the findings of other studies on the costs of pesticide reduction. This is probably due to the flexibility constraints and lack of freedom in adapting to severe restrictions on pesticide use that is implied by the model concept of this study.

[19] The so-called MVJ scheme for environmentally friendly agricultural practices.

5. Summary and conclusions

This report describes and demonstrates a novel model concept for analysing the environmental and economic effects of different agri-environmental policy scenarios. The model concept brings together farm specific economic and environmental data on a common platform and attaches economic behavioural parameters to these data. The concept is based on the agricultural accounts database of FOI comprising 1500-2000 sample farms for which annual structural and economic data is compiled. The concept facilitates integrated economic-environmental quantitative assessments of agri-environmental policy instruments targeting e.g. the use of nitrogen and pesticides, including regional and distributional effects within the agricultural sector.

In the report, the integrated model concept is applied to two scenarios of herbicide reduction. The target level of herbicides of the first Pesticide Action Plan is chosen as the target for two different forms of regulation. In the first scenario, the required reduction in herbicide use is implemented at the individual farm level, whereas in the second scenario, a system of transferable reduction requirements is used to achieve the same aggregate reduction in herbicide use. The focus of the analysis is on the economic and environmental consequences of the two scenarios. The actual design, implementation and administration of the regulation associated with the two scenarios are not considered in the report.

Although the reduction in the use of herbicides of 62 per cent is accompanied by slight increases in the use of other pesticides in both scenarios, the regulation implies a considerable and almost identical reduction of 39 per cent in total pesticide use in both scenarios. However, the geographical distribution of the herbicide use varies considerably between the scenarios. In scenario 1, the reduction in herbicide use is uniformly distributed across the country as implied by the regulation. There is a tendency in scenario 2 for reductions to be concentrated in regions that in the base situation have the lowest application rates, whereas reductions in herbicide use are smaller in regions that initially have the highest herbicide use. Thus, regulation by a system of transferable reduction requirements without geographical specifications or limitations tends to concentrate the use of pesticides in some areas, and results in a possibly undesired ‘polarisation effect’.

With respect to other environmental indicators analysed in the report, the scenarios are very similar to each other on the aggregate level. The regulations implied by the scenarios lead to increases in the acreage of cereals and decreasing seed and root crop areas, but the proportion of grassland and fallow is virtually identical in the base situation and in the two scenarios. If changes in the acreage of grasslands and fallow is accepted as a rough indicator of the variations in plant and wildlife habitats, i.e. that an increase in extensively farmed areas leads to an environmental improvement with respect to biodiversity, the two scenarios do not seem to offer much in this respect. This effect may however be under-estimated as the spatial distribution may play a role in this respect.

With respect to fertiliser use, the reductions in herbicide use are followed by a slight increase in nitrogen surplus that is somewhat smaller in scenario 2 than in scenario 1. Generally, the highest nitrogen surpluses before and after regulation are observed in the Western parts of Denmark while the highest relative changes due to regulation are observed in the Eastern parts of Denmark. Still, the percentage changes in the two scenarios are very similar.

The costs associated with the regulations are very different in the two scenarios. The reduction in herbicide use is achieved at a total cost per hectare of DKK 757 and 535 in scenarios 1 and 2, respectively. Accordingly, 29 per cent of the costs associated with the herbicide reductions can be avoided by the introduction of transferable reduction requirements in contrast to regulation at the individual farm level. On the national level, these figures translate into total costs of 2 and 1,4 billion Danish kroner for scenarios 1 and 2, respectively. The potential gain from the introduction of transferable reduction requirements (as opposed to individual regulation), amounts to around 2 per cent of the annual agricultural GFI.

The gains from transferability are due to the fact that total costs per hectare vary significantly between farm types. In both scenarios, crop and pig farms incur the highest per hectare costs, while cattle and part-time farms incur the lowest total costs per hectare. The largest differences between the two scenarios – and therefore the largest potential for trade – are found on cattle and crop farms as well as on part-time farms on sandy soils. Pig farms and part-time farms on loamy soils are only slightly affected by the transferability of reduction requirements. If a system of transferable reduction requirements is introduced, farms with high reduction costs will pay ‘low cost’ farms to reduce their pesticide use.

The cost analysis exclusively reflect the total costs to the primary agricultural holdings, not taking into account structural adaptations of farms, nor adjustments in prices on inputs as well as outputs. Furthermore, the costs of administration and enforcement etc. are not included in the analysis. The cost estimates from this study are relatively high compared to other studies concerning the economic implications of pesticide reductions. This is partly explained by the fact that costs in this study are based on actually observed figures and not hypothetical model data, and that the possibilities of structural adaptation in this study are limited to changes in the crop composition and animal husbandry on the individual farms.

Many of the target variables in Danish environmental policy (e.g. the treatment frequency indices, total nitrogen leaching, etc.) are stated at a national level. The analysis in the present report illustrates that if this is the case, the highest cost effectiveness will be obtained if the regulation is flexible, as is the case with the considered transferable reduction requirements as opposed to individual requirements. On the other hand, the analysis also shows that due to geographic differences in farm structure, some regional variation in the degree to which the stated target is met, if unlimited transferability exists. This result implies that the environmental problem may still be unsolved in some regions even though the aggregate target is met. This should have implications for the formulation of relevant environmental policy target variables.

Focusing on the relative differences between the two scenarios, the analysis shows that considerable efficiency gains in the form of reduced costs of compliance can be achieved by the introduction of transferable reduction requirements as opposed to a uniform regulation. However, as the implementation of transferable reduction requirements yields an uneven geographical distribution of herbicide use compared to the case of regulating individual farms, the results suggest that the setting of national environmental targets should be accompanied by spatial distribution objectives.

As previously mentioned, the report demonstrates a novel quantitative analytical framework for integrated economic and environmental analyses related to agriculture. The consistency, flexibility and versatility of the concept render it useful for a multitude of analyses of different scenarios of environmental and agricultural economic policy regulations. Especially the cross-effects between different policies, e.g. the observation in this report that a reduction in the use of herbicides is followed by an increase in nitrogen surplus – an observation which points out the need to consider nutrient- and pesticide-related environmental problems as integrated problems. In other words, the concept is a powerful tool for integrated agri-environmental analysis that can be enhanced in several ways to expand the applications to policy analysis.

The scope for analysis within the current analytical framework may be expanded to a number of issues related to pesticide and nitrogen use in agriculture. Furthermore, due to its' farm-level basis and explicit representation of farm heterogeneity, the framework is useful for analysing a range of potential regulation instruments, including instruments addressing the farm level. For example, the model has been utilised in an integrated model concept with a regional economic model to analyse the agricultural and economy-wide economic effects of changes in the livestock density requirements on a regional and local level (Hasler et al., 2002).

At present, environmental data are based on survey data and records from roughly ¼ and ¾ of the sample farms for pesticides and nitrogen, respectively. The integrated concept can be expanded to include other environmental parameters, e.g. phosphorus, and the economic and environmental data can continuously be updated. Future developments may also include an expansion of the empirical foundation for this data and thus elimination of some of the associated uncertainty. In addition, the scaling up of results from the level of the sample farms is continuously improved along with the estimates of the behavioural parameters – e.g. elasticities – that govern the farmers' adaptations to changes in regulations. Development of the concept may also include GIS-aided presentation of the results providing a better representation of the spatial consequences of different regulations. Furthermore, linking the model concept to hydrological or geological data may improve the analysis of the environmental effects.

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Appendix 1

Details of the approximation procedure

The main steps of the approximation procedure are shown in box A1.1. This appendix explains each step in terms of equations.

Box A1.1. Main steps in the approximation procedure

1. Establish structural data for all FOI farms (accounts database).
2. Establish structural and factor input use data for survey farms (merging databases).
3. Using 2, select structural variables with their variable-weights.
4. Using 1, select 10 survey farms and their farm-weights for each remaining FOI farm.
5. Adjust the farm-weights to match real and estimated farm area.
6. Use the adjusted farm-weights to estimated input factor use on each FOI farm.

Step 1

A farm, *f*, in the FOI database (*n* FOI farms) has a production structure that can be expressed with a vector of *m* structural variables:

$$Z^{f} = \{Z_1^{f}, \ldots\ldots, Z_m^{f}\} \quad \forall\, f = 1 \ldots n \tag{1}$$

Each variable in Z^{f} is normalised between its minimum and maximum values to lie between 0 and 1. This eliminates any scaling problems when values are aggregated in calculations below.

Step 2

In addition to these structural data, each of the *t* survey farms have factor input data X^{f} for up to *s* variables. Note that each survey farm is only a part of the FOI database, such that input data must be estimated for the remaining FOI farms (i.e. *n-t* farms):

$$X^{f} = \{X_1^{f}, \ldots\ldots, X_s^{f}\} \quad \forall\, f = 1 \ldots t < n \tag{2}$$

Step 3

To choose structural variables for the approximation procedure, correlation coefficients are calculated between structural and input data for *t* survey farms:

$$\mathrm{cor}_{X\,Zj} = \frac{\frac{1}{t}\sum_{f=1}^{t}(X^f - \mu_X)(Z_j^f - \mu_{Zj})}{\sigma_X \sigma_{Zj}} \quad \forall X;\ j = 1 \ldots m \tag{3}$$

where μ_{Zj} and μ_X are the survey farms' averages of structural and input use variables, while σ_{Zj} and σ_X are the corresponding standard deviations.

The structural variables with the highest correlation coefficients are chosen for the approximation. To reduce computational time, the number of variables is limited to *u*. Their initial weights are calculated according to their share of the aggregated correlation:

$$\gamma_j = \frac{\mathrm{cor}_{X\,Zj}}{\sum_{j=1}^{u}\mathrm{cor}_{X\,Zj}} \quad \forall\ j = 1 \ldots u \prec m \tag{4}$$

As such, a high correlation coefficient gives a higher weight. These variable weights are adjusted according to a procedure, which is better explained after the following details of the weighted average approach.

Step 4

The difference between a survey farm, *f*, and one of the remaining FOI farms, *i*, is measured as:

$$\eta^{fi} = \sum_{j=1}^{m}\left[\frac{Z_j^f - Z_j^i}{Z_j^i}\right]^2 \quad \forall\ f \neq i \tag{5}$$

However, the total difference is weighted such that structural variables with low correlation coefficients to the input factor, count less in the difference measure:

$$\eta^{fi} = \sum_{j=1}^{m}\gamma_j\left[\frac{Z_j^f - Z_j^i}{Z_j^i}\right]^2 \quad \forall\ f \neq i \tag{6}$$

For each FOI farm k survey farms are chosen. The farm weights are calculated as:

$$v^{fi} = \frac{1/\eta^{fi}}{\sum_{i=1}^{k} 1/\eta^{fi}} \quad \forall f \neq i \tag{7}$$

Thus, a smaller difference between two farms leads to a higher weight.

Step 5

Data are estimated for each of the remaining *n-t* FOI farms using the weighting system for structural and input use data, respectively:

$$\hat{Z}^{f} = \sum_{i=1}^{k} v^{fi} Z^{i} \tag{8a}$$

$$\hat{X}^{f} = \sum_{i=1}^{k} v^{fi} X^{i} \tag{8b}$$

where Z^{i} and X^{i} indicate the structural and input data of the *k* selected survey farms, *i*, representing FOI farm *f*.

Step 6

As expected there is a certain level of estimation error. An adjustment factor, $\hat{Z}^{f} / Z^{f}$, is applied to ensure that an estimated variable is equal to that observed on the farm. The adjustment factor must be a structural variable for which there are observed values for all FOI farms:

$$\hat{Z}^{f}_{adjusted} = \sum_{i=1}^{k} v^{fi} Z^{i} \cdot \frac{\hat{Z}^{f}}{Z^{f}} \tag{9a}$$

$$\hat{X}^{f}_{adjusted} = \sum_{i=1}^{k} v^{fi} X^{i} \cdot \frac{\hat{Z}^{f}}{Z^{f}} \tag{9b}$$

The structural variable is chosen as one that is important to estimate correctly (for example total farm area).

Step 3 revisited

Returning to the variable weights, these are adjusted such that input use data are estimated as closely as possible for all survey farms. Thus, input data for each survey farm is estimated as a weighted average of *k* other survey farms. The aggregate estimation error, ε, of input use, *s*, across all survey farms is minimised:

$$\varepsilon_s = \sum_{f=1}^{t} \left[\frac{\hat{X}^f - X^f}{X^f} \right]^2 \quad \forall f = 1 \ldots t \tag{10}$$

Equations 6, 7, 8b and 10 are used such that the variable weights (γ_j) are adjusted incrementally, while farm weights (v^{fi}), input use ($\hat{X}^f$) and the estimation error (ε) are recalculated for each increment. Because the large range of incremental adjustments increases computational time, only a limited number of structural variables can be used.

Appendix 2

Results of the approximation procedure for the 1997/98 harvest year

To establish a consistent modelling basis this report applies nitrogen data for 1997/98 to estimate input data for the previous year (1996/97) in line with the pesticide data. However, this appendix contains results comparable to those in chapter 2.2, as the 1382 survey farms are used to estimate nitrogen use on remaining FOI farms for the applicable year. Thus, the structural variables and -weights are maintained from chapter 2.2, while the approximation procedure from step 4 is repeated for the remaining 315 farms in 1997/98. Note that 13 farms are excluded, as they have no agricultural area.

Results

Table A2.1 shows soil surface balances for three farm types and scaled-up results indicating the basis situation at a national level. Equivalently to the 1996/97 results, the table shows a higher use of commercial nitrogen per hectare on crop farms, while to-

Table A2.1. Approximated soil surface nitrogen balance for the agricultural sector, 1997/98 harvest year.

	Crop farms kg N/ha	Cattle farms kg N/ha	Pig farms kg N/ha	National [3] 1000 tons N
Nitrogen input through:	172	283	242	591
Commercial nitrogen	120	95	85	271
Animal manure	11	160	122	228
Organic nitrogen other than manure	4	1	1	7
Seed	3	3	3	7
Biological fixation	17	7	14	34
Atmospheric deposition	15	15	15	39
Asymbiotic fixation (micro organisms)	2	2	2	5
Removed nitrogen by:	100	150	102	306
Cash crops [4]	90	33	93	190
Fodder crops	10	117	10	116
Nitrogen surplus: Input – Removed	72	133	139	285

Note: 1. Total agricultural area = 2.6 mill. ha;
2. Excludes trade in manure between farms;
3. Results are scaled-up using the FOI ESE-weights (economic size unit weights);
4. Cash crops include: cereals, pulses, rape, root- and non-food crops for sale. Own production of creals for fodder is also included here.

tal nitrogen use is highest on cattle farms. Together with this, the relatively high level of nitrogen removed with crops on cattle farms, leads to an estimated nitrogen surplus per hectare similar but lower than that on pig farms. Crop farms have the lowest estimated nitrogen surplus. At the national level the average nitrogen loss is estimated to 109 kg N/ha.

The modelling approach is centred on the eight farm types in table 2.2. The estimated nitrogen surplus and areas combine such that full-time pig farms on sandy soil have the highest surplus per hectare, followed by cattle, part-time and crop farms. With one exception, nitrogen surplus per hectare is also higher on sandy soils.

Table A2.2. Estimated nitrogen surplus (soil surface) on eight farm types 1997/98 harvest year

	Nitrogen surplus 1000 tons N	Area 1000 ha	Nitrogen surplus kg N/ha
Crop farms, loamy soil	22	325	69
Crop farms, sandy soil	21	267	77
Cattle farms, loamy soil	16	112	146
Cattle farms, sandy soil	89	642	139
Pig farms, loamy soil	26	221	119
Pig farms, sandy soil	61	406	151
Part-time farms, loamy soil	18	255	71
Part-time farms, sandy soil	31	399	78
Total	285	2626	109

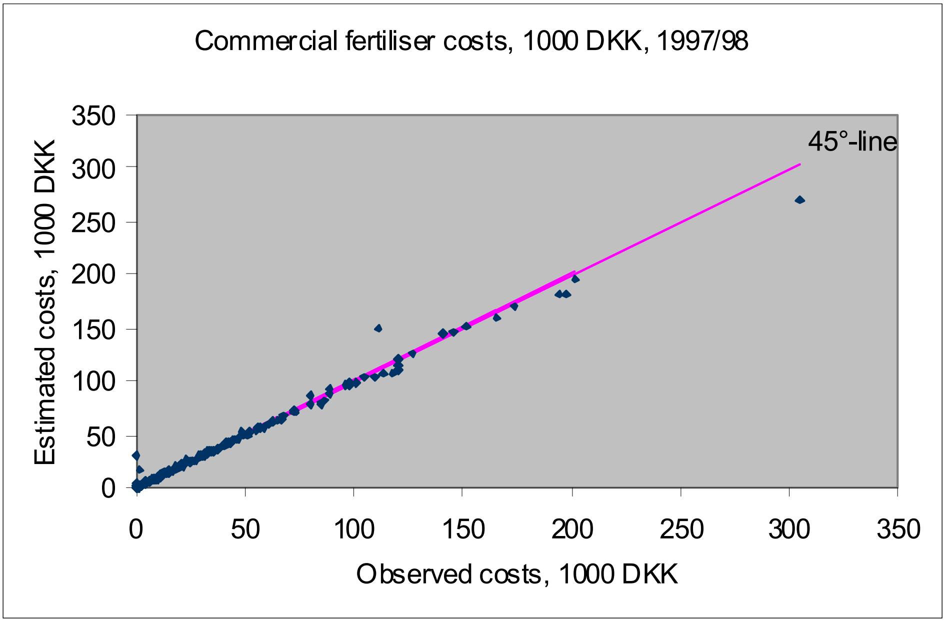
Note: 1. Excludes trade in manure between farms.

Evaluating the results

The approximation procedure used to estimate nitrogen use can be evaluated in several ways. Figure A2.1 shows one way whereby estimated commercial fertiliser costs are plotted against the observed data. The approximation is considered reasonable as most farms lie on or near the 45°-line in the graph.

The approximation can also be evaluated by comparing the scaled-up estimate for total use of commercial nitrogen in Denmark (271 million kg) with official statistics. In this case the estimate, using ESE weights for each FOI farm multiplied by their nitrogen use, is found to be 4 pct. lower than the national sales statistics of 283 million kg (Danmarks Statistik, 1999). It should be noticed that Danmarks Statistik produces statistics on purchased nitrogen, whereas data from the fertiliser accounts show applied nitrogen. Thus, the approximation appears reasonable, also given that storage of fertilisers between years is not accounted for.

Figure A2.1. Estimated versus observed commercial fertiliser costs, 1997/98 harvest year



Estimated nitrogen input through animal manure lies only 3 pct. under figures in Kyllingsbæk et al. (2000). Also compared to this source, current estimates of removed nitrogen from cash crops are overestimated by 1 pct. and fodder crops underestimated by 8 pct. As such, the total nitrogen surplus estimated in the current report (285 thousand tonnes) is 8 pct. lower. Thus, the approximated data of applied commercial nitrogen appears to be within the range of other studies, while the final nitrogen surplus indicator is underestimated.

Appendix 3

Econometric estimation of disaggregated price elasticities

This appendix briefly describes the econometric estimation procedure etc. underlying the behavioural parameters presented in chapter 3. A more comprehensive description can be provided from Jørgensen et al. (2000).

General estimation procedure

The methodological approach underlying the estimation is the so-called dual approach (see e.g. Chambers, 1988). According to duality theory, most relevant characteristics of a production technology can be derived from economic data, provided that producers exhibit economic optimising behaviour (e.g. cost minimisation), and that a cost function satisfying a number of regularity conditions[20] can be estimated. An advantage of the dual approach is that economic data reflect the overall economic optimisation on the farm, and that economic data are in general more accessible than physical data, e.g. farm accounts data. In the standard case, where the necessary data are available, the dual approach is straightforward and its performance is in well documented in the economic literature[21].

However, a major limitation to most existing farm accounts data is that their description of fertiliser and pesticide use is relatively aggregated. Thus, the use of nutrients is typically described in terms of an aggregate fertiliser cost, without distinction between nitrogen, phosphorus and potassium, and similarly with pesticides. In this case, the dual approach is less straightforward. This study attempts to overcome these problems by replacing 'missing data' with restrictions and conditions derived from economic theory. An overview of the estimation strategy is given in figure A2.1.

The overall approach is to estimate demand equations representing the available cost data (i.e. one equation per cost series in the data material, e.g. fertilisers), conditional on disaggregated input price data (e.g. prices of nitrogen, phosphorus, potassium, herbicides, energy, labour, etc.), output data and other quasi-fixed information. This estimation provides detailed price parameters in the demand equations for each (aggregated) input in the overall cost minimisation. Exploiting the symmetry conditions on

[20] including linear homogeneity, concavity and symmetry in prices.
[21] See Jensen (1996) for a survey of such studies.

the cost function, this also implies parameter estimates related to some of the prices in the underlying demand equations for fertiliser and pesticide components.

However, parameters concerning the intra-aggregate effects (e.g. the impact of a nitrogen price change on the use of phosphorus and potassium) cannot be estimated econometrically by this procedure. Hence, the econometrically estimated parameters are supplemented with further information in terms of theoretically based assumptions on the structure of substitution within the groups of fertilisers and pesticides, respectively. Specifically, on the fertiliser side, an assumption of separability between nitrogen and phosphorus/potassium has been imposed. With respect to pesticides nested separability has been assumed, with separability between herbicides and other pesticides, and within the group of other pesticides separability between insecticides and fungicides/growth regulators. In addition to these separability assumptions, quantitative assumptions concerning certain parameters have been imposed. Hence, the elasticity of substitution between phosphorus and potassium, as well as the elasticity of substitution between fungicides and growth regulators, have been assumed zero. Finally, it has been assumed that own-price elasticities of individual input components cannot be positive.

Assuming that farmers exhibit profit maximising behaviour for given activity levels, in addition to the above-mentioned cost minimising behaviour, implies that the marginal cost of raising the yield level in a specific crop sector should equal the output price. From this condition econometrically estimable 'yield supply' equations are derived, depending on output and disaggregated input prices, thus representing the interactions between yield and disaggregated input demands.

Data

The data material used for the econometric estimations comprises three different types of data[22], as illustrated in box A3.1.

[22] One might be surprised that the pesticide survey data, cf. chapter 2, have not been used in the estimation. However, these data are only available for one year, and hence the reactions to price changes cannot be revealed from these data.

Box A3.1. Data sources for estimating behavioural parameters

Farm data:	Anonymous farm accounts from a representative sample of Danish farms. Data concerning farm type, production structure, aggregate input composition.
Period:	1975-95
Source:	Danish Institute of Agricultural and Fisheries Economics: Farm Accounts Statistics database
Price data:	Prices of agricultural inputs – including prices of nitrogen, phosphorus, potassium, herbicides, fungicides, insecticides, growth regulators
Period:	1975-95
SSource:	Statistics Denmark, Danish Institute of Agricultural and Fisheries Economics.
Standard cost data:	Standardised budgeted costs of nitrogen, phosphorus, potassium, herbicides, fungicides, insecticides, growth regulators per hectare in different crop sectors
Period:	1995
Source:	Danish Agricultural Advisory Centre

The *farm data* comprise agricultural accounts from a sample of Danish farms, amounting to 1000-2000 accounts per year. As a large share of farms are represented in the sample in two subsequent years or more, the data material has been organised as a panel data set, thus yielding the opportunity for investigating on-farm changes from one year to the next. Among the data series extracted from the accounts database can be mentioned:

- costs of energy, fertilisers, pesticides, services and purchased feeds at the farm level
- number of working hours and amount of capital
- crop yields
- land allocation and number of animals in different livestock sectors
- various classification variables (Standard Gross Margin, municipality)

The data set is split up in eight subsets – representing the abovementioned different farm types – which allows behavioural parameters to differ between these eight farm types. See Kristensen et al. (1999) for a further description of the farm data material.

The applied *price data* include price indices for energy, labour, aggregated and disaggregated fertilisers and pesticides, services and feeds. Whereas disaggregated data for fertiliser prices were available, estimates of the disaggregated pesticide prices had to be constructed. One challenge in this respect has been the low degree of intertemporal homogeneity within these pesticide sub-groups, and hence the problem of weighting price of different single pesticides.

In transforming econometrically estimated parameters to price elasticities, there is a need for data on the use of different inputs in a given point of evaluation. From the agricultural accounts base, data for input use (in terms of cost shares) have been easily accessible, but as mentioned above, the data for fertilisers and pesticides are too aggregated. For calculating price elasticities concerning sub-components, there has been a need to decompose these aggregate cost shares into the relevant sub-components. This has been done by using *standardised disaggregated unit cost* estimates in different sub-sectors. Hence, for a given production structure, a standardised distribution of the fertiliser cost into nitrogen etc. has been calculated by means of such standardised per-unit costs, and the observed fertiliser cost share from the farm accounts data has been distributed according to this standardised distribution.

Model formulation

As is shown in figure A3.1, econometric estimation has been carried out in two stages – each stage involving a cost minimisation system and a profit maximisation system. In the first stage of estimation, the interrelation between aggregate variables (aggregate cost shares and aggregate prices) has been estimated. In the second stage, the relevant parts of the model from stage one (i.e. those relating to fertilisers and pesticides) are re-estimated in a more disaggregated form, conditional on relevant information from the first estimation stage. There are two attractive features to this two-stage approach. First, the risk of multicollinearity is lower, when the number of price variables in one estimation stage is smaller. Second, it enables the use of a 'core model' (the model from the aggregated stage), which can be supplemented with details/disaggregation in the relevant parts for specific issues. The two-stage approach requires orthogonality between fertiliser and pesticide price on the one hand and other explanatory variables on the other.

The model has been formulated in terms of translog cost functions, with input prices as well as sub-sector activity and yield levels together with other quasi-fixed variables as explanatory variables, i.e.

$$\ln C = \tfrac{1}{2}\sum_{i=1}^{N}\sum_{j=1}^{N}\alpha_{ij}\ln u_i \ln u_j$$

where $u = \{w_1,...,w_n, z_1,...,z_k, y_1,...,y_m, q, e\}$ includes input prices (w), activity levels (z) yield levels (y) and quasi-fixed variables (q). e is the basis of the natural logarithm, i.e. ln(e) = 1. Corresponding input demands, in terms of cost shares for inputs 1,..,n can be derived using Shepard's lemma:

$$S_i = \frac{w_i x_i}{C} = \frac{\partial C}{\partial w_i}\frac{w_i}{C} = \frac{\partial \ln C}{\partial \ln w_i} = \sum_{j=1}^{N} \alpha_{ij} \ln u_j, \quad i = 1,\ldots,n$$

Most of the α-parameters in these demand equations were estimated econometrically, whereas the remaining α-parameters have been determined with the help of theoretically motivated parameter bindings, cf. the above section. This procedure has been carried out for the considered eight different farm types, using panel data techniques on data spanning the period 1975-95. From the demand equations can be derived expressions for own- and cross-price elasticities of inputs

$$\varepsilon_{ij} = \frac{\partial x_i}{\partial w_j} \cdot \frac{w_j}{x_i} = \frac{\alpha_{ij}}{S_i} + S_j - \delta_{ij}$$

where $\delta_{ij} = 1$ if i = j, and zero otherwise. Hence, given estimates of the α-parameters and data for relevant cost shares, these price elasticities can be calculated. As mentioned above, decomposed fertiliser and pesticide cost data are not in general available, but given the standardised cost data, it is possible to provide estimates of the average cost shares – also at the disaggregated level – within each of eight farm groups.

As mentioned above, it is also possible to derive and estimate yield responses to price changes, provided that farmers are profit maximisers, in addition to the above cost minimisation assumption. The profit maximisation assumption implies that the marginal cost of increasing the yield level should equal the product price. This implies

$$S_y = \frac{py}{C} = \frac{\partial C}{\partial y}\frac{y}{C} = \frac{\partial \ln C}{\partial \ln y} = \sum_{j=1}^{N} \alpha_{yj} \ln u_j$$

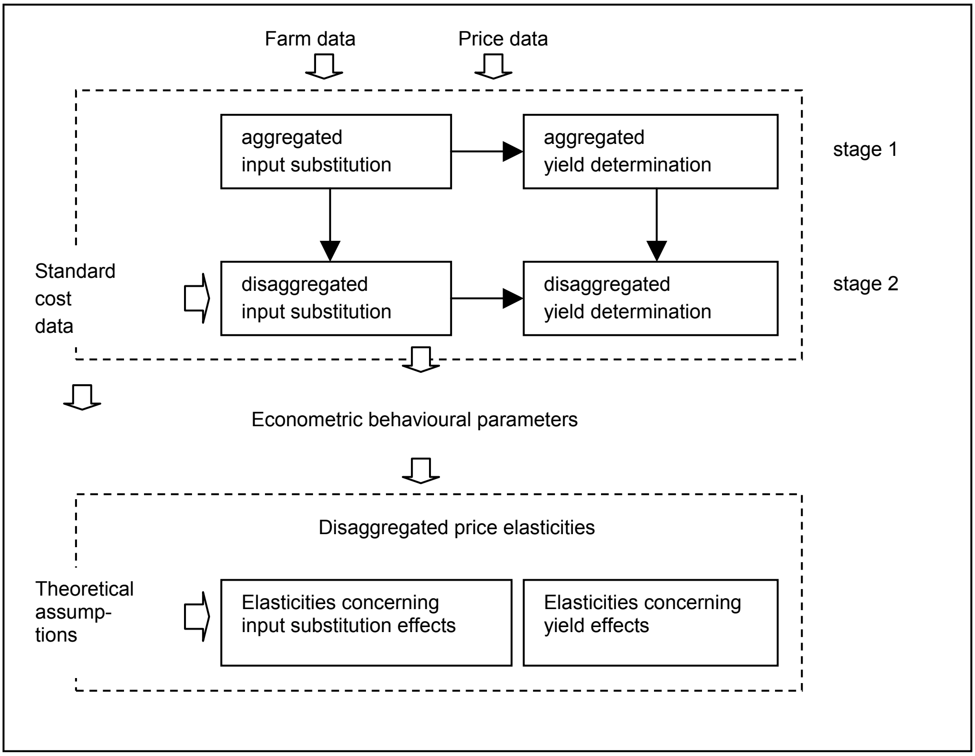
i.e. the ratio between the value of the considered output and total costs can be expressed as a linear function of the explanatory variables in the u-vector, cf. above. Equations of this type are estimated for each crop as a function of (disaggregated) input prices as well as quasi-fixed variables on the above-mentioned data set. From this expression, we can derive the following elasticity expressions:

$$\varepsilon_{iy} = \frac{\partial y}{\partial w_i}\frac{w_i}{y} = \frac{\alpha_{iy} + S_i S_y}{\alpha_{yy} + S_y (S_y - 1)}$$

$$\varepsilon_{yj} = \frac{\partial x_j}{\partial y}\frac{y}{x_j} = \frac{\alpha_{jy} + S_j \cdot S_y}{S_j}$$

Thus, provided econometric estimates of the α_{iy}-parameters, and data for the cost shares and output/cost ratios these price elasticities can be calculated using these expressions.

Figure A3.1. Overview of data and estimation procedure



Appendix 4

Map indicating the regions of Denmark