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# Bio-Economic Models applied to Agricultural Systems

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# Preface

This book has the purpose of providing the "state of the arts" concerning bio-economic modelling dealing with agricultural systems. In most cases, the contributions use a methodology combining the use of biophysical and economic models, in all cases, an engineering production function approach is totally or partially applied.

This practice is being developed in the last years as a response to concrete policy matters: agricultural policies are increasingly combined with environmental and natural resources policies, and this reality involves the need of an integrated assessment, that current economic models are not able to provide. But at the same time this type of approach involves the use of a multidisciplinary approach, extremely difficult to develop taking into account on one side the difficulty of communication between different disciplines and on the other the fact that in terms of scientific evaluation, the existing system is an obstacle to the development of this research orientation, as long as researchers are evaluated on a strictly disciplinary criteria.

**Part I** deals principally with theoretical and methodological issues, as well as a presentation of biophysical models, an important source that provides engineering production functions appropriate to be used by bioeconomic agricultural models.

Chapter 1 discusses the relations between bioeconomic modelling and economic theory. It is clear that all bioeconomic modellers do not share the points of view that are presented in this chapter, the intention is to open discussions that can be fruitful for future research development as well as for reminding the young generations of economists about some old theoretical issues that can be extremely useful for very new practical matters.

Chapter 2 is about production functions; it gives a wide perspective on the matter and provides a strong justification for using biophysical models' outputs as engineering production functions.

Chapter 3 presents a typology of dynamic modelling approaches as well as an application of an innovative method for analyzing the problem of soil degradation by salinization in a small irrigated region of Tunisia. The methodological part of this chapter may be of interest for modellers dealing with this type of problem.

Chapter 4 presents in a quite detailed manner the structure and principal characteristic of biophysical models. Economists willing to use the results of these models need to understand how they are able to represent the complexity of agricultural systems integrating in a single framework the relations between multiple inputs and multiple outputs.

**Part II** presents applications of bioeconomic models analysing different issues at regional levels - both “small” and “big” regions.

Chapter 5 Is an application at the level of a big region (California), of a calibrated agricultural production model presenting an important innovation concerning the use of outputs generated by a biophysical model to infer adoption of a new bio energy crop. As prior information on supply elasticities is not available to calibrate non-linear terms in the objective function, yield variation at the regional level – a piece of information typically available from highly disaggregated biophysical models of plant growth is used to construct such terms.

Chapter 6 presents the use of an Agricultural Model working at European level providing indicators of nitrate pollution obtained by the output of nitrate balances and different innovative indicators of Nitrogen use at a country level for all European Union Countries. The calculation of the N-cycle follows a mass-flow approach. The model keeps track of the nitrogen available at each step – net of all emissions that occurred at an earlier step – and uses this as the basis for the estimation of emissions.

Chapter 7 is an application of a dynamic multiobjective model of animal production (dairy sector) at a specific regional level, Reunion Island. The principal objective is to provide policy assessment. It allows simulating the effects of alternative management practices on Nitrogen emissions. The alternative management that are proposed are (a) and increase of spreadable land area for manure, (b) the transformation of manure to other forms as compost and (c) to utilise manure as a source of energy.

Chapter 8 The aim of this chapter is to provide a better understanding about on-farm risk reducing strategies encompassing both risk anticipation strategy and risk modify the production system and profit distribution of French suckler cow enterprises. The method used in this case is a sequence of recursive discrete stochastic model, close to the method applied on Chap. 4 in a completely different context. The advantage of this approach compared to the standard one of dynamic stochastic programming appears in a very clear way in both cases.

Chapter 9 This chapter presents an application of a bio-economic model to the Lunan Water catchment in Scotland to assess the relative cost-effectiveness of measures against agricultural nitrate pollution. The model used for this work is FFSIM-MP, integrating the outputs of a biophysical model, COUP. This contribution explores the challenges related to bio-economic modelling applications, presents the methodology and results, and evaluates the overall appropriateness of the approach for integrated policy impact assessment.

Chapter 10 integrates three models to provide a spatialised assessment of the relationships between alternative agricultural management and biodiversity. The farm optimization model FAMOS[space], the crop rotation model CropRota, and the bio-physical process model EPIC are used in this contribution. Crop rotations and crop yields are inputs to FAMOS[space], which explicitly considers alternative land use intensities as well as landscape elements. Biodiversity effects of land use choices are evaluated with a set of field and landscape indicators. The specific interest of this chapter is the explicit introduction of the spatial dimension in the bioeconomic model.

A short conclusion presents the principal achievements of this approach and the main obstacles for further development, considering the difficulties of multi-disciplinary approaches, and also the complicated issue of the information needed to use this type of models, even when applied to relatively small study cases.

Guillermo Flichman

# Acknowledgements

I want to gratefully acknowledge all the authors that accepted to give a contribution to this book. It took a lot of time and effort, but the possibility of putting together the work of different researchers developing and applying bio-economic models allows providing a better knowledge about this type of approach.

# Contents

## Part I



## Part II

# Part I

# Chapter 1
# Modelling the Relationship Between Agriculture and the Environment Using Bio-Economic Models: Some Conceptual Issues

**G. Flichman, K. Louhichi, and J.M. Boisson**

## 1 Introduction

In the last years there has been a significant development of bio-economic models, especially those integrating biophysical models and economic mathematical programming models. This development was enhanced by the conjunction of several factors such as the multiplicity of objectives in new agricultural policies, the increase of demand for multi-disciplinary approaches for integrated assessment, and the call for more dialogue and cooperation between scientists from various disciplines. Even though an important number of bio-economic models have been developed and tested on different farming systems and under various agro-ecological conditions (Flichman and Jacquet 2003; Janssen and van Ittersum 2007), there is a lack of literature regarding the implicit or explicit assumptions of these models and economic theory, their main advantages compared to conventional economic approaches, and their specific contributions in strengthening collaboration and improving integration between different disciplines. The aim of this paper is to clarify these conceptual and theoretical issues related to bio-economic modelling and to propose a consistent way of applying this approach for modelling the relationships between agriculture and the environment.

A bio-economic model is generally known as a linkage between models from different disciplines to provide multi-disciplinary and multi-scales answers to a

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given problem. In reality, the philosophy behind this approach is more complicated. The development and implementation of farm bio-economic mathematical programming models requires a good understanding of the relevant theoretical issues and a depiction of the main specifications required for ensuring a consistent integration between the components of coupled agronomic and economic models. A bio-economic model should not be a simple link between models through an exchange of information but a real integration in both conceptual and technical terms. This has twofold implications: first, we are facing a new approach, which should have a clear position in the economic and agronomic scientific corpus, and second, the construction of each model should take into account the specificity and the conceptual basis of the other.

For analysing the relation between agriculture and environment, economic theory has summoned up several approaches: the application of the standard microeconomic analysis, the integration of original methods and tools based on the agent's revealed preferences in the conventional theoretical corpus, or the exploration of new methodologies and knowledge stemming from other disciplines in particular from Natural Sciences (i.e. "Ecological Economics", see Costanza and Daly 1987). From this classification it appears at first sight that the bio-economic modelling method can be classified under the *Ecological Economics* approach, as it aims integrating economics with natural sciences, using as well physical and monetary values. However, as these models are often based on optimisation methods, they could also be situated under conventional economic theory approaches.[1] Regarding integration, the bio-economic farm models should have a set of specifications ensuring a consistent integration with agronomic models. We deal with this issue in the following Sections.

In the Sect. 1 we discuss the use of a primal based approach for the representation of technology, Sect. 2 deals with the issue of activity based modelling, in Sect. 3, we develop a suitable way to specify production and cost functions in bio-economic models, and in Sect. 4, the modelling of environmental externalities is described and demonstrated through an empirical example. Section 5 discusses and concludes.

## 2 Primal Approach of Technology

There are two ways of representing the scope of potential techniques in an economic model.

- Represent the production process, taking into account the physical quantities of inputs needed to produce one unit of output (or used per unit of a fixed resource as land, in the case of agriculture).

[1] The arguments behind these theses are discussed in more detail in Louhichi et al. (2007).

- Represent the production process through the production costs, using in this case a monetary measurement of inputs.

In the first case, it implies the use of an engineering production function approach, making technology representation explicit (kg of fertilizer/ha, $m^3$ of water for irrigation, etc.). This approach allows for switching between production processes defined in a transparent way (Flichman and Jacquet 2003; Janssen and van Ittersum 2007).

These engineering production functions constitute the essential link between bio-physical and economic models. The reasons justifying a primal representation of technology are very clear: with these models we have to deal simultaneously with bio-physic and economic systems and we need to quantify physical variables, as well in the inputs of the model as in the outputs, such as the level of nitrate pollution, soil erosion, etc.

The use of engineering production functions creates a strong information demand. It is necessary to have data about these engineering processes in terms if physical input– output matrices.

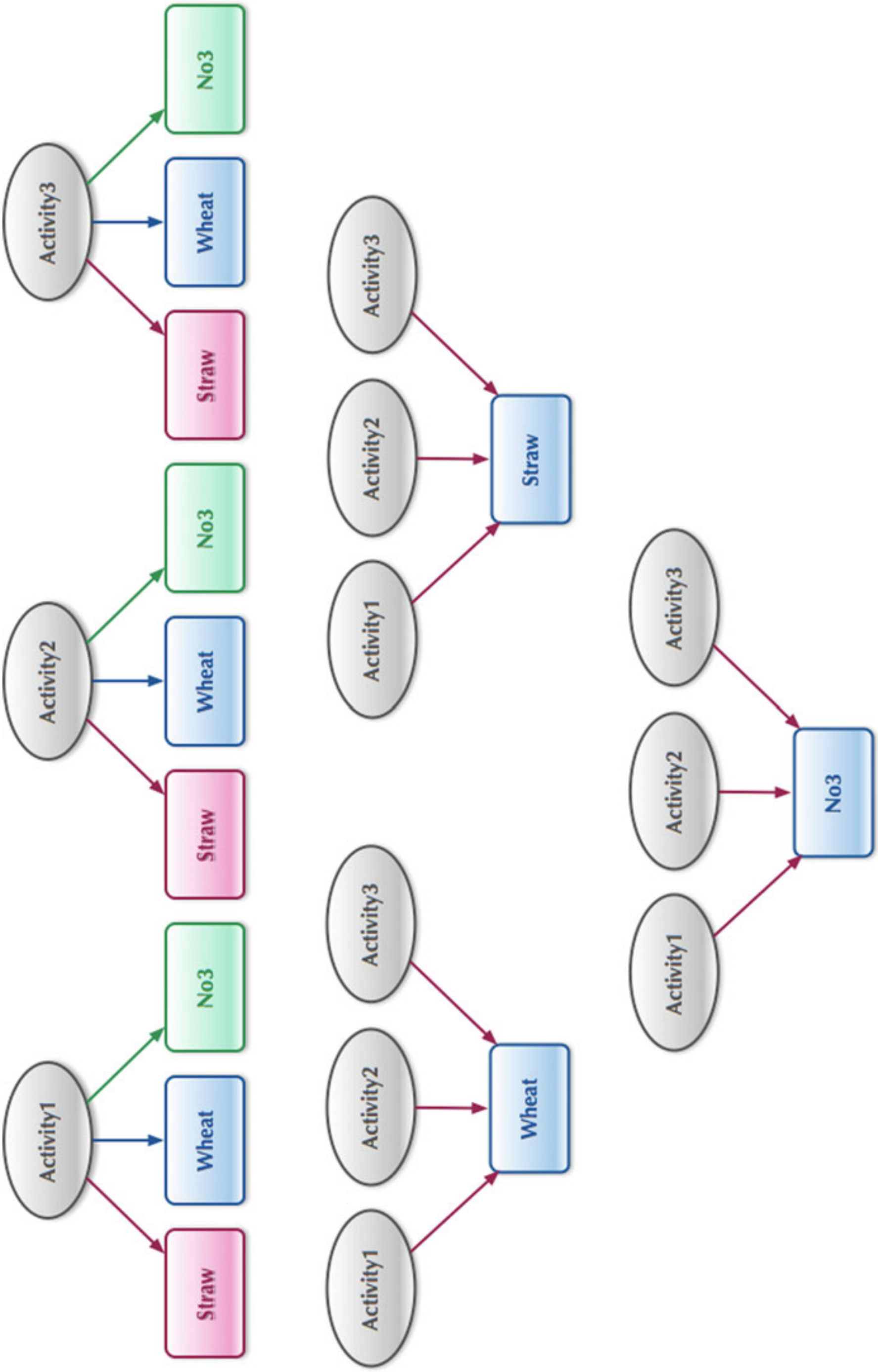
## 3 Activities and Products

The basic element of this approach is the production process, or the production activity, not the economic good (or product). In other words, and using an example from agricultural production, a unit of wheat grain is not the basic element, but the production process that allows obtaining a unit of wheat grain is the basic unit. A production activity describes a specific production process.

Each product can be produced by several activities, and each activity can produce several products. One activity is defined by the technical coefficients that represent the use of inputs needed to produce different outputs. In agricultural models, frequently these technical coefficients relate to one unit of the fixed factor (land) rather than to one unit of product. Koopmans already developed this approach many years ago:

> This method, which precludes the separate measurement of alternative processes to produce the same commodity, or the recognition of joint production, can be and is being supplemented by the study of engineering information. (Koopmans 1951)

We will develop in other section the issue of joint production, what we want to clarify at this point is the relationships between activities and products.


Activity1
Activity2
Activity3
Straw
Wheat
No3
Straw
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No3
Straw
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No3
Activity1
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Wheat
Activity1
Activity2
Activity3
Straw
Activity1
Activity2
Activity3
No3

**Diagram 1**

The scheme presented above allows seeing the causal relationships that are implied in this type of model. "Products" are the outputs of production processes that are described by the activities.

What is also important to realise is that this type of representation has two "faces":

– One activity (or production process) has several outputs – joint production
– One product is produced by several activities (or production processes)

Thanks to this representation we can take into account the positive and negative *jointness* (see Baumgärtner et al. 2001) associated to the production process, and to assess in an integrated manner new policies which are mainly linked to activities and not to products.

The inputs and the outputs (including externalities) in bio-economic modeling can be represented in *discrete forms:* the yield and cost functions per product are expressed as discrete functions, in order to make easily the integration with biophysical models and also to ensure that the impact of each input can be assed separately with respect to the others. Indeed, the biophysical model provides a set of multi-inputs and multi-outputs production functions, which are unsuitable to be properly represented through continuous forms. In the example developed in the last section, we will explain the difficulty of assuming continuous functions.

To illustrate these specifications in formal language: Consider **a**, each agricultural activity representing one production process, which produces several outputs and uses several inputs.

- Denote $\mathbf{J} = \{j_1, j_2 \ldots\}$ the set of economic outputs produced by each agricultural activity;
- $\mathbf{O} = \{O_1, O_2 \ldots\}$ the set of environmental outputs (i.e. externalities) produced by each agricultural activity;
- $\supset = \{I_1, I_2 \ldots\}$ the set of inputs applied in the production of agricultural activities.

And, finally, let **Y**, **E** and $\mathbf{F} \in \mathfrak{R}^{txm}$, where $\mathfrak{R}$ is the set of real numbers, represent respectively, the vector of economic outputs produced by each agricultural activity, the vector of environmental outputs associated to each agricultural activity and the vector of inputs used by each agricultural activity. To be more specific, $Y_{a,j,t}$ denotes the amount of the jth economic output produced in the tth year of the agricultural activity *a*, $E_{a,O,t}$ denotes the amount of the Oth environmental output produced in the tth year of the agricultural activity *a* and $F_{a,I,t}$ denotes the amount of the Ith input used in the tth year in production (Fig. 1.1).

All this seems trivial and is indeed trivial, but what is less trivial is the consequence of these specifications in terms of representation of the cost function in the economic programming models and in terms of analysing environmental externalities and natural resources related with agricultural production.

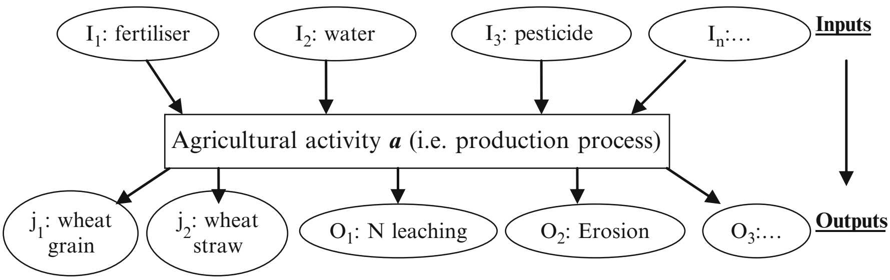



**Fig. 1.1** An agricultural activity as a production process with multi-inputs multi-outputs

## 4 Cost and Production Function in Bio-Economic Models

In bio-economic models, costs are usually defined from engineering surveys and/or statistical information and/or outputs of biophysical models. We always need the physical quantities and the prices in order to provide inputs to these models.[2] Costs should be specified by **activity**, not by **product**. This extremely simple evidence has very important consequences. As usually explained in the first pages microeconomic handbooks (cf. Varian 1992), a set of activities (techniques or production processes) is used to build a continuous production function out of an original discrete function.

An activity-based model representing technology with a primal approach, considers the costs per activity and not per product. Even if the costs of each activity are linear by construction, which implies that the average cost equals the marginal cost, the costs per product are non linear and thus the marginal cost will normally increase if the level of production increases. The reason for this increase – in the case of agriculture – is the presence of a fixed limiting factor; usually land, but it can also be another natural limiting factor such as water.

There are two options for representing cost and production functions of products in a bio-economic model:

- Keep a linear structure of multiple activities for each product
- Estimate a non-linear specification for each product, using the information obtained at an activity level.

There are at least three important reasons advocating the use of a linear structure of multiple activities per product:

[2] Information on costs per activity are not easily found. At European level, such information is available only for costs at farm level in monetary units (Farm Accounting Data Network). This poses a real difficulty in applying these models in Europe, as detailed technical information is required, which is available only at regional level and in some countries at national level.

- The represented activities correspond directly to information originated by engineers' knowledge or biophysical models' information. The activities approach allows a consistent integration of biophysical data in the economic model.
- The "joint products" are conceptually (cause-effects relationships) related to activities, and especially in the case of some agricultural externalities, they are usually proportional to level of the activity and may have quite complicated relations with the level of the product.
- The interpretation of results is straightforward. A certain level of profit, yield corresponds strictly to a level of environmental results as water pollution, erosion level, etc., for a certain mix of activities obtained in the solution of the bio-economic model.

## 5 Environmental Externalities as Joint Products

As it was developed above, each production activity has several outputs. In the simple example presented in the previous section, these products are grain, pollution and straw. All these **products** emerge from one **production activity**. **They are joint products** (Pasinetti 1980; Baumgartner et al. 2001). The relation between a production activity, the main product (from the point of view of the firm) and the joint(s) product(s) is a fatal relation. It is impossible to produce grain without polluting or producing straw. And these relations (production activity → joint products) are thus linear ones.

Adopting this vision, we should not approach the external effect (cost or benefit for other economic agent) as a direct consequence of wheat production; we have to identify what production activity generates this cost to other agents (nitrate pollution) in physical quantities. Doing this, we consider pollution, as an output of the activity that produces both **wheat and pollution as outputs**. This means that for calculating the externality as a cost, we need first to have some knowledge about it as a physical product, and we need to measure it in physical terms (tons of soil erosion, kg of NO3 pollution, etc.). Fortunately, we have access since about 20 years to dynamic biophysical models that simulate the different products related with an agricultural activity (in our case, grain, straw, pollution) within an integrated framework.

This type of representation intends to provide a mechanistic, cause-effect explanation of what is usually called externality.[3] Very frequently we see empirical approaches, trying to find statistical relations between some crop production (considered as "the" product) and some externality, like soil erosion. By construction, even if sometimes it is possible to find elegant functional forms that fit well, **these relationships will always be limited to the specific case from where they**

[3] Mechanistic in the sense of providing the mechanisms through cause-effects relationships, this is the terminology usually employed by biophysical modellers as opposite to empirical. It has not at all a pejorative meaning.

**have been calculated. They are purely empirical**: there is a complete lack of analysis of the processes that connect, for example, grain production with soil erosion. What produces erosion is not the wheat production itself, but the way it is produced, what type of tillage is used, in what period, in connection with the weather, with the type of soil, the previous crop and many other technical issues. In other words it is the process of production, represented by a specific activity. A certain amount of nitrate leaching is not provoked by maize production, but by a certain production activity of which maize grain is one of the outputs (i.e. a wheat-maize rotation with a specific input combination). The relation between a maize non-linear production function and the level of nitrate pollution can be extremely complicated to set up and, if set up, it will not be in a chain of cause-effect relationships (because there is not a direct relation between these two variables), **the empirically obtained function will be applicable only to the specific situation where it was estimated**. Each agricultural technique represented by each production activity is related, in a defined environment (soil-weather) with one value of pollution or erosion, and **there is not any functional form that can be *a priori* applied to represent the relations between two of the joint products, as they are an outcome of extremely complex processes that can be properly represented by fixed technical coefficients relating activities and products**. Of course, it can be possible, out of a post-optimisation exercise at the farm level, to estimate non-linear relationships between different outputs of the model, using parametric procedures. But no functional form should be introduced *a priori* in the optimisation model. The results of simulations done using a biophysical model (BM) can be synthesized in an appropriate way and introduced as linear technical coefficients in a mathematical programming model (MPM). And this procedure can be applied in a dynamic MPM as well as in a comparative-static one. The quality of soil, in terms of its production capacities, changes with the type of use of it over time. This implies that, by essence, this issue should be analysed using a dynamic approach. That is why the biophysical models are perfectly appropriate for doing this.

In brief, modelling the relations between agriculture, natural resources and environment needs to mobilise different type of models and of knowledge. It is difficult to do so and it is also difficult to expose it.

## 6 An Empirical Example of Joint Production Using a Biophysical Model

In order to provide a very simple empirical example, we implemented a simulation using a biophysical model CropSyst (Stockle et al. 2003) for a biannual rotation of wheat and sunflower, applying different amounts of nitrogen and testing the introduction of a catch crop to reduce nitrate pollution. Applying the concepts defined before, we simulate here an important number of production activities, i.e. each rotation with a specific management is one production activity. All the activities produce wheat, sunflower and nitrate pollution (products) and each product is produced by all the activities. We will observe only partial results of these activities, the yields of wheat and the nitrate pollution (in physical units) mapped with the total costs of production.

It is possible to define an efficiency frontier in terms of yields and a different one in terms of nitrate pollution. In a discrete function like the one we use, the conditions for a certain activity to be efficient concerning yields are:

- $y_{ci} > y_{ci-1;}$ if the cost increases, the yield also increases
- $y =$ yields,
- $c =$ costs,
- $i =$ activity
- $(y_{ci-} y_{ci-1}) > (y_{ci+1} - y_{ci})$ ; yield increases when cost increases but at a decreasing rate.

Of course, this is equivalent to the situation of continuous functions, with a first derivative positive and a second negative. **The difference, and it is an important one, is that discrete functions like these do not have intermediate points, only linear combinations of two points that correspond to specific production activities.**

The efficiency in terms of yields is the one that will be taken into account for defining the choice of the producer. The environmental efficiency may be a policy objective, not a producer objective. And we try to develop a positive approach, useful to define policies in order to ameliorate the environmental situation, in this case to reduce the pollution levels.

The efficiency frontier in terms of pollution is the following:

$p_{ci} > p_{ci-1}$ ; if the cost increases, the pollution also increases
$(y_{ci} - y_{ci-1}) > (y_{ci+1} - y_{ci})$ ; pollution increases when cost increases at an increasing rate
$p =$ pollution, $c =$ costs

It is possible to define a frontier taking into account only the activities that are efficient both from the yield and the environmental point of view, but it would be a normative approach, “realistic” only in a perfectly managed collective farm...

But it is still necessary to give an explanation concerning the pollution discrete function, as we gave for the case of soil erosion. There are management techniques that may allow increases in production and costs with decreases in pollution. When we introduce catch crops that use the nitrogen surplus, reducing pollution, they imply an increase of costs but also a higher yield (more organic matter in the soil, better soil structure). We are always in a situation in which a product can be an output of many different activities. This basic issue explains the apparent paradox.

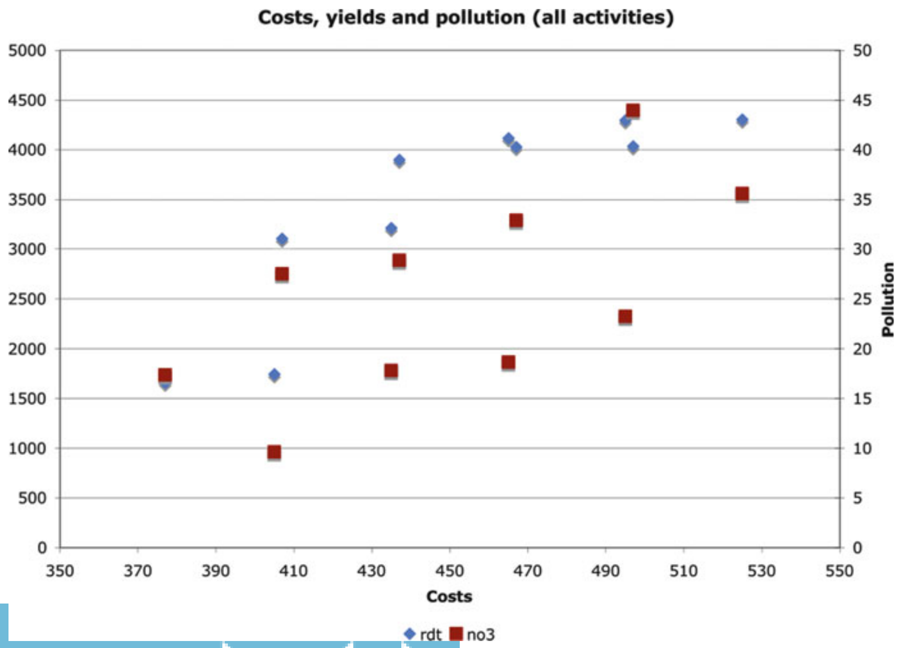
We can see in this example that there are activities on the yield frontier – potentially candidates to be chosen by the farmers – that are inefficient from the environmental point of view. That is why it appears usually necessary to use cross-compliance policies to achieve good results. Taxes or subsidies related directly to production levels or even to polluting inputs may give inappropriate results.

These are the findings when we look at a field level. If we move to the farm level, these “bad behaved” relations appear even more clearly, because there is a mix of activities defining the global result of the farm. It is possible to show how in certain circumstances, a tax on the amount of nitrogen fertilisation may produce an increase in pollution (Flichman and Jacquet 2003).

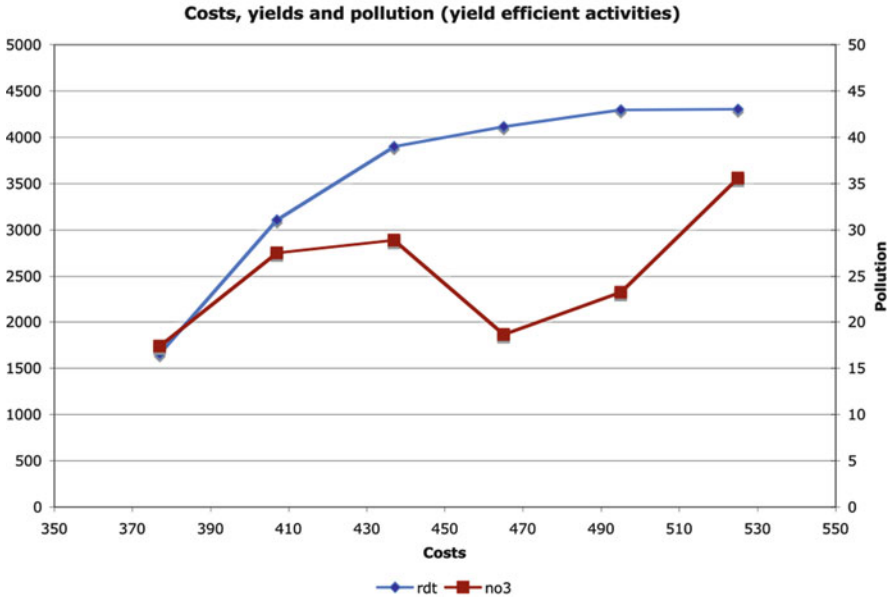
It is necessary to clarify one important issue concerning the definition of a production function in this context. The proper one that applies here is the one from Joan Robinson (Robinson J 1953), “the book of blue prints of possible techniques”. This means that we can include the whole universe of available production processes (activities) at a certain moment in a certain place. It results in a set of points representing all the possibilities of production. A “different production function” can only be legitimately defined if we assume the introduction of technical progress, what we are not doing here. All the activities that are shown in the following example are a part of available techniques at this beginning of the twenty-first century in the South West of France.

Using bio-economic models based on production activities allows calculating cost-efficiency indicators of different policies that intend to induce farmers to adopt a more environmental friendly way of production; policies based on environmental targets based on physical thresholds. And this is the way policies are established in our days.

The following graphs summarise the result of the numerical example. A wheat-sunflower rotation is simulated, with different levels of fertilisation, without irrigation and with and without a catch crop. We present here the wheat yields and the average nitrate pollution, for all the simulated activities in Graph 1 and for yield efficient activities in Graph 2.




**Graph 1**




**Graph 2**

Looking at these graphs, it is possible to see that several points correspond to both frontiers. If we where dealing with a well-managed State Farm, it would be possible to choose between these points. But for a private farmer, in a market economy, what counts is the sole yield frontier. That is why realistic policies tend to influence the production procedures directly, in order to obtain acceptable results in terms both of yield and environment (*Good Agronomic and Environment Conditions of the European policy*). That is the reason why policies affecting only inputs or output prices usually do not appear to be really efficient.

## 7 Conclusion

The use of bio-economic models for analysing agriculture production experimented an important development in recent years. We tried in this chapter to examine the relations between modelling approaches and theoretical orientations. Some conclusions can be obtained out of this analysis.

In terms of different theoretical approaches, this type of models appears close to an Ecological Economics approach: it has an interdisciplinary orientation, but it uses also optimisation methods. Externalities are considered as joint products of production processes, not merely as costs for external agents; economists that belong to the Ecological Economics approach also share this point. From a different point of view, these models are based on a representation of production activities, what is an old orientation that began with the first economists that developed the linear programming techniques, as Koopmans. Some empirical results show in a clear way that it is not always correct to impose “well behaved” functional forms to externalities, that may be non convex in many situations.

In terms of policy implications, also important conclusions can be obtained. When policies intend to influence the production of externalities, it appears very clearly that they should target production processes (or activities), what cannot be done with standard policies of subsidies and taxes to products. In the real world, this is already happening.

We believe that bio-economic models will play an important role for better understanding the relations between agriculture, the environment and the natural resources, as well as for evaluating and formulating policies, but a big obstacle is the availability of appropriate information. Available data exist only at small regional level, if these tools are to be applied at large scale, a big effort in terms of modification of the production of statistics should be undertaken.

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# Chapter 2
# Bio Physical Models as Detailed Engineering Production Functions

**J.M. Boussard**

"A production function purports to exhibit the technical relation between factors of production and output, and to derive from them the relation between their prices, and the relative quantities employed in condition of long run equilibrium". This famous and celebrated definition of the production function by Joan Robinson (1955) clearly put emphasis on the technical character of the concept.

A production function should be designed in such a way as to allow for the determination of the consequences of techniques for the price system and the supply and demand relationships, "in the long run equilibrium" It does not mean that economists have no interest in the short run at all – it is too obvious that the long run being a succession of short runs, anybody interested in the long run development of economic phenomena cannot ignore this aspect. Rather, it means that when designing production functions for analytical purposes, we have to capture the underlying permanent (or relatively permanent) technical relationships, ignoring transitory shocks coming from storage, speculation, or other circumstances which shape price evolutions from one day to another, without reflecting actual production costs and technical relationships.

Indeed, beyond appearance, the above definition conveys an idea of analytical dichotomy: observed prices depend upon intermixed short and long run effects. Disentangling these effects stands among the task of economic analysis. While the technical aspects are most important for the long run, when (hopefully!) prices converge to the minimum feasible average cost, they might be masked by other factors in the short run. To assess the respective importance of each aspect, and isolating its influence from others, it is therefore important to rely on good descriptions of technical relationships. And one does not see who better than an engineer can provide a reliable description of techniques.

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Therefore, only engineers should be qualified to define production functions, perhaps with the help of economists just to indicate how to present results in order for them to be usable in economic models. This precept has frequently been neglected by economists, who often make use of highly simplifying "soft" proxies, such as the widely used "Cobb-Douglas" or "Translog" functions, as representations of "true" production functions. The usefulness of such practices cannot be denied, because it is much easier in large economic models to handle simple mathematical beings. Most of the time, models based on really observed production functions would simply be intractable, and therefore, useless, while simpler functions can be easily handled. In addition, it is possible to make use of statistical inference to determine the parameters shaping these simple functions from samples of observations across time or space (or both!). The practice of statistical inference is the more attractive as it is associated with the possibility of testing the quality of the adjustment between model and reality, thus bringing the "proof" of its validity.

Yet, the method is justified only insofar as the errors resulting from this approximation are of secondary importance with respect of the purposes and accuracy of the model under consideration. If these errors are large, then the model might lead to unjustified and misleading conclusions. At the same time, the interpretation of the statistical tests assessing the validity of estimations is itself subject to additional assumptions which might be discussible. Unfortunately, when decisions pertaining to agriculture are at stake, such situations are the rule rather than the exception …

There are two reasons for that. One is the fact that technical relationships do not always exhibit the regularity properties, such as derivability or smoothness which are so attractive and useful in the "soft" formulations of the production function. The second is linked with the methods by which the latter's specifications are obtained, using statistical methods for estimating parameters.

## 1 Easy to Use, Simple Functions Do Not Necessarily Represent Actual Production Possibility Sets: The Case of the Cobb Douglas Function

A typical production function made use of by economists is the "Cobb-Douglas" production function[1]:

$$y = aW^{\alpha}K^{\beta}L^{\gamma}, \tag{2.1}$$

where $L$ represents land (in ha), $K$, the capital (in \$) and $W$ the labour (in man day, or man year), while $y$ is the production (in tons) and $a$ scaling factor.

[1] Among others, see Malinvaud (1965).

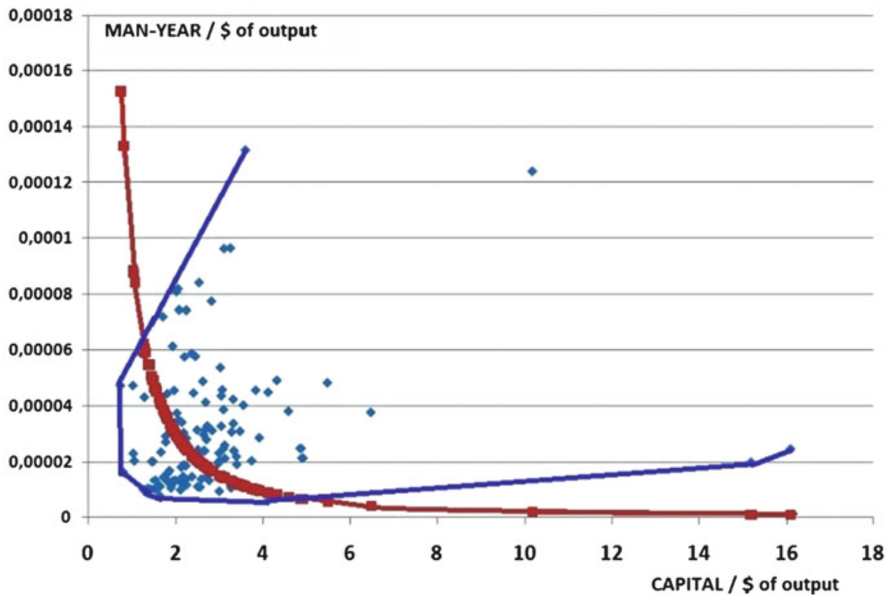


**Fig. 2.1** Cobb-Douglas isoquant and reality

Anybody knows that production increases when quantities of either capital, land or labor increase. Indeed, with the Cobb Douglas function, each partial derivatives of y with respect to *W, K* or *L* is positive. For instance:

$$\frac{\partial y}{\partial W} = \alpha a W^{\alpha-1} W^{\beta} L^{\gamma} = \alpha \frac{y}{W}, \tag{2.2}$$

which is clearly always positive). At the same time,

$$\frac{\partial^2 y}{\partial y^2} = \alpha(\alpha - 1)y/W, \tag{2.3}$$

which is clearly negative if $\alpha < 1$, thus reflecting the "law of decreasing returns" for such values of the parameter. At the same time, this function is indeed compatible with the idea of "constant returns to scale": with $\alpha + \beta + \gamma = 1$, it is easy to check that $y(\lambda W, \lambda K, \lambda L) = \lambda y$, which is the very definition of this concept, expressing that, for instance, it is possible to get twice as much wheat over 2 ha than over 1 ha, with twice as much capital and labor.

Thus, the Cobb Douglas production function with $\alpha + \beta + \gamma = 1$ and $0 < \alpha < 1$, $0 < \beta < 1$, $0 < \gamma < 1$ , reflects some of the essential properties expected from a production function.

Yet, if one compares the lay out of a Cobb-Douglas function with what can be observed in the real world, discrepancies become apparent. On Fig. 2.1 are plotted observations extracted from the FADN, the farm accounting network of the EC. This is an "isoquant" diagram : Each dot represents a combination of labor and capital observed in one region of the EC. In abscissa, are the quantities of capital per ha associated with the production of 1$/ha. In ordinate are the number of man/year per ha associated with the production of 1$/ha. The brown curve represents the relationship between these two variables as they can be derived from a Cobb Douglas production function, that is: $K = \left[\frac{y}{aK^{\beta}L^{\gamma}}\right]^{1/\alpha}$, for y = 1 and L = 1. Clearly, it is far from having any similitude with observed the reality as depicted by the FADN cloud of points. Indeed, two important observations may be derived from this graph:

1. We see from this diagram that with 2 units of capital, it is possible to get 1$/ha employing only 0.00002 men/ha (actually, a little less!). Now, one observes also that in some cases, with the same quantity of capital, some regions of the EC needed 0.00008 men/ha to get the same result: does that mean that the farmers in these regions were inefficient, and wasted labor without care? If yes, this the case of all the points lying in the north east side of the blue polygonal curve joining the points such that it is impossible to decrease the quantity of one factor without increasing the other – this is case of the vast majority of the points! Now, although it is quite possible that some farmers in the EC are really stupid, not all of them are so. If they seem squandering labor or capital, there must exists good reasons for that. One of these reasons might be that the "Cobb Douglas approach" is not the best way to understand what happens in reality.
2. The blue curve which joins the "efficient" points could be assumed to be derived from a better version of the "true" production function. Indeed, this is the approach popularized by the "data envelopment analysis" method, which does not seek any analytical representation of the production function, but only to determine an efficiency surface (in a multidimensional space), in such a way as to determine whether a firm is efficient or not. We see that even assuming that the Cobb Douglas function is a proxy for the efficiency surface is not tenable: the efficiency surface is much more "kinked" and "curved" than the isoquant derived from the Cobb Douglas function, which in this case implies the possibility of substitutions between labor and capital which do not exist in reality.

## 2 The Difficulties of Statistical Inference in Estimating Simple Functions Parameters: The Case of the Cobb Douglas Production Function

Even with these limitations, is it possible to make use of statistical inference to estimate the parameters (a,$\alpha$,$\beta$,$\gamma$) of the Cobb Douglas function from a sample of firms across space or time ? Actually, it is often possible to consider almost any economic as a random variable, the mean of which depends upon some controlled "explanatory" other variable, but the observed realizations of which are determined by some stochastic process.[2]

Of course, if one observes a sample of firms, the firm $i$ producing $y_i$ from quantities $W_i$, $K_i$, $L_i$ of labour, capital and land, it would be surprising that Eq. 1 be satisfied exactly : obviously, there exist many reasons for that Eq. 1 be satisfied only "approximately". In view of that, it is tempting to rewrite (1) as (2.1bis):

$$y = aW^{\alpha}K^{\beta}L^{\gamma} + \varepsilon \tag{2.1bis}$$

where ε represents a random variable, which can be described as Gaussian, with mean 0 and standard deviation σ. Under such a hypothesis, it tempting to make use of regression analysis for estimating the parameters a, $\alpha$ , $\beta$, and $\gamma$. It is the more easy as (2.1bis) can be rewritten as (2.1ter):

$$\log(y) = \log(a) + \alpha\log(W) + \beta\log(K) + \gamma\log(L) + \log(\varepsilon) \tag{2.1ter}$$

thus allowing the estimation of parameters with a mere linear regression, with the additional comfort of having the possibility of "testing" the validity of the regression, for instance with the "$t$ test": the estimated parameters $\hat{\alpha}, \hat{\beta}, \hat{\gamma}$ and $\hat{a}$ are random variables depending upon the probability law of ε. Indeed, under the above assumptions, $\hat{\alpha}, \hat{\beta}, \hat{\gamma}$ and $\hat{a}$ are Gaussian too, with standard deviations which can be computed. It is therefore possible to check whether all the estimates stay within the 0–1 interval where they should lie.

Unfortunately, such a trick would not provide any safe nor even likely estimates. There are many reasons for that. The first and most obvious one is that it is simply impossible to find any sample of firms where the triplets $W_i$, $K_i$, $L_i$ are chosen "at random". Most likely, firms are trying to maximize profits, thus choosing their production factors so as to equate marginal value products with prices. Thus, $p_w$, $p_k$, $p_l$ and $p_y$ being the prices of factors and output, one must expect: $p_y\partial y/\partial W = p_w$, or, equivalently (because of (2)) : $W = \alpha y p_y/p_w$, (and similar formulae for $K$ and $L$). But in this case, reporting these values for W,K, and L in (2.1ter) gives: $\log(\varepsilon) + (\alpha+\beta+\gamma-1)\ \log(y) + \alpha\ \log(\alpha p_y/p_w) + \beta\ \log(\alpha p_y/p_k) + \gamma\ \log(\alpha p_y/p_l) = 0$, implying a linear relation between the residual and the explained variable: this is precisely a situation where there is no point trying to minimize the sum of square of the residual, and where the matrix which have to be inverted for this purpose is singular . . . In such a context, it is much easier to make use of the above observation to set $\alpha = yp_y/Wp_w$ (and the same for $\beta$ and $\gamma$).

[2] The historical reference is Cobb and Douglas (1928). The discussion here is mainly derived from Malinvaud (1965). A more recent discussion can be found in Barnett (2007).

## 3 Removing the Limitations of the Cobb Douglas Function

The Cobb Douglas is the simplest and crudest of the functional forms having been proposed for playing the role of a "true" production function. Much more sophisticated alternatives have been envisaged, such as the "CES" or the "Translog" production function.

The CES production function is specified as:

$$y = \left(\sum_i \delta_i x_i^{-\rho}\right)^{-1/\rho}$$

where $y$ is the quantity of output, $x_i$ , the quantity of input i (such as $K$, the quantity of capital, or $W$ the quantity of labor, etc.), while the $\delta_i$'s and $\rho$ are parameters. The Cobb Douglas function is a special limit case of CES, with $\rho = 0$.

The translog (or "transcendental") production function (Christensen et al. 1971) is:

$$Log(y) = \sum_j a_j \log(x_j) + \sum_j \sum_i b_{ij} \log(x_i x_j)$$

Where, as above, y is the output, $x_i$ is the ith input, and $a_j$ and $b_{ij}$ are parameters. Again, the CES function can be considered as a special case of translog function.

Indeed, the whole family of the above production functions is based on the notion of "elasticity of substitution": the elasticity of substitution $\sigma_{1,2}$ of the production function $q = f(x_1, x_2)$at the point where the input "1" is at the level $x_1$, and the input "2" at the level $x_2$, indicates by how much is it possible to increase the quantity of one input and decrease the quantity of the other without changing the quantity produced. Specifically, $\sigma_{1,2} = d(x_1/x_2)/(x_1/x_2)$is the elasticity of substitution between inputs "1" and "2" if $dx_1$ and $dx_2$ are such that $dq = f'_1(x_1, x_2) + f'_2(x_1, x_2) = 0$. In the case of the Cobb Douglas function, the elasticity of substitution between any two inputs is 1. For the CES production function, it is some number $\sigma \neq 1$between any two pairs of inputs i and j. The Translog function allows for $\sigma_{ij} \neq \sigma_{i'j'}$, specific to each pairs: it is therefore much more general, hence the qualifier "transcendental".

If it possible to assume that the $\sigma$'s are more or less constant across empirical observations of the $x_i$ 's, if we assume in addition that the producer minimize cost, then it is possible to get statistical estimations of the functions parameters. In effect, under these assumptions, a few algebra shows that $\sigma_{12} = p_1 x_1 / p_2 x_2$, a relation between prices and quantities of inputs: the holy grail of economists!. Just observe a few firms with different $x_i/x_j$, assume small disturbances, and the job is done. Some complications arise because the residuals of regressions are not independent, thus requiring the estimation of a system of equations, (and still additional assumptions regarding the variance covariance matrix between these disturbances) but this is a matter of detail. What is not a matter of detail is the assumption according to which the $\sigma_{ij}$ are constant: there are indeed very few serious indications that it might be true.

Many other functional forms have been proposed to represent production functions. Each of them has its own merits, although none is fully satisfactory. Be it the CES, Translog or any other function, because they are much more “flexible”, they allow correcting the defect of the Cobb Douglas function regarding the substitutability between factors. The parameters are estimated from the “first order conditions” for profit maximization, thus avoiding the contradiction noticed above between these conditions and the estimation procedure. And they have other qualities of which the Cobb Douglas function is deprived. Nevertheless, although they can indeed improve the quality of estimations, and stand as valuable proxies, they must be handled with caution, for essentially the same reason: at best, they are good approximations of the vicinity of one point in the multidimensional space of inputs and outputs. But they cannot pretend covering the whole production possibility set . . .

## 4 Another Difficulty with Statistical Inference: The Specification of Non-technical Aspects of Observed Situations

Apart of the difficulty of specifying a reasonable functional form, and to estimate parameters from a set of observed factor combinations, there exists another potential inconsistency with such approaches. It is linked to the fact that observed factor combinations, although often decided on the basis of utility maximization, might very well be constrained by non technical considerations. To understand this point, let us again have a look at Fig. 2.1.

Obviously some of the points observed on this diagram are “inefficient”, in the sense that it is clearly possible to produce 1$ of output with both less capital and less labor than has been used in these cases. But does it mean that the farmers in the “inefficient” regions[3] had actually the choice to decrease the quantities of input they used? Although deeper investigations would be necessary to answer the question, it is quite possible to imagine alternative hypothesis. Is it possible to speak of a “capital input” or of a “labor input” without regard to their heterogeneity? Is it possible to add up the hours of labor of an engineer and of a second class private? Is it possible not to make any distinction between a tractor and a bag of fertilizer? Now, when looking at Fig. 2.1, it is difficult to admit that the producer who uses 12 units of labor and 10 units of capital for producing 1$ of output, is in exactly the same situation as is the colleague who get the same 1$ output from 1 unit of labor and 1 of capital. Along all probabilities, the first one is submitted to other

[3] Since each point, here, corresponds to one region.

constraints than those of the availability of capital and labor ... And explicitly stating this constraint (or these constraints) would probably have reduced the apparent inefficiency of this production system.

One possible way for expressing this constraint would be to assume that the “capital’ and “labor” of the “inefficient” producer are not the same as those of the “efficient” one: for instance, because one of them uses more engineering than the other, the costs and nature of the “labor” inputs are not the same, and should not be compared. This remark applies to any simplistic aggregation of physical quantities.[4] But the problem largely encompasses such considerations. In particular, in presence of uncertainty with respect to the prices and quantities of final outcomes, it might be quite rational to over capitalize, or over work. Having a tractor bigger than necessary to cope with risk of rain, increase the number of pesticide spreading to avoid any chance of contamination, such precautions are perfectly rational behaviors leading to apparent inefficiencies. Then, if one infers a production function from observations only, such considerations are ignored, and therefore, the apparent parameters of the function are biased. In the case of the situations represented on Fig. 2.1, it is clear that the error would be enormous.

This line of criticisms against “estimated” production functions is also valid for all the methods known under the label of “data envelopment analysis”,[5] as illustrated by the blue line of Fig. 2.1: indeed, it would be possible to derive an estimated isoquant (from which a full production function could be derived) from this “frontier”, the convex hull of observed production points. But apart from the fact that such a function would be difficult to handle (it is not smooth nor derivable), there are no indications of how much even the frontier production points are “inefficient”, nor why other points are so.

## 5 How to Cope with the Difficulty?

Thus, the popular trick of “estimating” production functions through statistical analysis, or event data envelopment analysis, although certainly useful, and a priority in the toolbox of the production economist, is far from being free of reproaches. And the fact that reality is difficult to handle is not a reason to replace

[4] This “problem of aggregation” used to be subject of important controversies in the 1950s and 1960s economics. It seems to be now unjustly forgotten. See Leontief (1966).

[5] The“data envelopment analysis”(DEA) methods have initially been developed by Farrell (1954). They stand as a generalization to the N dimensional space of the method illustrated here in two dimensions only. They got a large popularity, resulting in hundreds publications during the recent years. Because they do not constrain estimation by any predefined functional form, they are certainly much more flexible and realistic than the functions derived from the “elasticity of substitution” family. Yet, as indicated, they still are constrained by the specification of inputs and the ensuing dimension of the “convex hull”.

it by something simpler if the simplification is bound to produce erroneous conclusions and false expectations.

Here is the basic rational to make use of bio-economic models if not in all circumstances, at least whenever doubts arise on the validity of simpler classical statistically estimated production function. We shall see now how such functions can be built up.

## 5.1 Biological Models

For long, agronomists anxious to understand the plant growth process have developed purely biological "plant growth models". At the origin, they were not built for economic analysis purpose, quite the contrary. In effect, the problem was rather to check the validity of theories regarding the physiology of plants, and the laws governing their growth. Obviously, the growing of a plant is a complex process, explained by a number of rival theories. Integrating such theories into a model, and checking its resemblance with reality is a natural idea for any scientist to decide which of them better explain observations. Yet, a side product of these models is the possibility of predicting which yield will obtain a given plant, submitted to given climatic, soil, and agronomic practices. Next chapter of the present book develop the characteristics of these models.

Similar models – although probably not so accurate as for plants – exist also for animal production, again providing relations between animal raising practices and production.

All these models are complicated, not amenable to simple literal functions expression. Yet, much more than any other device, they deserve the name of production function in Robinson's sense. Therefore, they cannot be ignored by economists. But because they are not easy to handle, they normally need some sort of "post harvest treatment" before being made use of in economic reasoning and model building. Linear input/output analysis provides here an elegant way of doing it.

## 5.2 Input/Output Analysis

Input/output analysis was first developed by Wassily Leontief, who summarized his experience in a famous book (Leontief 1966). The basis of any economic model built along that line is the famous "Leontief production function" – in fact, only a vector of fixed coefficients, saying that for producing one unit of a given product using the technique j, need the quantities $a_{1j}\ldots a_{ij}..a_{nj}$ of $n$inputs. There is no functional form, nor any other mathematical intricacy. The only assumption is that any production level is feasible, provided the corresponding quantity of input is available. In effect, this means that the production depends only upon the level of the most binding availability constraint.

Such a device could be deemed extremely primitive in front of the sophisticated above mentioned mathematical instruments. Indeed, it would be so, if it was to be used in isolation. But this is not the case. In almost all the models built along that line, for each output, there exist a number of such techniques in competition. They are subject to constraints from the availability of inputs (each technique competing with others for input), but also from other considerations: for instance, the model can be constrained to secure at least enough cash to reimburse loans, thus allowing for a clear distinction between technical and risk constraints. It is possible to envisage constraints which have never been historically active, but which could become so under never historically encountered conditions: for instance, in agriculture, (and elsewhere) it is possible in that way to assess the effect of possible environmental constraints which have never been implemented.

The linear programming models stand as the archetype of these instruments. But modern input/output models do not need to be "linear": it is by now quite possible to accommodate non linear constraints and objective functions into this framework, even if, because of the characteristics of the Leontief function, most parts of the model will actually be linear. But linearity does not necessarily mean excessive simplicity, quite the contrary. Indeed, the combination of a large number of constraints limiting the development of competing Leontief techniques in the production of one product leads to the description of the "production possibility set" by a network of "hyperpolygons" in the N dimensional space enveloping a more or less complicated "feasible set". It is not simple at all, but much more realistic than most commonly used so called "analytical production functions".

Yet, such a complicated device remains quite manageable because the production possibility set represented by the constraints is subject to optimization. Because of optimization, each combination of constraint levels and production techniques results into one solution only. In this solution, usually, there is only one of the "Leontief techniques" emerging for each product – although this is not a rule: the coexistence of several techniques is quite possible. But the important thing is that different techniques "break even" under different circumstances and economic environments, thus allowing for a careful analysis of the interactions between technical choices and other variables.

Thus, the input/out framework provides the possibility of a complicated but easy to handle description of a production possibility set. It is the natural complement of the biological models, the results of which can therefore be accommodated into economic models. Yet, a question stands up: at which level should such a model be made use of? Clearly, it can be (and has been) used at the firm level. It can also be used at the country level, to assess the consequences of some envisaged policy: for instance, to look at the consequences of trade liberalization, or of any other policy. In that case, one has to assume that the political power can instruct economic agents to act as it is assumed by the model. In particular, if one maximizes the national welfare, a solution of the model could very well that one category of the population should be sacrificed. This is not realistic, nor compatible with the basic theorems of the welfare economics. But the input/output analysis is also compatible with the consideration of many agents, each optimizing their own objectives, under common constraints and information channels.

### 5.3 Multi-agent Input/Output Models

Consider a model with two agents, 1 and 2, agent 1 producing $x_1$ and agent 2 $x_2$ ($x_1$ and $x_2$ can be some vector of productions or techniques) subject to agent specific resources the levels of which are $k_1$ and $k_2$respectively and common resources the level of which is $h$. The agent 1 maximizes $U_1(x_1)$, and the agent 2 $U_2(x_2)$, under the constraints:

1. $f_1(x_1) + e_1 = k_1$
2. $f_2(x_2) + e_2 = k_2$
3. $g(x_1, x_2) + e_3 = h$

   Where :
   $f_1(x_1), f_2(x_2)$ and $g(x_1, x_2)$are functions of the quantities of output through a set of Leontief or other productions functions, as indicated above.
   $e_1, e_1$ and $e_3$are slack variables set to zero if the constraint is active, or positive otherwise.
   Obviously, it is out of question to maximize: $F = U_1 + U_2$, because a solution implying, say, a low $U_1$to get a larger $U_2$would be possible, while unsatisfactory.Yet, a classical theorem of optimization says that to any optimization problem, such as maximize $U_1(x_1)$subject to $f_1(x_1) + e_1 = k_1$ is associated a set of "dual" values $y_1$ such that: $y_1 f'_{x_1}(x_1) = U_1(x_1)$, and : $e_1 y_1 = 0$ (These last conditions being known as the "optimality conditions"). These conditions can be made use of to set up the problem as a system of equations with n equations and n unknowns instead of an optimization problem. Indeed, let us call $y_1$and $y_2$ the dual values associated with constraints (1) and (2), while $\lambda$ is associated to (3). Then we can write :

4. $\lambda g'_{x_1}(x_1, x_2) + y_1 f'(x_1) = U'(x_1)$
5. $\lambda g'_{x_2}(x_1, x_2) + y_2 f'(x_2) = U'(x_2)$
   Finally, we can add the conditions :
6. $\lambda e_3 = 0, y_1 e_1 = 0, y_2 e_2 = 0$

The system (1)–(5) and (6) provides eight equations for the eight unknowns: $x_1, x_2, e_1, e_2, e_3, \lambda, y_1, y_2$. Under certain conditions, it can be solved and return a unique solution. The conditions (4) and (5) imply that the "optimality conditions" are satisfied for agents 1 and 2, inasmuch as their objectives are corrected as:

$$u_1 = U_1(x_1) - \lambda g'_{x_1}(x_1, x_2) \text{ and } u_2 = U_2 - \lambda g_{x_2}{}'(x_1, x_2)$$

This trick to solve multi-agent problem is not specific to input/output models. It can be used with standard productions functions. But it is especially suitable for this case, in particular because with Leontief production functions the derivatives are just fixed constant numbers. It makes the building of the model straightforward, and its interpretation easy, while at the same time allowing for an extreme complexity in the setting up of the production functions and accessory constraints.

### *5.4 An Example*

In order to clarify the above reasonings, let us terminate by a sketchy example. It is inspired by a real situation in Tunisia. Two types of farms, "A" and "B" can grow wheat or raise sheep. For sheep raising, they are competing for a 300 ha resources of common grassland. Other resources are as follows:

Resources endowment, denoted by endo (r, f), with r = ("labor" or "land"), f = ("A" or "B")

| Farm | A | B |
|---|---|---|
| Labor availability (man day) | 1,500 | 1,800 |
| Land availability(ha) | 14 | 2 |

The input/output coefficients: denoted by Io (c, r) , with c = ("wheat" or "sheep") and r = ("labor" or "land") are:

| | Wheat | Sheep |
|---|---|---|
| Labor (man-day per ha or per sheep) | 15 | 0.45 |
| Land (ha per ha or per sheep) | 1 | 0.0091; |

The benefit (dinar per ha or per sheep) denoted by gm(c) with c = ("wheat" or "sheep") is given by

| Wheat | Sheep |
|---|---|
| 1,500 | 800 |

The common grass land usage com(c) is :

| Wheat | Sheep |
|---|---|
| 0 | 2 |

Let us call:

X(c, f) the quantity produced of product c (c="wheat" or "sheep") in farm type f (f = "A" or "B")
P(r,f) the dual values of the private ressource r (r = "labor" or "land") for farm f (f="A" or "B")
L the dual value of the common grassland

Then the equations are :

DV (Dual values): $\sum_r io(c,r)P(r,f) + com(c)L \geq gm(f)$ (written for each c and f);
PRA (private resources availability): $\sum_c io(c,r)X(c,f) \leq endo(r,f)$ (writen for each private resource)
PRC (common resources availability): $\sum_c com(c)X(c,f) \leq 300$

With a software like GAMS (General Algebraical Modeling System, cf Brooke et al. (1992), the slack variables are generated automatically. It is only necessary for technical reasons to associate each unknown with an equation: here the set of

DV equations is associated with the X's, the PRA equations with the P's, and PRC with L. The solution follows:

| | |
|---|---|
| X(xwheat,.A) = | 13. |
| X(wheat, B) . | 1.146 |
| X(sheep,A) | 56.141 |
| X(sheep,B) | 93.859 |
| P(labor,A) | 1272.400 |
| P(labor, B) | 1740.575 |
| P(land,A) . | 1500.000 . |
| P(land,B) . | 1,500. . . |
| L . | 393. |

As such, this solution is not of much interest (except that, surprisingly enough in view of the sketchy nature of the model, it reflected the observed reality fairly well). Yet, one can easily see that this model can easily be expanded by introducing different methods for sheep raising (as the output of cattle raising biological models) or wheat cultivation (coming from plant growth models). It would also be possible to add different species of crop or of cattle, to increase the number of common resources, to introduce risk constraints preventing income to fall under a certain minimum level in case of stochastic prices or yields, etc. There is hardly any limit to the details, which could be introduced in such a model, and to the variety of situations it can depict. At the same time, the programming burden is reduced to a minimum when using the GAMS software (or similar), thus allowing for a relatively easy data management.

## *5.5 Concluding Remarks*

Thus, the combination of biological models with Leontief production functions provides an ideal framework for economic investigation based upon data from real life, (unfortunately) contrary to many reasoning mathematically correct but deprived from empirical content. Although not necessarily confined to agricultural economic problems, this approach have been (and continue to be) a specificity of this discipline. This is because, here, production economics is confronted to such a complexity that only this approach is capable of sound results. It could certainly be extended to other sectors. Let us finally consider the Wassily Leontief opinion himself on this point :

"An exceptional example of the healthy balance between theoretical and empirical analysis and of the readiness of professional economists to cooperate with experts in the neighboring disciplines is offered by agricultural economics as it developed in this country over the last fifty years. A unique combination of social and political forces has secured for this area unusually strong organizational and generous financial support. Official agricultural statistics are more complete reliable and systematic than those pertaining to any other major sector of our economy.

Close collaboration with agronomists provides agricultural economists with direct access to information of technological kind. When they speak of crop rotations, fertilizers or alternative harvesting techniques, they usually know what they are talking about. They also were the first among economists to make use of advanced methods of mathematical statistics. However, in their hands, statistical inference became a complement to, not a substitute for empirical research" (Leontief 1971).

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# Chapter 3
# Dynamic Optimisation Problems: Different Resolution Methods Regarding Agriculture and Natural Resource Economics

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## 1 General Introduction

The need to take into account sustainability in agricultural resource management is now universally admitted. While the term "sustainability" can mean different things to different people, it always involves a consideration of the future. From an economic point of view, sustainability can be defined as an improvement of the performance of a system so as not to exhaust the basic natural resources on which its future performance depends (Pearce et al. 1990). This definition emphasizes the importance of preserving the natural resource base.

Thus, sustainability is a dynamic concept with underlying inter-temporal trade-offs. Implementing the notion of sustainability requires not only knowing to what extent short term profit is preferable to future profit, but also the effect of current production decisions on the future performance of the system.

From this standpoint, natural resources can be understood as stocks of natural capital. In the case of renewable resources, the availability of a resource will decrease if its extraction rate exceeds its rate of natural regeneration. For instance, if the water extraction rate from an aquifer exceeds the rate of replenishment, the availability of this resource will decrease over time.

Most natural resource problems involve sequential, risky and irreversible decisions. Thus the problem of natural resource management is one of inter-temporal

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allocation in a context of uncertainty and irreversibility. The mathematical basis for solving these dynamic problems is provided by the optimal control theory. The analytical solution of optimal control models, however, is only possible in the case of very simple problems. As a result, applied research calls for operational research techniques to treat increasingly complex resource management problems. Tools such as simulation, mathematical programming and dynamic programming can be used, depending on the problem's complexity.

Simulation models are designed to analyse the evolution of a system over time under a given policy and/or management scenario. These models are pertinent for extremely non-linear systems containing stochastic elements.

On the other hand, mathematical and dynamic programming models are inter-temporal optimisation models capable of obtaining an optimal solution, given the system's objective function and constraints. A reason often cited for the low adoption of these stochastic optimisation techniques is what Richard Bellman, the father of dynamic programming, termed the *curse of dimensionality*, which refers to the explosive growth of the model as the number of variables increases.

As we will show further on, dynamic programming models – well suited for fisheries and forestry management – are more difficult to apply to agricultural resource problems.

In this paper, these different techniques for solving dynamic optimisation problems are compared, particularly mathematical and dynamic programming. We emphasize the advantages and disadvantages of these methods and their respective fields of application. Furthermore, we propose an alternative technique for solving stochastic dynamic problems.

## 2 Dynamic Optimization Problems: Resolution Methods

Here, we deal with the different methods of solving inter-temporal optimisation problems, giving emphasis to dynamic programming and mathematical programming techniques. Non-sequential dynamic optimisation problems are presented in Sect. 2.1. In these types of model, the sequence of optimal decisions is determined at the beginning of the decision process and no modification is made afterwards.

In the sequential dynamic models presented in Sect. 2.2, decisions are taken sequentially and the decision-maker can adjust them when additional information is available. There are two well-known methods mathematically equivalent to solve stochastic dynamic optimisation problems concerning natural and agricultural resources: Stochastic Dynamic Programming (SDP) and Discrete Stochastic Programming (DSP).

## 2.1 The Problem of Non-sequential Dynamic Optimisation

As said previously, from a management standpoint, natural resources are better viewed of as stocks of natural capital that provide a flow of services (Wilen 1985). Thus the resource allocation problem consists in maximising the benefit obtained from using flows of resources through time, taking into account that current use can influence future availability. *Therefore the problem of allocating natural resources is a dynamic problem.* Consequently, optimal control theory provides the correct approach to natural resource management.

In this paper analysis is limited to discrete time, finite horizon problems. These problems include a decision sequence through time and can be represented by a decision tree. Decisions taken for each stage influence the possible results for the following stages. This kind of optimisation is inter-temporal.

Consider a simple problem of optimal allocation of a natural resource in a dynamic framework. For each period of time $t$, the system is described by a *state variable* ($x_t$) and a *control variable* ($u_t$); the former represents the stock of the resource while the latter represents the extraction decision. Let us suppose that we have an initial quantity of resource ($x_1$), that the decisions on the use of the resource ($u_t$) are taken at the beginning of each decision stage $t$ and that the profit obtained in each stage is given by $r_t(x_t,u_t)$.

The *objective function* to be maximised (profit or inter-temporal utility[1]) is generally expressed as a function of the first stage:

$$v_1(x_1, u_1, \ldots, u_T) = f\ [r_1(x_1, u_1), r_2(x_2, u_2), \ldots, r_T(x_T, u_T)] \tag{3.1}$$

Equation 3.1 expresses that the current value of the resource ($v_1$) is a function of the returns obtained throughout the planning horizon ($t = 1,\ldots,T$).

In a dynamic framework, the stock of the resource in year $t + 1$ is a function of both the decisions taken in year $t$ and the *autonomous* progression of the resource from $t$ to $t + 1$. This relation of dependence is expressed by the *equation of motion* or transition equation:

$$x_{t+1} - x_t = g_t(x_t, u_t)$$

To simplify the problem, we make a certain number of hypotheses:

- The objective function is an additively separable function defined by the discounted sum of the returns obtained throughout the planning horizon, given that $\rho$ is the discount factor;
- Functions $r_t$ and $g_t$ are assumed to be continually differentiable to the order two; and
- The stock of the resource at the end of the planning horizon has a final value given by $F(x_{T+1})$.

[1] The problem of defining individual or social utility functions is extremely difficult and is not dealt with in this paper.

The standard problem consists in determining the sequence of decisions ($u_t$) that maximise the objective function by respecting the constraints:

$$\text{Maximize} \quad \sum_{t=1}^{T} \rho^{t-1} r_t(x_t, u_t) + \rho^T F(x_{T+1}) \tag{3.2}$$

$$\text{subject to} \quad x_{t+1} - x_t = g_t(x_t, u_t) \quad t = 1, 2, \ldots, T \tag{3.3}$$

$$x(1) = x_1 \tag{3.4}$$

Therefore the problem consists in maximising the current value of the profits obtained throughout the planning horizon increased by the final value of the resource.

Equation 3.3 is the equation of motion or transition equation that reflects that the stock of the resource through time is both a function of resource extraction and resource renewal. Bio-economic models are spoken of in the case where biophysical models are used to obtain functions $g_t(x_t, u_t)$ or $r_t(x_t, u_t)$.

The analytical solution of the natural resource management problem given by Eqs. 3.2–3.4 calls on optimal control theory. The principle of maximum defines the inter-temporal optimality conditions of the optimal control problems.

Functions $r_t$ and $g_t$ are assumed to be continuous and differentiable to the order two. The search for the optimum is done by introducing the Hamiltonian:

$$H_t(x_t, u_t, \lambda_{t+1}) = r_t(x_t, u_t) + \rho\, \lambda_{t+1}\, g_t(x_t, u_t) \tag{3.5}$$

In the framework of natural resource economics, the Hamiltonian can be interpreted as the total profit resulting from the use of the resource: the first part represents the profit in the current stage ($t$), whereas the second part represents the change in the value of the stock. The multipliers $\lambda_{t+1}$ represent the values (measured in $t = 1$) given to an additional unit of stock $x_{t+1}$ in stage $t + 1$.

Maximising the Hamiltonian for each stage $t$ therefore amount to maximising the objective function. Introducing the Hamiltonian permits transforming the problem of constrained optimisation into one free of constraints.

The first-order conditions for profit maximisation are:

$$\frac{\partial H_t}{\partial u_t} = \frac{\partial r_t(\cdot)}{\partial u_t} + \rho\, \lambda_{t+1} \frac{\partial g_t(\cdot)}{\partial u_t} = 0 \quad t = 1, \ldots, T \tag{3.6}$$

$$\rho\, \lambda_{t+1} - \lambda_t = -\frac{\partial H(\cdot)}{\partial x_t} \qquad t = 2, \ldots, T \tag{3.7}$$

$$x_{t+1} - x_t = \frac{\partial H(\cdot)}{\partial(\rho\, \lambda_{t+1})} = g_t(\cdot) \qquad t = 1, \ldots .T \tag{3.8}$$

$$\lambda_{T+1} = F'(\cdot) \tag{3.9}$$

$$x(1) = x_1 \tag{3.10}$$

In Eq. 3.6, the right hand term is divided into two parts: the former is the marginal profit from using the resource in the current stage, whereas the latter part reflects the influence of the decision taken $u_t$ on the value of the resource over the remaining stages, i.e. the inter-temporal cost of resource extraction, or *user cost*. The shadow price $\lambda_{t+1}$ reflects the increase of profit throughout the remaining stages if the stock of the resource increases by one unit (or the loss of profit throughout the remaining planning horizon due to the consumption of an additional unit in the current stage).

Equation 3.7 indicates the change of Lagrange multipliers through time. Equation 3.8 is the transition equation while Eqs. 3.9 and 3.10 are the boundary conditions (final and initial conditions).

Equations 3.6–3.10 form a system of (3T + 1) equations with (3T + 1) unknowns: $u_t$ for $t = 1,\ldots T$; $x_t$ for $t = 2,\ldots T+1$ and $\lambda_t$ for $t = 1,\ldots, T+1$. It is not always possible to solve this system of equations simultaneously. Although the theory of optimal control is well-suited for dealing with natural resource problems, if the extremes are not interior points, or the functions are not continuous and differentiable, no analytical solution is possible. In practice, it is usual to use resolution algorithms such as dynamic programming and mathematical programming.

These methods of solving dynamic optimisation problems will be presented in what follows, with emphasis being given to possibilities of applying each method in the field of natural resource economics rather than to the resolution procedure.

### 2.1.1 The Dynamic Programming Method

The dynamic programming method was developed by Richard Bellman during the 1950s. It permits solving this type of problem, provided that the objective function is separable.

By designating the optimal value of the resource stock in stage $t$ by $V_t(x_t)$ – i.e. the value of the resource when optimal decisions $u^*_t, u^*_{t+1},\ldots,u^*_T$ have been taken – the problem consists in finding $V_1(x_1)$:

$$V_1(x_1) = \max_{u_1,u_2,\ldots,u_T} f[r_1(x_1,u_1), r_2(x_2,u_2),\ldots, r_T(x_T,u_T)] \tag{3.11}$$

If the objective function respects the conditions of separability, the multi-stage problem can be broken down into $T$ one-stage problems by using the *recursive relation*:

$$V_t(x_t) = \max_{u_t} f[\mathrm{r_t}(x_t,u_t),\ V_{t+1}(x_{t+1})] \quad t = T, T-1,\ldots 1 \tag{3.12}$$

In this equation $V_t(x_t)$ represents the optimal value of the objective function throughout the remaining planning horizon under optimal decisions. If the objective function is the sum of the discounted profits of each stage, the problem consists in determining the optimal sequence of decisions $u^*_1,\ldots,u^*_T$ which obeys:

$$V_t(x_t) = max_{u_t}\left[r_t(x_t, u_t) + \rho \ \ V_{t+1}(x_t + g_t(x_t, u_t))\right] \qquad t = T, T-1, \ldots, 1 \tag{3.13}$$

$$V_{T+1}(x_{T+1}) = F(x_{T+1}) \tag{3.14}$$

$$x(1) = x_1 \tag{3.15}$$

Functional Eq. 3.13 permits determining $V(x_t)$ once $V(x_{t+1})$ is known. Since the final value is assumed as known, it is possible to determine the optimal value for stage $T$:

$$V_T(x_T) = max_{u_T}\left[r_T(x_T, u_T) + \rho \ \ F(x_{T+1})\right] \tag{3.16}$$

Solving this equation for each possible value of the state variable ($x_T$) allows us to obtain $u^*_t$ and $V_T(x_T)$ and repeating this procedure for stages $T$-$1$, $T$-$2$,. . ., $1$ permits solving the problem.

In practice, the analytical resolution of the recursive relation (3.13) demands that functions $V_t(x_t)$ and $r_t(x_t,u_t)$ be differentiable and that an interior solution exists. If these conditions are not respected, the problem can still be solved by using numerical methods. However, numerical resolution limits the possible values of the state and control variables to a discrete set for each stage $t$. Furthermore, any decision taken in stage $t$ must lead to one of the possible values of $x_{t+1}$.

The numerical formulation of the problem can be interpreted as the search for an optimal path through a nodal network, since the characteristics of the optimal path are given by *Bellman's principle of optimality* (Bellman 1957): «an optimal policy is one in which, whatever the initial state and initial decision, the following decisions must constitute an optimal policy in relation to the state resulting from the initial decision». The dynamic programming method permits obtaining a decision rule so that it is easy to determine the optimal trajectory for different initial conditions.

Numeric dynamic programming is a very flexible method that permits the resolution of inter-temporal optimisation problems even when functions $r_t$ and $g_t$ are not continuous and differentiable. It permits obtaining a full decision rule, whereas other techniques, such as the multi-stage programming methods seen further on, only give solutions for specific initial conditions.

Since the recursive relation must be solved for all the values related to the state variables, the main disadvantage of this model is that its size explodes when the number of state variables increases (the "curse of dimensionality").

Many applications of this method exist in the field of agricultural and natural resource economics. Kennedy (1986) wrote a detailed review on this subject.

### 2.1.2 The Mathematical Programming Method

The problem of dynamic optimisation can also be solved as a constrained optimisation problem, by using mathematical programming techniques.

With this method, state and control variables are defined as activities while the transition equations are defined as multi-period constraints that link the stages together. Mathematical programming permits obtaining an optimal solution, given the constraints and the objective function. Non-linear programming algorithms and techniques now exist that allow incorporating uncertainty not only in the objective function, but also in the constraints.

Whereas the dynamic programming method solves problems recursively, by backward induction, the mathematical programming method consists in solving all the following equations simultaneously, by using one of the existing algorithms:

$$\text{Maximize} \quad \sum_{t=1}^{T} \rho^{t-1} r_t(x_t, u_t) + \rho^T F(x_{T+1}) \tag{3.17}$$

$$\text{subject to} \quad x_{t+1} - x_t = g_t(x_t, u_t) \quad t = 1, 2, \ldots, T \tag{3.18}$$

$$x(1) = x_1 \tag{3.19}$$

The mathematical programming method permits incorporating in the model the diversity of activities and constraints specific to decision-making in agriculture. The advantages of this method are considerable in comparison to dynamic programming when the problem is deterministic or when stochastic components can be approached with the non-sequential techniques used in risk programming.

Although dynamic programming remains the technique used most often for solving dynamic optimisation problems, several authors emphasise the advantages of mathematical programming when incorporating the interdependencies between the different resource allocation decisions in the model (Standiford and Howitt 1992; Yates and Rehman 1998).

The mathematical programming method permits working with continuous variables and incorporating all the activities and constraints considered necessary. Nevertheless, it is not always possible to obtain a global maximum for very complex non-linear models. This difficulty could be overcome by using genetic algorithms (Cacho 2000).

## 2.2 *Problem of Sequential Dynamic Optimisation*

The problem becomes stochastic if the state variables and/or the results of each stage depend not only on the state of the system and the decisions taken, but also on random variables that the decision-maker cannot control.

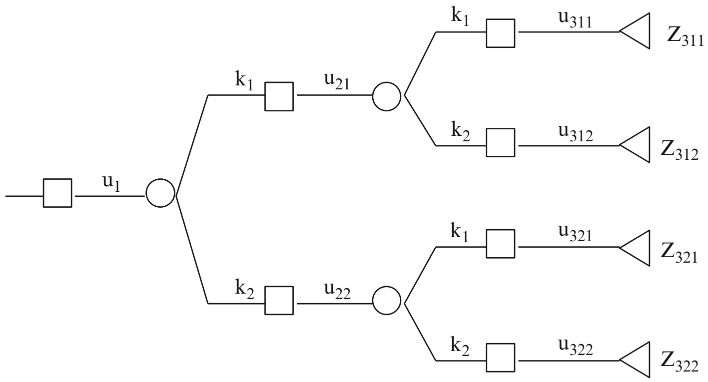


**Fig. 3.1** Decision tree (three decision stages and two states of nature)

Generally, a sequential stochastic decision problem can be represented by a decision tree. For example, Fig. 3.1 represents a problem with three decision stages and two states of nature.

The diagram shows that by starting from an initial state of the system (represented by a small square), the farmer takes decisions in stage 1 ($u_1$). Later, according to the state of nature occurring ($k_1$ or $k_2$), the farmer can take other decisions ($u_{21}$ is, for example, the decisions taken in stage 2, taking into account the state of nature $k_1$).

In sequential stochastic problems, one of the objective functions used most frequently is the mathematical expectation of total discounted profit[2]:

$$v_1(x_1) = E[f\{r_1(x_1, u_1, k_1),\ r_2(x_2, u_2, k_2), \ldots,\ r_T(x_T, u_T, k_T)\}] \tag{3.20}$$

Let us suppose that random variables ($k_t$) take different discrete values in each stage $t$ with associated probabilities $p_t(k_t)$, and that the objective function is the expected present value. By hypothesising that the problem can be written as a Markovian decision process, i.e. that the state of the system in stage $t + 1$ only depends on $x_t$, $u_t$ and $k_t$, the problem is written as:

$$\text{Maximize} \quad \sum_{t=1}^{T} \rho^{\,t-1}\, E\left[r_t(x_t, u_t, k_t)\right] + \rho^T\, F(x_{T+1}) \tag{3.21}$$

$$\text{subject to} \quad x_{t+1} - x_t = g_t(x_t, u_t, k_t) \quad t = 1, 2, \ldots, T-1 \tag{3.22}$$

$$x(1) = x_1 \tag{3.23}$$

[2] In the case where the decision-maker is not considered risk-neutral, other objective functions can be proposed.

given that:

$$E[r_t(x_t, u_t, k_t)] = \sum_k p_t(k_t) \;\; r_t(x_t, u_t, k_t) \tag{3.24}$$

As seen further on, this problem can be solved by using dynamic programming or discrete stochastic programming.

### 2.2.1 The Stochastic Dynamic Programming Method

The stochastic dynamic programming method (SDP) permits breaking down the inter-temporal optimisation problem into $T$ single stage problems. The problem consists in solving the recursive relation:

$$V_t(x_t) = \max{}_{u_t}\{E[r_t(x_t, u_t, k_t)] + \rho \;\; V_{t+1}(x_t + g_t(x_t, u_t, k_t))\} \quad t = T, T-1, \ldots, 1 \tag{3.25}$$

subject to:

$$V_{T+1}(x_{T+1}) = F(x_{T+1}) \tag{3.26}$$

$$\sum_k p_t(k_t) = 1 \tag{3.27}$$

$$x(1) = x_1 \tag{3.28}$$

As in the deterministic case, the recursive relation permits solving the problem by starting with the last stage and working backwards, stage by stage, to the initial stage. One of the great advantages of dynamic programming is that it permits treating deterministic and random processes similarly.

Applications of this technique to agriculture decision problems have been reviewed by Taylor (1993).

### 2.2.2 The Discrete Stochastic Programming Method

Discrete stochastic programming (DSP) can be used to process sequential decision-making problems in discrete time with a finite horizon when the state and control variables are continuous. This approach was developed by Cocks (1968) and then Rae (1971a).

The DSP method requires that the problem be formulated as a problem of constrained optimisation. Equations are solved simultaneously by using a mathematical programming algorith. Although the notation of stochastic programming models is complicated, they are relatively simple conceptually.

The logic of this technique can be understood from the formulation of a model with two decision stages:

$$\text{Maximize} \quad \sum\nolimits_k p_k[r_1(x_1, u_1) + \rho \;\; r_{2k}(x_{2k}, u_{2k})] \tag{3.29}$$

$$\text{subject to} \quad x_{2k} - x_1 = g(x_1, u_1) \tag{3.30}$$

$$u_1, \;u_{2k} \geq 0 \tag{3.31}$$

where sub-indices 1 and 2 represent the two decision stages, $k$ the state of nature, and $p_k$ the vector of probabilities of states of nature.

This formulation of the model implies that the agent takes several initial decisions ($u_1$) with uncertain knowledge of the future. This is followed by one of the states of nature ($k$) and the agent will take other decisions ($u_{2k}$) later on that depend on the decisions made in the first stage and the state of nature having occurred.

Discrete stochastic programming models have been used by Rae (1971b) to model decision-making in agriculture. They are very flexible and do not require the utility function to be separable; moreover, they permit considering the different sources of risk that influence the objective function and the constraints. However, they are often very large and need considerable amounts of data, thus few DSP applications exist to date (Blanco-Fonseca 1999). See Apland and Hauer (1993) for a review of the applications of this method.

## 3 Recursive Stochastic Programming: A New Method of Solving Dynamic Problems?

In practice, the methods for solving the inter-temporal optimisation problems mentioned above suffer from major limitations. Despite the existence of powerful algorithms capable of tackling these problems, the model's variables and/or stages must always remain small in number.

Despite the fact that a large number of decision stages can be considered using the dynamic programming method, since the multi-stage problem in question is broken down into several one-stage problems, the number of state and control variables must remain limited. In practice, this technique requires limiting the possible values of the model's state and control variables to a discrete set. The solutions obtained are therefore approximate and the degree of precision will depend on the differences between the values inside the discrete set. In the case of non-linear functions, the errors can be non-negligible. Furthermore, all the decisions made in the current stage must lead to a "possible" state of the system in the following stage, sometimes requiring that other approximations be made.

Undoubtedly, the most serious disadvantage of dynamic programming is the difficulty of considering the diversity of activities and constraints specific to the field of agricultural and natural resource economics.

On the contrary, discrete stochastic programming permits simultaneously taking into account the uncertainty and the diversity of activities and constraints specific to agricultural decision problems. DSP permits working with continuous variables and non-linear functions[3]. Nonetheless, its application remains limited to problems with a low number of stages. Since optimisation is inter-temporal, the model's size increases exponentially with the number of decision stages.

In the previous models, the decision-maker makes decisions by taking into account their consequences on the future. This entails inter-temporal optimisation under uncertainty and irreversibility. It can be likened to a game of chess: the player takes into account the possible reactions of his adversary and his own counter-reactions in full knowledge of the rules.

This leads us to raising the question of whether the rules are as well known in natural resource economics, i.e. does the agent have full knowledge about the possible responses of nature?

By making the hypothesis that the decision-maker is perhaps more *myopic* than the dynamic programming would like, we propose another method of solving dynamic problems. The main difference of this method in comparison to the previous ones is the way the information enters the problem. In this case, the decision-maker does not have all the information available when making decisions; hence he is unable to fully anticipate the responses of nature and must opt for a sub-optimal decision. Once the first decision has been carried out, the system evolves (the decision-maker knows the response of nature) and the agent can adjust later decisions according to the new information available.

The method consists in solving the dynamic problem by making a series of sequential optimisations, thus it is a recursive method where each optimisation comprises a dynamic model.

Consequently, at moment *1*, the decision-maker chooses a decision plan by taking into account all the information available at this moment. At moment 2, the decision taken for the first stage ($u_1$) has already been carried out and, as a function of the state of nature happened, the system will have progressed to reach state $x_{2k}$. The agent can now revise the decision plan, not for stage *1* but for the following stages depending on the new information available. This procedure is illustrated in Fig. 3.2.

The first iteration therefore consists in solving the optimisation problem given by equations:

$$\text{Maximize} \quad \sum_{t=1}^{T} \rho^{t-1}\, r_t(x_t, u_t) + \rho^T\, F(x_{T+1}) \tag{3.32}$$

$$\text{subject to} \quad x_{t+1} - x_t = g_t(x_t, u_t) \quad t = 1, 2, \ldots, T \tag{3.33}$$

$$x(1) = x_1 \tag{3.34}$$

[3] Obtaining a global maximum cannot always be achieved by using available non-linear programming algorithms, though it can be obtained by adequate formulation of the problem.

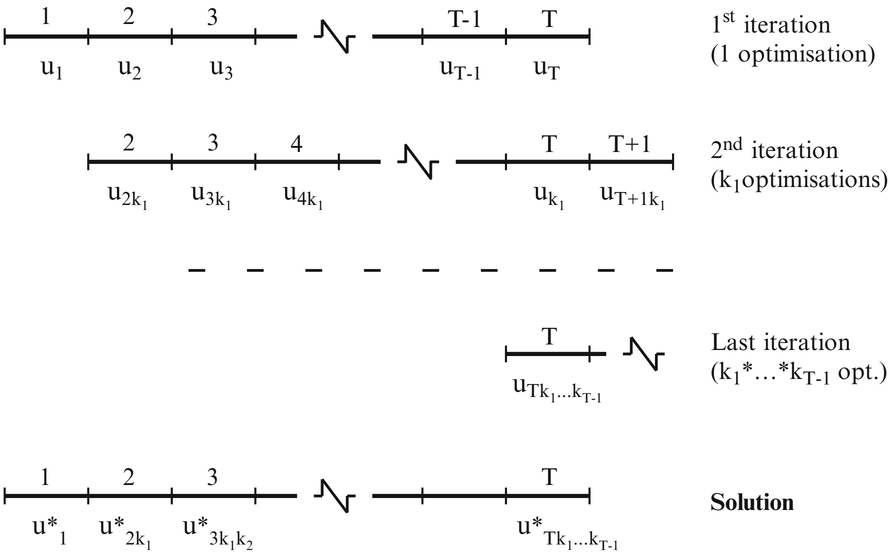


**Fig. 3.2** Diagram of the recursive stochastic programming method

In this case, function $g_t(x_t,u_t)$ does not depend on the state of nature happened, rather it has a definite value resulting, for example, from taking into account the mathematical expectation of random variable $k$.

Once the solution has been obtained, we will only take into account the result for the first stage, $u_1$, and we will determine the state of the system in the following stage for each state of nature $k$:

$$x_{2k_1} - x_1 = h_1(x_1, u^*{}_1, k_1) \quad k_1 = 1, 2, \ldots K \tag{3.35}$$

The second iteration consists in solving a series of optimisations, one for each initial state of system $x_{2k}$:

$$\text{Maximize} \quad \sum_{t=2}^{T+1} \rho^{t-1}\, r_t(x_{tk_1}, u_t) + \rho^T\, F(x_{Tk_1+1}) \quad \forall k_1 \tag{3.36}$$

$$\text{subject to} \quad x_{t+1,k_1} - x_{tk_1} = g_t(x_{tk_1}, u_t) \quad t = 1, 2, \ldots, T \tag{3.37}$$

$$x(2) = x_{2k_1} \quad \forall k_1 \tag{3.38}$$

This process is repeated $T$ times and the solution to the problem is obtained by retaining the result for the first stage at each iteration:

$$u^* = (u^*{}_1, u^*{}_{2k}, u^*{}_{3k_1k_2}, \ldots) \tag{3.39}$$

As will be seen in what follows, in the case of simple models this technique gives the same results as the two previous ones, while permitting taking into account a large number of variables and decision stages. However, the sequence of optimal decisions in more complex problems will be different.

This method has major advantages when the system must be represented by a considerable number of state variables or in the case of a large number of possible activities (reservoir management, irrigation management, soil erosion, etc.). Furthermore, it allows the introduction of exogenous changes other than stochastic resource availability.

This type of model permits a sequential representation of decision-making by assuming that decisions are irreversible, as in sequential decision stochastic models. A decision tree can be modelled similar to that used in sequential decision stochastic models. This type of model permits getting round the *curse of dimensionality* and solve a problem with many variables and decision stages.

Several applications of recursive models with multi-stage components have been described in the literature (Louhichi et al. 1999; Barbier 1998; Barbier and Bergeron 1999). However, the objective of recursivity in these models is not to represent the sequential stochastic nature of the problem, but to permit exogenous changes of some of the model's parameters. What is original in this work is that it proposes using recursive programming as a method for solving sequential stochastic problems.

We could go even further and consider a problem with two different decision horizons: a short-term horizon and a long-term one. We can, for instance, introduce a more thorough modelling of the nearest stages and reduce details as distance increases through time.

To illustrate this procedure, imagine that we wish to model an agricultural decision process in a context of climatic uncertainty. We can assume that long-term decisions (e.g., investment decisions) are taken according to the probability of the occurrence of states of nature. Nonetheless, the farmer can make adjustments (amount of fertiliser, irrigation, etc.) throughout the year. To model this behaviour, we can build a multi-stage model whose first stage of simulation is divided into several sub-stages. Decisions throughout the year are taken sequentially as a function of the state of nature occurring. Investment decisions are taken at the beginning as a function of the probabilities of states of nature throughout the planning horizon.

Since we have to repeat this procedure by using a sliding planning horizon, the model is formulated to adjust the decisions taken for the following years.

## 4 A Numerical Example

Comparison between the different methods of solving dynamic optimisation problems can be shown by using the crop-irrigation problem proposed by Kennedy (1986). In this example, a farmer owns 100 ha and produces three horticultural crops each year in successive seasons, i.e. each crop occupies the soil for 3 months.

The production of each crop ($y_t$, in thousand tonnes) depends on the depth (cm) of water applied ($w_t$) according to relation $y_t = 0.1\ (w_t - 0.1\ w_t^2)$. The farmer has a small reservoir for irrigation whose stocks of water vary as a function of consumption and rainfall during each season. The depth of water applied to each crop depends on the water released from storage at the beginning of each season ($u_t$, in metres) and the rainfall occurring during this season ($q_t$), i.e. $w_t = u_t + q_t$.

The maximum level of water in the reservoir is 3 m and it is assumed to be full at the beginning of the year. The amount of water which can be released at the beginning of each season is limited do integer values (metres of water) and by the amount in storage.

The farmer seeks to determine water release in each season ($u_t$) so as to maximise the present value of receipts from sale of the crops. Thus it is a dynamic problem with three decision stages, where the water used in each stage ($u_t$) is the control variable while the water stock ($x_t$) is the state variable.

We approach the deterministic problem first before going on to the stochastic version in which the rainfall of each season is a random variable.

## 4.1 *The Deterministic Dynamic Model*

As mentioned above, the farmer seeks to determine the quantity of water released from the reservoir in each stage ($u_t$) in order to maximise the current value of the farm's revenue ($V_1$). Since $b_t$ is the price of the crop (thousand dollars per thousand tonnes) corresponding to stage t, the revenue (thousand dollars) of the farm in this stage ($r_t$) is:

$$r_t = 0.1\ b_t \left[u_t + q_t - 0.1(u_t + q_t)^2\right] \tag{3.40}$$

Given that ρ is the discount factor, the problem is written as:

$$\text{Maximize} \quad \sum_{t=1}^{3} \rho^{t-1}\ r_t \tag{3.41}$$

$$\text{subject to} \quad 0 \le u_t \le x_t \le 3 \quad u_t, x_t \text{ integers} \tag{3.42}$$

$$x_{t+1} = min\{(x_t - u_t + q_t),\ 3\} \tag{3.43}$$

$$x_1 = 3 \tag{3.44}$$

given that $b = [50, 100, 150]$
$q = [2, 1, 1]$
$\rho = 0.95$

This problem can be solved indifferently by dynamic programming and mathematical programming.

By using dynamic programming, we can express the backward recursive relation as:

$$V_t(x_t) = \max_{0 \le u_t \le x_t} \{r_t + \rho\ V_{t+1}(x_{t+1})\} \quad t = 3, 2, 1$$

And the problem consists in solving this equation with the conditions:

$$x_{t+1} - x_t = -u_t + q_t$$

$$V_4(x_4) = 0$$

By starting with the last stage:

$$V_3(x_3) = \underset{u_3}{max}\ r_3 = \underset{u_3}{max} \left\{0.1\ \ b_3\left[u_3 + q_3 - 0.1(u_3 + q_3)^2\right]\right\}$$

we can determine the optimal decision $u^*_3$ and value $V_3(x_3)$ for the different possible values of $x_3$. Then, on the basis of equation:

$$\begin{aligned} V_2(x_2) &= \underset{u_2}{max} \{r_2 + \rho\ V_3(x_3)\} \\ &= \underset{u_2}{max} \left\{0.1\ \ b_2\ \left[\ u_2 + q_2 - 0.1\ (u_2 + q_2)^2\right] + \rho\ V_3(x_3)\right\} \end{aligned}$$

we obtain $u^*_2$ and $V_2(x_2)$ for the different possible values of $x_2$. Lastly, equation:

$$\begin{aligned} V_1(x_1) &= \max_{u_1} \{r_1 + \rho\ V_2(x_2)\} \\ &= \max_{u_1} \left\{\ 0.1\ \ b_1\ \left[\ u_1 + q_1 - 0.1\ (u_1 + q_1)^2\right] + \rho\ V_2(x_2)\right\} \end{aligned}$$

allows us to determine $u^*_1$ and $V_1(x_1)$ for the different possible values of $x_1$. This resolution procedure leads to the following results (Table 3.1):

This procedure allows us to determine the optimal decision path for other initial water levels.

The same results can be obtained by using a non-linear programming algorithm[4].

[4] All the models have been solved by using the GAMS software.

**Table 3.1** Results of the deterministic problem

| Decision stage, t | State variable, $x_t$ | Control variable, $u_t$ | Current value, $V_t(x_t)$ |
|---|---|---|---|
| 1 | 3 | 2 | 60.4 |
| 2 | 3 | 2 | 50.9 |
| 3 | 2 | 2 | 31.5 |

**Table 3.2** Distribution of rainfall probability

| | Decision stage | | | | | |
|---|---|---|---|---|---|---|
| | 1 | | 2 | | 3 | |
| State of nature k | $p_1(k_1)$ | $q_1^k$ | $p_2(k_2)$ | $q_2^k$ | $p_3(k_3)$ | $q_3^k$ |
| 1 | 0.25 | 1 | 0.25 | 0 | 0.25 | 0 |
| 2 | 0.50 | 2 | 0.50 | 1 | 0.50 | 1 |
| 3 | 0.25 | 3 | 0.25 | 2 | 0.25 | 2 |

## 4.2 *Stochastic Dynamic Model*

Consider now a stochastic version of the crop-irrigation problem introduced in last section. Let us suppose that rainfall in each stage, which influences both crop yields and water stocks in the reservoir, is a random variable (*q*) and that three states of nature can be distinguished (*k*) (Table 3.2):

Note that expected rainfall in each stage is the same as for the deterministic problem.

In this case, for each stage *t*, the revenue from the farm depends on the state of nature occurring in stage *t-1*:

$$r_{tk} = 0.1 \;\; b_t \left[ u_t + q_{tk} - 0.1 (u_t + q_{tk})^2 \right] \tag{3.45}$$

Supposing that the objective function is the expected discount value, the problem can be written as:

$$\text{Maximize} \quad \sum_{t=1}^{3} \rho^{t-1} \left[ \sum_{k=1}^{3} p_{tk} \; r_{tk} \right] \tag{3.46}$$

$$\text{subject to} \quad 0 \le u_t \le x_t \le 3 \qquad u_t, x_t \;\; \text{integers} \tag{3.47}$$

$$x_{t+1} - x_t = -u_t + q_{kt} \tag{3.48}$$

$$x_1 = 3 \tag{3.49}$$

This problem can be solved by backward induction by using stochastic dynamic programming or as a constrained inter-temporal optimisation problem by using discrete stochastic programming. Further on we comment on these methods and on resolution by recursive stochastic programming.

**Table 3.3** A few results of the stochastic problem (dynamic programming)

| | Less favourable | | | More favourable | | | More probable | | |
|---|---|---|---|---|---|---|---|---|---|
| Stage t | Rainfall, $q_t$ | State, $x_t$ | Control, $u_t$ | Rainfall, $q_t$ | State, $x_t$ | Control, $u_t$ | Rainfall, $q_t$ | State, $x_t$ | Control, $u_t$ |
| 1 | 1 | 3 | 2 | 3 | 3 | 2 | 2 | 3 | 2 |
| 2 | 0 | 2 | 1 | 2 | 3 | 2 | 1 | 3 | 2 |
| 3 | 0 | 1 | 1 | 2 | 3 | 3 | 1 | 2 | 2 |
| Current value | | 31.2 | | | 69.1 | | | 60.4 | |

### 4.2.1 Resolution by Stochastic Dynamic Programming

In this case, we simply need to write the backward recursive relation:

$$V_t(x_t) = \max_{0 \le u_t \le x_t} \left\{ \sum_{k=1}^{3} p_{tk}\, r_{tk} + \rho\, V_{t+1}(x_{t+1}) \right\} \qquad t = 3, 2, 1$$

with the transition equation and boundary conditions:

$$x_{t+1} - x_t = -u_t + q_{tk}$$

$$V_4(x_4) = 0$$

$$x_1 = 3$$

Since the final value is known, it is possible to determine the optimal decisions in the third stage for each possible value of $x_3$ from:

$$V_3(x_3) = \max_{u_3} \sum_{k=1}^{3} p_{3k}\, r_{3k}$$

which allows us to determine the optimal decisions in the second stage by solving:

$$V_2(x_2) = \max_{u_2} \left\{ \sum_{k=1}^{3} p_{2k}\, r_{2k} + \rho\, V_3(x_3) \right\}$$

and the decisions in the first stage by repeating this process. Now, starting from the system's initial state ($x_1 = 3$), we can obtain the sequence of optimal decisions for each stage as a function of the state of nature occurring.

Obviously, the objective function optimal value will be less in the stochastic case (57.1) than in the deterministic formulation of the problem (60.4). Table 3.3 shows the results for the series of "less favourable", "more favourable" and "more probable" rainfalls.

### 4.2.2 Resolution by Discrete Stochastic Programming

We shall now solve the example as an constrained inter-temporal optimisation problem.

Notation becomes more complex, because the state and control variables depend on the states of nature occurring in the past; the model is not solved recursively but by an optimisation algorithm. In our example, although the decision to be taken in the first stage ($u_1$) does not depend on states of nature, that of the second stage ($u_{2k}$) will depend on the state of nature occurring in the first stage, while that of the third stage ($u_{3km}$) will depend on the states of nature in the two previous stages.

To simplify notation, we designate the possible states of nature in stage 1 by $k$, those of stage 2 by $m$ and those of stage 3 by $n$. The formulation of the discrete stochastic programming problem requires differentiating the state and control variables for each decision stage. In our example:

| Decision stage | State variables | Control variables |
|---|---|---|
| 1 | $x_1$ | $u_1$ |
| 2 | $x_{2k}$ | $u_{2k}$ |
| 3 | $x_{3km}$ | $u_{3km}$ |
| 4 | $x_{4kmn}$ | |

The farm's revenues in each stage are:

$$r_{1k} = 0.1\ b_1\ \left(u_1 + q_{1k} - 0.1\ (u_1 + q_{1k})^2\right) \tag{3.50}$$

$$r_{2km} = 0.1\ b_2\ \left(u_{2k} + q_{2m} - 0.1\ (u_{2k} + q_m)^2\right) \tag{3.51}$$

$$r_{3kmn} = 0.1\ b_3\ \left(u_{3km} + q_{3n} - 0.1\ (u_{3km} + q_{3n})^2\right) \tag{3.52}$$

For each possible path (for each branch of the decision tree), the current value of the farm's revenues will be:

$$VA_{kmn} = r_{1k} + \rho\ \ r_{2km} + \rho^2\ r_{3kmn} \tag{3.53}$$

Since we attempt to maximise the expected present value, the problem is written as:

$$\text{Maximize} \quad \sum_{k=1}^{3}\sum_{m=1}^{3}\sum_{n=1}^{3} p_k\ p_m\ p_n\ \ VA_{kmn} \tag{3.54}$$

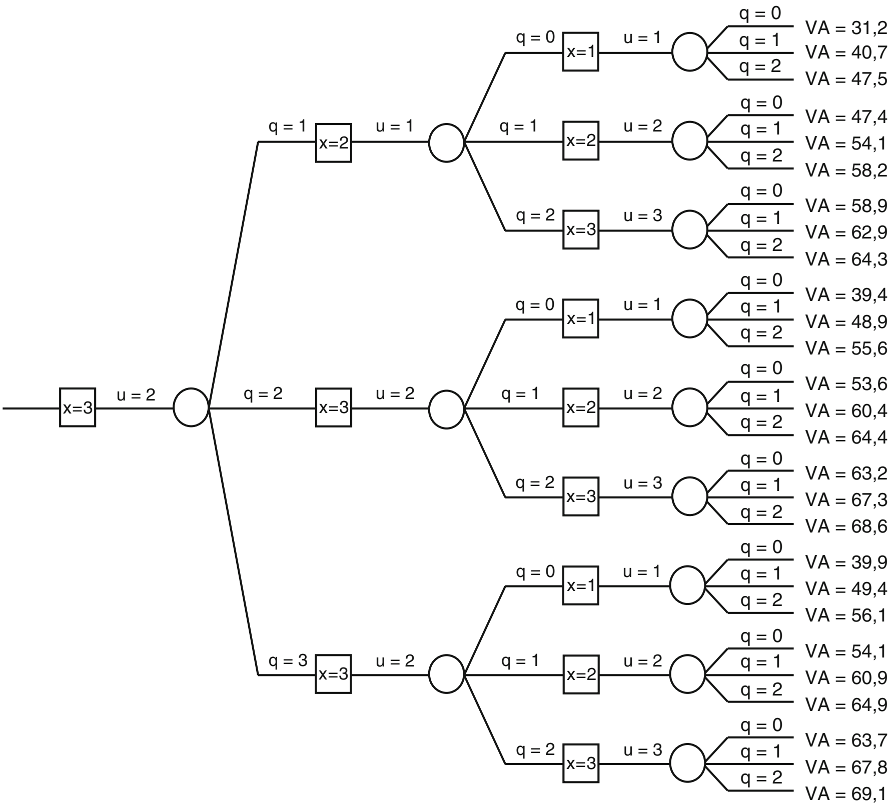


**Fig. 3.3** Decision tree of the stochastic problem

subject to
$$x_{2k} = x_1 - u_1 + q_{1k} \tag{3.55}$$

$$x_{3km} = x_{2k} - u_{2k} + q_{2m} \tag{3.56}$$

$$x_{4kmn} = x_{3km} - u_{3km} + q_{3n} \tag{3.57}$$

$$u_1 \leq x_1 \leq 3\,; \quad u_{2k} \leq x_{2k} \leq 3\,; \quad u_{3km} \leq x_{3km} \leq 3 \tag{3.58}$$

$$x_1 = 3 \tag{3.59}$$

with non-negativity conditions for the variables.

The results obtained with this method are the same as those obtained with the dynamic programming method. The main difference is that dynamic programming permits obtaining the optimal sequence of decisions for any initial state of the system, whereas discrete stochastic programming only gives the solution for $x_1 = 3$ (Fig. 3.3):

### 4.2.3 Resolution by Recursive Stochastic Programming

This method consists in solving a series of inter-temporal optimisation problems.

In the first iteration, we make the hypothesis that the agent reasons in terms of expected rainfall values. Thus we solve the problem by:

$$\text{Maximize} \quad \sum_{t=1}^{3} \rho^{t-1} \, r_t \tag{3.60}$$

$$\text{subject to} \quad 0 \leq \; u_t \leq \; x_t \leq \; 3 \qquad u_t, x_t \;\; \text{integers} \tag{3.61}$$

$$x_{t+1} = \min\{ \, (x_t - u_t + q_t) \; , \; 3 \, \} \tag{3.62}$$

$$x_1 = 3 \tag{3.63}$$

Once the first decision has been carried out ($u^*{}_1$), uncertainty related to the rainfall of the first stage will be cleared (one of the possible states of nature $q_k$ will occur), which will affect both the revenues generated $r^*{}_{1k}$ and the state of the system at the beginning of the second decision stage ($x^*{}_{2k}$). These relations are given by the following recursivity equations:

$$r^*_{1k} = 0.1 \;\, b_1 \left[ u^*_1 + q_{1k} - 0.1 \; (u^*_1 + q_{1k})^2 \right] \tag{3.64}$$

$$x^*_{2k} = x_1 - u^*_1 + q_{1k} \tag{3.65}$$

The agent will find himself in one of the states of nature $x^*{}_{2k}$ with a probability $p_k$ instead of finding himself in the state expected $x_2$. The agent can revise his decisions for the following stages as a function of the state reached for the system (which will depend both on the decisions taken and on the rainfall during the stage).

The second iteration permits determining the optimal decisions throughout the remaining planning horizon given the state of the system and the expected rainfall values. The second iteration therefore consists in solving the dynamic problem shifted by one stage for each initial value $x^*{}_{2k}$:

$$\text{Maximize} \quad \sum_{t=2}^{3} \rho^{t-1} \, r_{tk} \tag{3.66}$$

$$\text{subject to} \quad 0 \leq u_{tk} \leq x_{tk} \leq 3 \quad u_{tk}, x_{tk} \;\; \mathit{integers} \tag{3.67}$$

$$x_{t+1,k} = \min\{ \, (x_{tk} - u_{tk} + q_t) \; , \; 3 \, \} \tag{3.68}$$

$$x_{2k} = x^*_{2k} \tag{3.69}$$

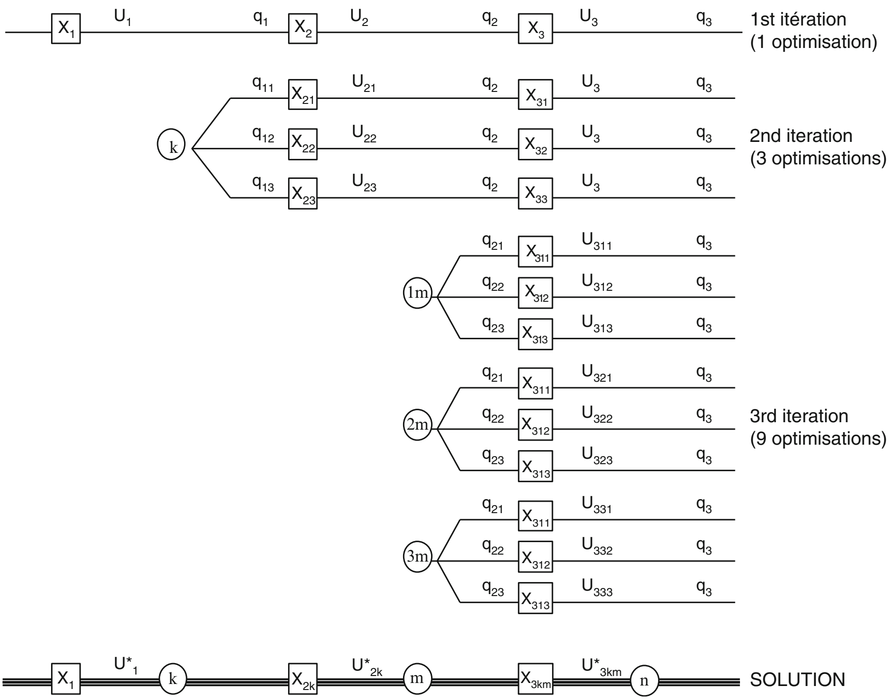


**Fig. 3.4** Diagram of the resolution by recursive stochastic programming

We shall now only take into account the results obtained for the second stage ($u^*{}_{2k}$) and determine the revenues generated $r^*{}_{2km}$ and the state of the system at the beginning of the following stage ($x^*{}_{3km}$) as a function of the state of nature having occurred ($m$):

$$r^*_{2km} = 0.1 \ b_1 \left[u^*_{2k} + q_{2m} - 0.1 \ (u^*_{2k} + q_{2m})^2\right] \tag{3.70}$$

$$x^*_{3km} = x_{2k} - u^*_{2k} + q_{2m} \tag{3.71}$$

The third iteration permits determining the optimal decisions throughout the remaining planning horizon given the state of the system ($x^*{}_{3km}$) and the expected rainfall values (cf. Fig. 3.4).

In our example, the results obtained with recursive stochastic programming coincide with the results obtained with the previous methods. This is not, whatever the circumstances, the general case since the problem's information structure is different. In fact, when the solution to the stochastic problem is obtained by using continuous and non-discrete state and control variables, the results are not the same. The discrete stochastic programming model gives an expected value of 57.258, whereas the recursive stochastic programming model gives 57.18. Nonetheless, it should be borne in mind that this value is close to that obtained with discrete variables (57.1).

On reaching this point, the reader may ask, “What is the point of using this “forward” recursive method instead of the “backward” recursive method? The advantage of this method is that it does not require using discrete values for state and control variables and it allows us to incorporate a large number of variables and equations into the model. We try to clear up this point in the next section.

## 5 Application to Assess the Sustainability of Farming Systems

The aim of this study was to evaluate the sustainability of farm irrigation systems in the Cébalat district in northern Tunisia. It addressed the challenging topic of sustainable agriculture through a bio-economic approach linking a biophysical model to an economic optimization model. A calibrated crop growth simulation model (CropSyst) was used to build a database to determine the relationships between agricultural practices, crop yields and environmental effects (salt accumulation in soil and leaching of nitrates) in a context of high climatic variability (Belhouchette et al. 2008). The database was then fed into a calibrated recursive stochastic model set for a 10-year plan, that allowed us to analyse the effects of cropping patterns on farm income, salt accumulation and nitrate leaching (Belhouchette 2004). We assumed that the long- term sustainability of soil productivity might be conflict with farm profitability in the short-term.

### *5.1 Study Area: Farmers’ Production Strategies*

The Cebalat area totals 3,200 ha and was created in order to reuse wastewater as irrigation for fodder and cereal crops near the capital, Tunis. However, the use of treated saline wastewater in combination with a saline and shallow water table increased the risk of soil degradation (Hachicha and Trabelsi 1993; Belhouchette et al. 2008). Long-term meteorological data indicated that the region is characterised by irregular and variable seasonal and yearly rainfall. The mean annual rainfall for the period from 1970 to 2000 is 475 mm/year ($\sigma = 133$ mm/year). Furthermore the distribution varies between fall (from September to December the mean rainfall is 201 mm), winter (from January to April the mean rainfall is 259 mm) and spring-summer (From May to August the mean rainfall is 25 mm).

Faced with biophysical conditions (climate, soil, water quality…), farmers are often forced to take decisions concerning: (i) the type of winter crops, their management practices and their allocated area will depend on the amount of rainfall in the fall period (September to December), i.e. often more barley and oats are cultivated than soft and durum wheat when the fall proves to be particularly dry. In addition, complementary irrigation is usually required in the fall period for sowing durum wheat, and (ii) the summer forage area depends mainly on fall and winter rainfall, i.e. after a wet fall and winter, the summer forage area is reduced as the production of winter forage (mainly berseem and alfalfa) is sufficient for summer feeding. After a dry fall or winter, farmers generally cut or harvest the cereals to use as forage during the spring and summer period and increase the surface area of forage crops during the summer.

To simulate farmer decisions and analyse cropping system behaviour and performance in the short and long-term, the recursive stochastic programming (RSP) method was used. The first year of the planning horizon was divided into two decision steps. In the first step (fall), farmers allocate areas with winter crops (oats, barley, wheat, beseem) before knowing the amount of rainfall for the winter period. In the second step (winter), the amount of fall rainfall is known and it is on this basis that the farmer decides on the type and the area allocated to spring-summer crops (alfalfa, maize and sorghum, each of which may be sown either for fodder or grain). Both steps in the decision-making process are modelled. In the first step, the farmer decides on the cropping pattern, the cropping management parameters (the amounts of irrigation and nitrogen fertiliser), and the area allocated to each crop, while taking rainfall probability into account. In the second step, decisions concern the cropping pattern and the cropping management parameters (amounts of irrigation and nitrogen fertiliser) for the spring-summer period.

The only source of uncertainty is the rainfall during the two periods, i.e. fall and winter. Rainfall variability is taken into account using associated probabilities of occurrence as described in Sect. 5.4, assuming that the probability of each rainfall event does not depend on the previous period.

## 5.2 Bio-economic Farm Model

For this study a bio-economic farm model was developed in order to assess the economic and environmental impacts of agricultural and environmental policies and technological innovations on farm and crop system sustainability (Belhouchette 2004). It is a dynamic model which optimizes an objective function to determine which decisions are taken, over a time frame of years. It is a primal-based approach, in which technology is explicitly represented (Louhichi et al. 1999), using engineering production coefficients generated from biophysical models (Hengsdijk and Van Ittersum 2003). These engineering coefficients constitute the essential linkage between the biophysical (CropSyst in this study) and economic models.

Concretely, the bio-economic farm model has a time horizon of 10 years, assuming that long-term decisions are taken according to rainfall probability. In the first simulation year, we considered two decisions, respectively on crop area, crop management and crop products in the fall and winter, and three states of nature for each decision. The main decision variables are the area in which each crop is managed (vector $X_{cpkit}$), taking into account the previous crop (p), a state of nature (k) and crop management (i) during 1 year (t).

For each activity, the crop yields are adjusted over the years for soil salt accumulation by means of the following equation (Mass and Hoffman 1977):

$$Y_{(c,p,i)} = \alpha_{(c,i)} + \beta_{(c,i)} * EC_{(c,p,i)}$$

Where Y is the crop yield (c) depending on the previous crop (p) and the amount of water and nitrogen applied (i); EC is the soil salinity, α and β are estimated for each crop by the CropSyst model and depend on the amount of water and nitrogen applied (i).

The objective function is written as:

$$\text{Maximize } NPV = \sum_k \left[ p_k * \sum_t \frac{Z_{kt}}{(1+r)^{t-1}} \right]$$

where NPV represents the excepted net present value, $p_k$ is the probability of state of nature k; r is the discount rate and $Z_{kt}$ is the farm income for each state of nature and each year (t).

The bio-economic model maximizes this objective function using a recursive process under three types of constraints.

1. Land constraints

   For the first year of the planning horizon, the following land constraints apply:

   $$\sum_{C_1,p,i} X1_{(C_1,p,k,i)} \leq S \quad \text{For t} = 1$$

   $$\sum_{C_2,p,i} X2_{(C_2,p,k,i,t)} \leq S \quad \text{For t>1}$$

   where X1 represents the area allocated to crop rotations in the first decision step (fall) of the first year, and *X*2 the area in the second decision step (winter) of the first year. In each case, the allocated areas must be less or equal to the total available arable land (S).

2. Transfer and rotation constraints.

   The transfer constraint indicates that the area (*X*2) allocated to each crop in the second decision step of the first year cannot exceed the area (X1) of the same crop in the first decision step of the same year while taking into account the previous crop (p), production techniques (i), and states of nature (k):

   $$\sum_{C_2} X2_{(C_2,p,k,i,t)} = \sum_{C_1} X1_{(C_1,p,k,i)} \quad \text{For t} = 1$$

   The rotation constraints also show that the area of each crop with a previous crop (p) for the year (t) cannot exceed the area allocated to this crop during year (t-1). $\sum_i X2_{(C_2,p,k,i,t)} = rot_{(p,t-1)}$; Where *rot* is the surface allocated to crop *p* during year (*t-1*)

3. Feed constraints
   In order to simulate the development of farm animals, the herd is represented as animal units. The number of animal units has been kept fixed for the entire 10 year simulation timeframe. The parameters and the regional technical coefficient for bovine breeding used for this analysis are those obtained for the 2000/2001 season.

The aim of this constraint is to guarantee an optimal feed ration capable of satisfying the energy demands for bovines by achieving a balance between the animal demands and the available forage resources.

### *5.3 Survey: Field and Farm Data*

In order to identify the main current activities in the investigated region (crop rotations and crop practices: fertilization, irrigation etc.) a survey, completed by local experts with the use of statistical databases, was carried out in 2000 (Belhouchette 2004; Belhouchette et al. 2008). In the Cebalat region 64 rotations were identified, with ten different crops. The principal rotations are soft wheat-maize, barley-sorghum, perennial alfalfa and berseem-fallow. Combined with the results of surveys on management information and climate-soil types, these rotations were defined as the current agricultural activities.

Management information collected for each crop included the different types, quantities, application dates and methods for inputs: sowing, harvesting and tillage events, water management, nutrient management, etc.

In addition, for each crop a set of economic data was specified including the 1999–2000 average producer sale prices and the variable costs. Variable costs were calculated by adding input costs for fertilizers, seeds, irrigation, biocides and the application costs associated with each management event.

Overall, the detailed analysis of the survey showed that farms in the studied area are homogeneous with respect to farm size, land use and farm specializations (Belhouchette 2004). Accordingly only one farm type was selected as being representative of all the farms in the studied area (Table 3.4).

### *5.4 Rainfall Variability and Classes*

The different rainfall probabilities for the RSP method were estimated using the statistical frequency analysis approach, based on a frequency curve with the ordinate of the curve being the magnitude of the event and the probability of rainfall excess as the abscissa (Hann et al. 1994). Seven rainfall patterns for the fall and winter periods were chosen based on rainfall probabilities:

– The fall period was considered to be wet ($F_w$), normal ($F_n$) or dry ($F_d$) if the total rainfall during the period was respectively between 227 mm and 189 mm, between 189 mm and 97 mm and below 97 mm.
– The winter period was considered to be wet ($W_w$), normal ($W_n$) or dry ($W_d$) if the total rainfall during this period was respectively between 260 mm and 175 mm, between 175 mm and 89 mm and below 89 mm.

## *5.5 Simulation Scenarios*

Two scenarios were developed and compared to analyse the effects of climate variability on farm decision (land use and crop practices) and three performance indicators: leached $NO_3$–N, accumulated salt in the soil and farmer's income. For both scenarios a 10 year-horizon was set and for each activity the crop yield was adjusted by calculating the soil salt accumulation over a number of years. The updating of the expected net present value is the only difference between the two scenarios. Details:

– The base scenario assumed that farmers prefer present to future income. For this assumption, the discount rate was set to 10%.
– The sustainable scenario assumed that farmers value the future as much as the present. For this assumption, the discount rate was set to 0%.

## *5.6 Main Results*

To prevent salt from accumulating in soil profile, leaching part of this salt is essential in irrigated agriculture, especially in arid regions (Ritter 1989; Rhoades et al. 1974; Smith et al. 1986). However, this strategy can be a source of a high nitrogen leaching. Table 3.4 gives a good illustration of this dilemma. Salt in soil is less important in the case of sustainable scenario than for the base one (except for the rainy sequence K01). This is mainly produced by a high level of irrigation. Moreover, the reduction of salt accumulation in soil observed for the sustainable scenario will be accompanied for most of crops by a yield increase (Van Genuchten 1993; Mass et al. 1999). In fact, when salt increase above a threshold level, both the growth rate and ultimate size of crop plants progressively decrease (Rajak et al. 2006; Botía et al. 2005). However, the threshold and the rate of growth reduction vary widely among different crop species. Some crops like maize, berseem are highly sensitive to the salinity (threshold about 2.5 dS/m). Other crops like wheat, sorghum and barley are more tolerant (threshold about 7.5 dS/m) (Mass et al. 1999). The increase of yield owed by the decrease of salt

**Table 3.4** Comparison of the amount of nitrogen fertiliser applied, irrigation, nitrogen leaching, soil salinization rate and gross margin for the two types of scenarios and for each rainfall sequence (Belhouchette et al. 2011)

| | Fertilisation (kg/ha) | | Irrigation (mm) | | Nitrogen leached (kg/ha) | | Salinization rate (dS/m) | | Grosse margin (Dinar/ha) | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Base | Sustainable | Base | Sustainable | Base | Sustainable | Base | Sustainable | Base | Sustainable |
| K01 | 131.2 | 139.0 | 1114.7 | 1015.1 | 18.0 | 19.4 | 5.1 | 3.8 | 19,840 | 19,570 |
| K10 | 138.6 | 155.2 | 713.9 | 860.6 | 15.4 | 22.8 | 8.9 | 5.2 | 15,636 | 15,998 |
| K11 | 133.0 | 144.1 | 673.9 | 885.5 | 4.4 | 4.4 | 8.5 | 4.8 | 14,425 | 15,177 |
| K12 | 130.9 | 146.6 | 653.9 | 934.3 | 14.9 | 15.6 | 7.7 | 4.6 | 13,375 | 14,676 |
| K20 | 128.0 | 128.6 | 389.8 | 851.5 | 9.8 | 10.7 | 8.0 | 5.6 | 13,597 | 13,823 |
| K21 | 127.4 | 124.3 | 342.9 | 760.1 | 4.7 | 6.2 | 8.5 | 4.9 | 14,388 | 14,415 |
| K22 | 130.8 | 131.3 | 660.2 | 863.6 | 9.6 | 11.9 | 7.2 | 4.6 | 13,045 | 13,105 |

in soil explains the difference of nitrogen implemented by the sustainable and the base scenarios. In fact, for all climatic sequences, more nitrogen fertilizer is implemented for the sustainable scenario than for base one. If we considered that numerous studies (Powlson 1988; Carpenter et al. 1998; Di et al. 2002) have indicated that leaching of soil NO3 – from the plant root zone to groundwater is mainly determined by two important factors: the amount of NO3 – used for fertilisation and the irrigation volume we can concluded that the high level of nitrogen leached observed for the 0% discount rate compared to 10% one is caused by those factors.

Our research revealed that quantities of water and NO3-N fertilizer used for the sustainable scenario are higher compared to the base one. In fact, for all climate sequences the gross margin is more important in the sustainable scenario than for the base one. This unexpected result is explained by the fact that profit caused by the increase of crop yields is higher than the costs of supplement of water and nitrogen observed for the sustainable scenario, procreated by the low costs of nitrogen fertilizer and water, because the prices of water and nitrogen in developing countries are usually greatly under priced (Tsur et al. 1995, 1977).

Overall, it also appears that in cases like this a high pricing of the irrigation water (as it is often advised) may induce negative effects concerning salinity. Reducing water use may induce higher soil degradation. This is of course a specific result concerning a particular situation, but it shows the dangers stemming from generalizing usual recommendations concerning these complex environmental and natural resources issues (Table 3.4).

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# Chapter 4
# Biophysical Models for Cropping System Simulation

**M. Donatelli and R. Confalonieri**

## 1 Introduction

The definition of mathematical models to estimate plants growth as a function of environmental variables has started many decades ago, for instance expressing the biomass growth of a plant as a function of the solar radiation intercepted (Warren Wilson 1967). Since then, crop models have evolved including sub-models to estimate plant development, and several other processes relevant to the simulation of the interaction plant-soil as affected by weather and agricultural management. Two main goals can be identified as drivers in plant model development: (1) studying the genotype $\times$ environment interaction, as a support tool to variety selection within a given species, or (2) studying production enterprises, hence comparing, from a biophysical point of view, yield, resource use, and externalities of agricultural production systems. Whether most of the models of the former group are specialized to a single crop, the latter includes multi-crop models to simulate crop sequences as in most production systems.

The study of the interaction genotype $\times$ environment has been performed via several types of modeling studies ultimately to assess either the potential or the actual performance of specific varieties, and to some extent providing breeders with an estimate of value of some plants traits desirable for a given environment (e.g. Hammer et al. 2002). This has been further extended targeting the modeling of crop improvement using a genotype $\times$ environment $\times$ management framework, via relationships gene-trait-phenotype (Hammer et al. 2005; Messina et al. 2009).

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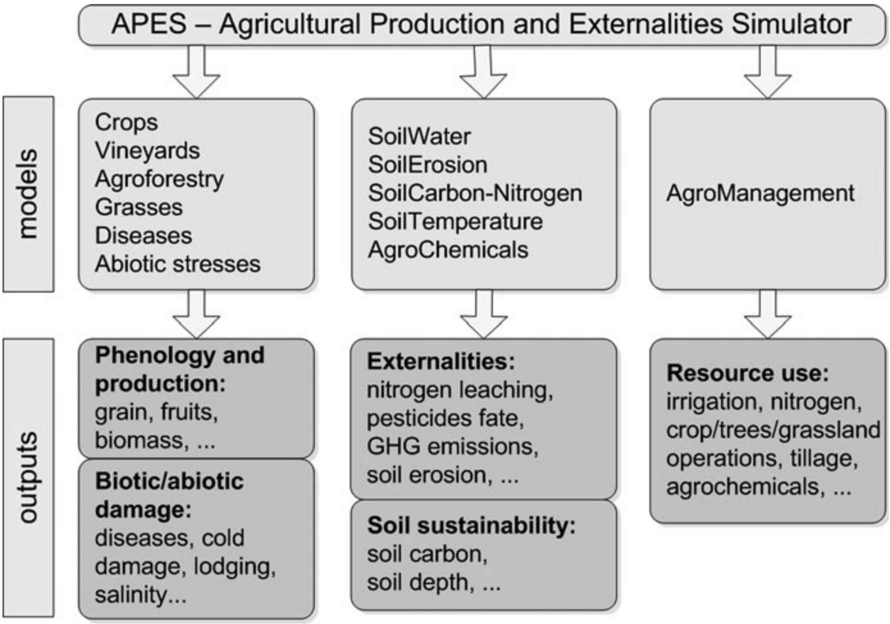


**Fig. 4.1** Main models and outputs of the agricultural production and externalities simulator

Several simulation tools allow estimating the impact of agricultural management on production activities in specific environments to be studied (e.g. Williams et al. 1983, 1989; Brisson et al. 2003; Keating et al. 2003; Jones et al. 2003; Stöckle et al. 2003; Van Ittersum et al. 2003). Because of their capability of simulating the dynamics of soil, water, and nutrients, in response to weather and agricultural management, such models allow exploring hypothesis of use of resources, and allow defining adaptation strategies to changing climate, to scenarios of resource availability, and to thresholds of externalities which may be set to limit the environmental impact of production systems. New developments in the technology to develop simulation systems have lead to modular software platforms OMS (David et al. 2002), TIME (Rahman et al. 2003), APSIM (McCown et al. 1996), APES (Donatelli et al. 2010) to allow for fine granularity model comparison, to facilitate the transfer from research to operational tools, and for an easier extension of the system being simulated by including new processes. As an example, the list of type of models and outputs for the system APES is shown in Fig. 4.1.

The objective of this chapter is to describe biophysical models for simulating agricultural production in order to highlight their capabilities and the assumptions in the perspective of using them in modeling chains.

## 2 Type of Models

Crop models originated by detecting the relationship between the amount of solar radiation intercepted by a crop and the biomass produced. A regression could be made stating that the amount of biomass produced by a plant is a function of photosynthetically active radiation intercepted by the crop. The knowledge of the biochemistry of photosynthesis confirmed that the cause-effect relationship could be correct, considering other factors non-limiting. In other environments, a similar relationship was observed between the amount of water used by a crop and its biomass production. Again, there was a biochemistry basis to justify the type of relationship. Both types of relationships proved to have good predicting skills in the environments and for the crops which were the origin of the data to develop the relationship. However, attempts to use such relationships in other environments, or even in the same environment but changing the context for instance in terms of soils, agricultural management, pressure from pests, soon showed that the relationships either did not hold anymore, or required new values of the parameters used in the relationship. Also, grain production showed to be much less predictable than biomass. New terms were tentatively added to the relationship above, to account for levels of factors observed, such as type of soil, level of fertilization, etc. At their best, such relationship could describe known systems exposed to specific weather patterns, but again could not be used either to model crop production in other environments or to explore management options. The reasons for such limitations were basically two. Firstly, the type of model built was a fully empirical model in which the level of empiricism was at the same level of the prediction. Secondly, the parameter(s) of the model did not have any biophysical meaning and encapsulated a set of efficiency factors peculiar of the system under study.

A production system based on a sequence of crops is sketched in Fig. 4.2. The dotted box can represent the approaches described above in this paragraph: a part of system is isolated, and the response of the crop is presented as a function of one or two regressors (e.g., solar radiation intercepted, water reaching the crop as irrigation and rainfall). However, that type of response holds in the *system* (contained by the dashed box of Fig. 4.2) in which that sub-system is included. For instance the sequence of crops leads to a given load of weeds, together to the typical agricultural management of the area; it implies, also as response to agricultural management, the nutrient balance and likely the structure of the soil. In other words, it sets the *state of system* prior to the crop, affecting its performance. Further, other processes occur which set the response to a given factor; for instance, the response to nitrogen fertilization in maize is not the same for different levels of water availability. Water availability itself is the result of rainfall patterns, soil permeability and slope, which may lead to runoff, and of soil evaporation, which is also function of soil type, soil surface management, possibly tillage, and so on. Several processes are interlinked and their outcome is a function of the state of the system at the beginning of a simulation step and of exogenous variables, which allow computing the *rate* of change for each state.

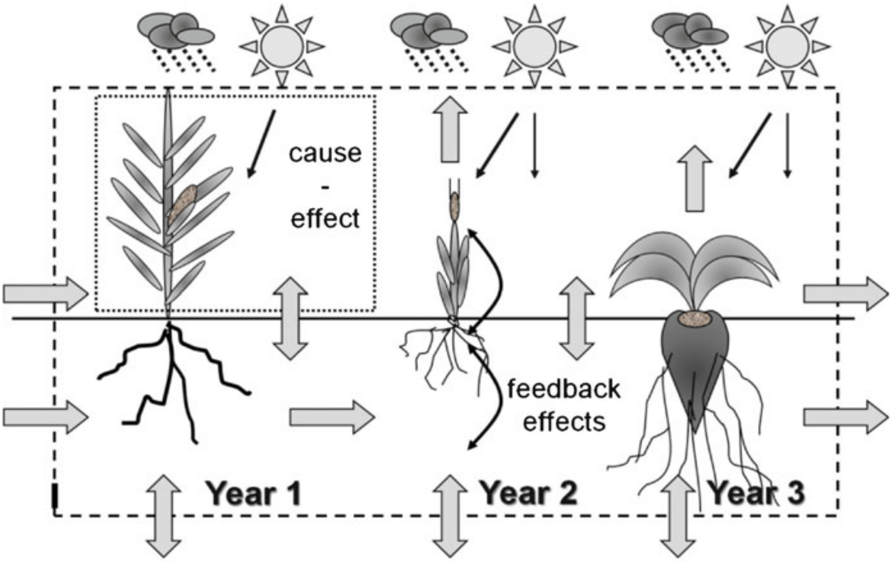


**Fig. 4.2** An agricultural production system based on sequence of crops: reductionist and system approaches (see text)

The state of the system is then updated at each simulation step, creating an artificial history of the system which represents its behavior and leads to the estimate of summary variable of interest. The simulation step duration, called time step, is a function of the processes being simulated and must be short enough to allow capturing variations of the system (e.g., if a crop can wilt irreversibly in a week, a monthly time step cannot be used). In crop and cropping system models the time step is frequently 1 day (i.e., all processes are simulated every day), or even 1 hour or less. The system is governed by several relationships among processes, non-linear by their nature. The point is that there is no mathematical function with an analytical solution to represent the dynamics of the system. Hence, static models of the type response curve functions to single factors cannot be used to estimate the behavior of the system outside contexts in which all other factors are fixed, and the impact of weather is accounted for via the empirical parameter which sets the efficiency of the use of the factor considered. In other terms, even having calibrated functions for all production factors of interest, such response functions can neither be used to explore new systems building management strategies, nor to estimate system behavior in different environmental contexts.

When the goal is studying the response (as yield, resources use, externalities) of the system crop-soil to weather and agricultural management, the modeling solution to be used must include the simulation of the processes which influence the response of interest. The relationships used, which are described in the

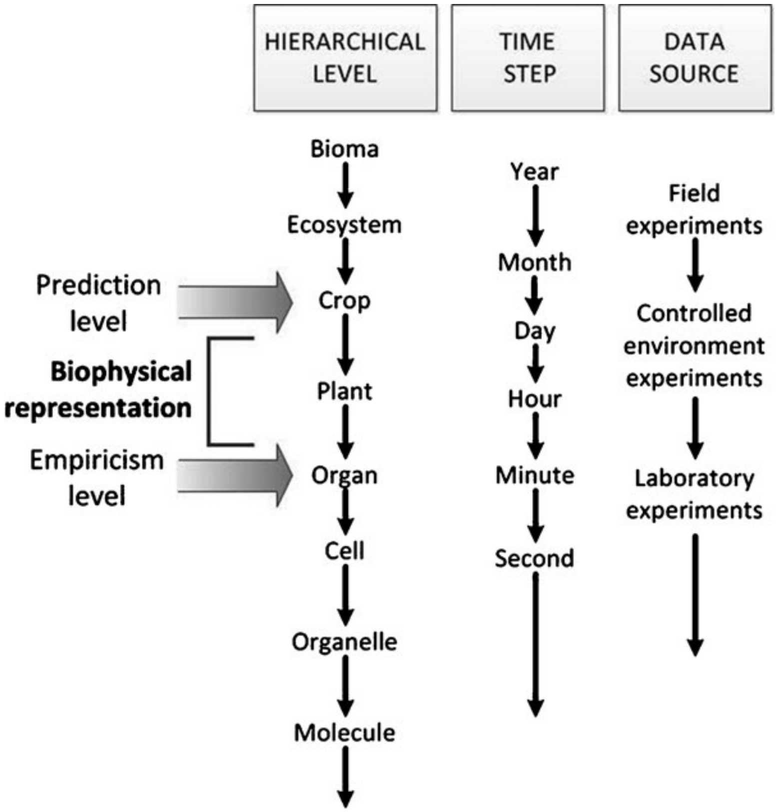


**Fig. 4.3** Level of prediction and level of empiricism in process-based models (Redrawn from Acock and Acock 1991)

following paragraphs, will also have some empiricism, but that empiricism will be one or more levels below the level of the prediction (Acock and Acock 1991), as shown in Fig. 4.3. The aim in defining new models is to use relationships known from physics or chemistry, and parameters which have either a biological or physical meaning. A process-based model can, in principle, be used to extrapolate to conditions outside the ones used to develop it, whereas a fully empirical model, as any statistical model, can be considered usable only for the context which originated the data used to build it. The main features on these two type of models are summarized in Table 4.1.

Process-based models are deterministic, that is they provide one set of outputs for one set on inputs. However, they are run in a stochastic fashion, that is against a sample of weather. Outputs can then be presented as average responses and probability distributions, and risk estimates can be derived, which can be used in subsequent modeling steps in a modeling chain for integrated evaluation of agricultural systems.

**Table 4.1** Main features of fully-empirical and process-based models

| Empirical models | Process-based models |
|---|---|
| Forecasting value strictly related to the representativeness of the data used to develop the models | Simplified description of the system including sub-models accounting for drivers of change given their purpose |
| Valid exclusively in the context from which the data were collected | Can be used in conditions different from the ones used to develop them, if system analysis confirms their conceptual validity |
| Neither explain the system nor its behavior | Provide insight on system behavior when shocked |
| Can be robust for very specific conditions | Allow analyzing the system dynamically |
| | Provide outputs for multiple-research questions |

The following paragraph will illustrate the most common approaches to model biophysical processes in agriculture, and assumptions, problems and techniques to use them in simulation experiments.

## 3 Modelling Crop Growth

Different macro-processes can be identified, analyzed and formalized separately to get an effective representation of crops dynamics. Crucial biophysical relationships exist among such macro-processes and between the macro-processes themselves and the environmental and management factors driving crops growth. It is possible to identify five main macro-processes: development (the progression of a plant moves through different phenological phases), potential growth (the plant increases in terms of dry weight driven only by temperature and solar radiation), and processes involved with water limitation, nitrogen limitation, impact of biotic and a-biotic stresses.

Crop development is, in most of the cases, simulated as a function of the thermal time accumulated between base and maximum temperature, optionally corrected to account for photoperiod and vernalization, that is, a genetic control which requires a given number of could hours to progress in development. The transition of the crop from a certain phenological phase to the following one is based on the attainment of specific thermal time thresholds.

Since the end of the 1960s, different approaches for modelling potential crop growth were developed. The first approaches derived from the pioneering work of C.T. de Wit and colleagues in Wageningen, which led to a family of models aiming at formalizing the knowledge available on the relationships between crops and environment. The most known models belonging to this family are SUCROS (van Keulen et al. 1982), and the derived WOFOST (van Keulen and Wolf 1986) and ORYZA (Bouman et al. 2001). These models are based on the simulation of net carbon fixation as a balance of gross $CO_2$ assimilation and maintenance and growth respiration. Gross assimilation is calculated each day by integrating the instantaneous $CO_2$ assimilation rates computed at three moments of the day and for three (or five) canopy depths.

Maintenance respiration is based on two assumptions: (i) different plant organs have different respiration to weight ratios and (ii) respiration is proportional to the dry weight of the organs themselves. Growth respiration is considered dependent from the different chemical composition of leaves, stems, storage organs and roots. Total dry matter accumulated each day is partitioned among the different plant organs according to a fixed scheme of development-dependent coefficients. Leaf area is considered growing exponentially as a function of temperature during early growth, whereas it is derived from leaf biomass using a development-dependent specific leaf area after canopy closure. Leaves death is simulated as a function of leaves senescence and self-shading.

The approach to crop growth implemented in these model is conceptually sound and detailed enough to draw attention to gaps in understanding and to allow the analysis of biophysical processes at the level of plant components (Confalonieri and Bechini 2004). However, the high level of detail used by these models to represent plant morphological and physiological features implies a high requirement in terms of information needed for their parameterization and, thus, on the effort required to use them operationally (Monteith 1996). Moreover, this approach has not always proved to give advantages when biomass estimates were evaluated against reference data (Jamieson et al. 1998). This is why simplified approaches to biomass accumulation have been proposed in the last decades, when the attention of part of the modelers community moved from the formalization of knowledge to the development of tools for evaluating alternate management scenarios at field and farm level and, more in general, for supporting decision-making. These approaches, developed mainly during the 1980s and 1990s, formalized daily biomass accumulation using the concept of net photosynthesis. In this case, no respiration is simulated, and daily accumulated crop biomass is considered proportional to one (or both) of two main driving factors involved in the photosynthetic carbon fixation, i.e. intercepted radiation and transpired water (Confalonieri et al. 2009a). Potential aboveground biomass accumulation is linearly related to cumulative light interception (Monteith 1977), with the slope of the linear relationship being the radiation use efficiency in models based on intercepted radiation. In this way, radiation use efficiency, optionally corrected for thermal limitation, senescence, saturation of the enzymatic chains, is considered a parameter for converting intercepted radiation into aboveground biomass. Most of the models developed in the last decades are based on this approach, e.g., part of the APSIM crop models (Keating et al. 2003), the models of the CERES-DSSAT family (Jones et al. 1984), the STICS (Brisson et al. 2003) and WARM (Confalonieri et al. 2009a) models. A different way to model biomass accumulation using the concept of net photosynthesis is based on the transpiration use efficiency (Tanner and Sinclair 1983). In this case, aboveground biomass is computed by multiplying a biomass/transpiration coefficient by the ratio of potentially transpired water to the mean vapor pressure deficit, based on the knowledge that a given amount of water transpired leads to a different amount of biomass synthesized according to the evapotranspirative deficit of the environment. Because of the division of the estimate of growth by the vapor pressure deficit of the atmosphere, this approach cannot be used in conditions where the vapor pressure deficit is modest. This is why the most widely used model implementing this approach, i.e., CropSyst (Stöckle et al. 2003), also computes each day a second aboveground biomass value using the radiation use efficiency-based approach and takes the minimum of the two. It is worth noting, referring to the concepts of simplified representation of the system, that the approach based on intercepted radiation was developed in environments where radiation was limiting, whereas the transpiration efficiency approach was developed in environments with atypical high evapotranspirative demand.

The simplification intrinsic into the concept of net photosynthesis led to simplifications also in the representation of other processes, like partitioning (e.g., growth respiration is not simulated anymore). Other simplification (e.g., leaf death only due to senescence, monolayer canopy representations) have been instead introduced into this typology of models to maintain a consistent level of detail in the representation of the different processes related with crop growth.

Once reference evapotranspiration is estimated via, e.g., the Penman-Monteith approach, crop potential evapotranspiration can be derived and successively split in the evaporative and transpirative terms basing on the leaf area and on synthetic information on canopy architecture. The transpirative term represents the atmospheric demand to the crop, which needs to be compared to the water available to plants in the soil explored by roots. In case of nitrogen limiting conditions, the stomatal resistance increases, thus reducing the amount of water the plant is able to transpire. The ratio actual (what the soil-root system is able to provide to the transpiring canopy) to potential transpiration (the atmospheric demand) represents the water stress index, and it is used to reduce – in case it assumes a value lower than one – potential biomass accumulation. Water stress, moreover, accelerates development, because of the assumption that low transpiration leads to a warmer plant (Stöckle et al. 2003).

In order to quantify the effect of insufficient nitrogen availability on growth, most of the crop models quantify the crop nutritional status through the definition of a critical nitrogen threshold, in turns used to derive maximum and minimum thresholds. Crop nitrogen demand is quantified as the amount of nitrogen able to lead crop nitrogen concentration to the maximum threshold. If the availability of nitrogen in the soil or the roots capability to uptake it are not enough, the amount of nitrogen actually uptaken leads to a crop nitrogen concentration which can be (i) below the minimum threshold (no growth in that day), (ii) between the minimum and critical thresholds (the crop is growing under nitrogen limiting conditions), (iii) or above the critical threshold (the crop is experiencing luxury consumption). The critical nitrogen threshold is usually derived using an allometric function which returns lower values for increasing values of aboveground biomass (Salette and Lemaire 1981; Justes et al. 1994). Other approaches derive the critical threshold from leaf area index (Confalonieri et al. 2011) or relate it to crop development (e.g., Williams et al. 1989; Hansen et al. 1991). The explanation for the time-decreasing trend of the critical threshold is due to leaves self-shading and to the allocation of photosynthates to plant organs with different nitrogen tenor: mainly to leaves (high nitrogen concentration) during early stages, to stems or other nitrogen-poor structures later. Self-shading leads the plant to realize that lower canopy layers are experiencing a decrease in the red/far red photons ratio because of a preferential chlorophylls absorbance of red photons in the upper layers. This leads the plant to start recycling nitrogen-rich compounds, not needed anymore, from shaded leaves and to re-allocate them to the youngest, thus lowering the nitrogen needs by the whole plant (Seginer 2004).

It is rare to find in the literature examples of cropping systems models implementing the effect of other factors affecting crop growth (i.e., biotic and a-biotic damages, or weed competition). Examples of the processes involved are the effect of diseases ultimately on net photosynthesis. The simplification of not modeling explicitly biotic and a-biotic damages can be accepted assuming an almost constant impact of adversities on plants; this assumption can hold, for both insects and diseases on plants, and for weeds, in high input systems where there is a chemical control. However, accepting a constant impact of a-biotic stress in changing climates or, more in general, simulating species and varieties non-adapted to the environment under study, generally implies an underestimate of the year by year variability of yield estimates. Approaches specific for these processes are described and discussed in a dedicated section of this chapter.

## 4 Modelling Soil Water

The goal of modeling soil water is primarily the one of estimating water available for plants over time, that is the water content between the lower limit, called permanent wilting point (plants are not able to extract water below this content, because it is strongly retained by chemical relationships to soil particles) and the upper limit, called field capacity (the soil water content that a soil reaches when it is allowed to drain freely). Whether different plant species vary noticeably in terms of extractive strength from soils, in the majority of cases this does not make changing substantially the amount of water that they can extract at very low soil water contents. Instead, field capacity is an abstraction and its value varies across soils due to both texture and organic matter content, and by soil structure, which can be strongly influenced by tillage and compaction.

A simplified representation of the processes involved with soil water balance is presented in Fig. 4.4.

Potential evaporation and transpiration are derived by partitioning the atmospheric evapotranspirative demand to a specific crop according to the amount of green leaves area present: the highest the green (transpiring) LAI, the lowest the evaporative term of the evapotranspiration. The atmospheric evapotranspirative demand to a specific crop in a specific moment during its cycle can be derived from reference evapotranspiration, the latter estimated using different methods, some more demanding in terms of data needs, e.g., the FAO Penman-Monteith (Allen et al. 1998),

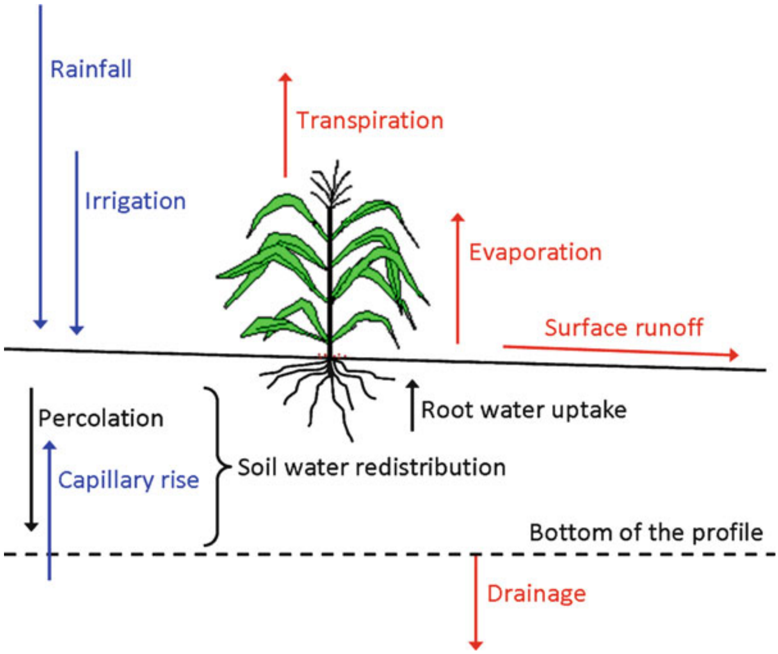


**Fig. 4.4** Schematic representation of input output fluxes of the system crop-soil

others more simplified, to be possibly used when data availability is not sufficient for using one of the formulation of the Penman equation. Among the simplified approaches, the most used are: Priestley-Taylor, derived by removing the aerodynamic terms form the Penman-Monteith approach and by adding an empirical term (Priestley and Taylor 1972) which is modeled as a function of the vapor pressure deficit (Steiner et al. 1991), Blaney-Criddle, recommended only where temperature data are the only available (Blaney and Criddle 1950), and Hargreaves, driven by solar radiation and daily temperature range (Hargreaves and Samani 1982).

Once potential transpiration and evaporation are estimated, the related actual terms are calculated according to the availability of water in the soil profile. For evaporation, only the top-soil is usually considered whereas, for actual transpiration – influenced also by the root capability to uptake water from the soil and the soil capability to cede water – the profile of the soil explored by roots is considered.

Runoff is the water not infiltrating but flowing on the soil surface because of rainfall (or of other water sources) exceeding the hydraulic conductivity of the first soil layers. Runoff (which also causes soil erosion) can be estimated using empirical and process-based approaches. The most used empirical method was developed by the USDA Soil Conservation Service (1972) basing on runoff observations collected in small catchments and hilly areas. The method is largely based on an empirical parameter, called the Curve Number, summarizing information on soil composition and structure, slope, land use, and hydrologic conditions. The relevance of this parameters led to associate the name of the parameter to that of the model. The most famous physically based approach for modeling surface runoff is based on the kinematic wave approach, obtained by combining the continuity and Manning's equations. Manning's equation provides the relation between water height and water flow. The Kinematic Wave derives from Sant Venant's equation assuming that water height and momentum effects are negligible. The equation can be solved numerically using different methods of calculation, one being the finite difference method. Water infiltration can also be computed by using different methods, ranging from the simple Green-Ampt approach (Green and Ampt 1911) to the improved Smith and Parlange (1978) method. The latter is used within some of the most mechanistic (i.e., physically detailed) models for soil runoff and erosion, i.e., KINEROS (Woolhiser et al. 1990) and EUROSEM (Morgan et al. 1998).

Different approaches are available also for the redistribution of water within the soil profile. The cascading one (also known as 'tipping bucket') is the simplest, although it is implemented in models widely used worldwide, like CERES (Jones et al. 1984). It assumes only downward water movements along the soil profile and that layers are filled up until field capacity is reached, with the fraction of water exceeding this threshold moving to the deeper layer (Jones and Ritchie 1990). The cascading with travel time approach is a modification of the cascading; in this case, the downward water movements can be reduced by soil hydraulic conductivity. This could lead water content in the layers above the least permeable to be higher than field capacity. This approach, adopted in various simulation models (e.g., SWAT, Neitsch et al. 2002; WARM, Confalonieri et al. 2009a) is often considered as a good compromise between parsimony in data needs and good representation of the real biophysical system. The most mechanistic approaches are based on solutions of the Richards' equation (Richards 1931), based on the concept that water flux between two points is driven by the pressure gradient between the points themselves, and it is a function of the hydraulic conductivity. Given that hydraulic conductivity does not vary linearly as a function of water content, the equation has no analytical solution, and is solved using numerical methods. This approach is the more demanding in terms of input data, needing water retention curves and hydraulic conductivity as a function of soil water content and/or water pressure. Nevertheless, it proved its reliability under a variety of conditions, leading to its implementation in many cropping systems models, like CropSyst (Stöckle et al. 2003), SWAP (Van Dam et al. 1997) and MACRO (Jarvis 1994). Although Richards' based approaches are the only ones considering explicitly capillary rise, semi-empirical approaches for its estimation were developed (Driessen 1986; Liu et al. 2006) to allow cascading-based models to account for the water table contribution.

Most of the modeling solutions for soil water dynamics do not account for preferential flow (as one of exception, the MACRO model by Jarvis 1994). This has a sizeable impact on the simulation of water redistribution in cracking soils, both for water content and for solute transport simulation.

## 5 Modelling Soil Temperature

Soil temperature has a crucial influence on the dynamics of many biophysical and biochemical processes, like the emission of greenhouse gases (Boeckx and Van Cleemput 1996), emergence of crops and weeds (Plauborg 2002), mineralization of soil organic matter (Leiròs et al. 1999), chemicals fate (Tsiros and Dimopoulos 2007), survival and dynamics of soil borne plant pathogens (McLean et al. 2001). Simulation of soil temperature is usually carried out by estimating first the temperature at the soil-atmosphere interface given by soil surface, and then the heat propagation along the soil profile.

Examples of widely used models for the simulation of surface soil temperature are the one proposed by Parton (1984) and the one implemented in the SWAT model (Neitsch et al. 2002). The former needs as input only daily air maximum and minimum temperature and day length, whereas the latter – more sophisticated – accounts for the effect of vegetation in intercepting radiation, thus requiring aboveground biomass, average temperature of the previous day for the first soil layer, albedo, global solar radiation and water equivalent of snow pack.

The simulation of soil temperature along the profile requires also the simulation of soil water content, because of the impact of soil moisture on the heat fluxes in the soil profile. Among the models for simulating soil profile temperature, the Campbell one (Campbell 1985) and the approach implemented in SWAT (Carslaw and Jaeger 1959) are probably the most used. The former is based on a finite difference solution of a differential equation describing the temperature flux along the soil profile as a function of water content and soil mineral and organic components; temperature conditions at the bottom of the soil profile are set to the average annual air temperature. The SWAT model estimates first a maximum damping depth as a function of bulk density, and then modulates this value by accounting for soil water content in order to derive the actual damping depth. Temperature for each soil layer is finally calculated as a function of average annual air temperature, temperature of the previous day, surface temperature and a depth factor.

Often albedo is not modeled hence used as a fixed value, and this is somehow surprising given the sensitivity of both soil temperature and water simulations.

# 6 Modelling Soil Nitrogen

The interest on modeling soil nitrogen is due from one hand on its importance for plant growth, and from the other to the fast dynamics in the soil of its mineral forms which are uptaken by plants. Some forms are not retained by the soil and transported by downward water movements in the soil profile. This has impact not only in terms of how critical nitrogen management is with respect to plant growth, but also on environmental aspects because of leaching and potential nitrous oxide emission as high impact green-house gas.

Modeling of soil nitrogen is extremely critical because of its general level of empiricism in most cropping system models, definitely higher than both the one of soil water and plant growth modeling. The level of empiricism is due to ignoring explicitly the dynamics of the soil microbial communities of fungi and bacteria, which govern the transformations from soil organic matter to mineral nitrogen, and which are responsible for transformations from one form of mineral nitrogen to another. The dynamics of nitrogen in the soil are modeled as responses to temperature and water which impact on microbial communities, via the proxies represented by various pools of organic matter. The SOILN model (Johnsson et al. 1987), uses three pools of organic matter. All transformations (except denitrification) are simulated with first-order kinetics. The three pools represent stable soil organic matter and added organic materials. The two labile (i.e., rapidly decomposing) pools are called “litter” (L) and “manure-derived-faeces” (M). The pool L contains crop residues and dead roots, while the pool M contains animal manure. Added organic materials are assigned in total or in part to pool L and/or M. The two labile pools allow managing very different materials: generally, materials with high C/N (carbon/nitrogen) ratio are assigned to L, and materials with lower C/N ratio to M. A third pool, decomposing slowly, represents the stable soil organic matter (H; humus). As discussed in the opening of this paragraph, no microbial biomass pools exist, as microbial biomass belongs to labile pools. The L and M pools therefore represent the association of added organic materials with their decomposers. The soil nitrogen model of the model CropSyst is sketched in Fig. 4.5 showing pools and input-output flows.

The reason why the level of empiricism of nitrogen modeling does not impede using these models in various environments is due to robustness of parameters according to soil typologies. However, model parameters have no biological meaning and cannot be used in modeling approaches different from the ones for which they have been calibrated for.

# 7 Modelling Abiotic and Biotic Stress

For many years, cropping systems modellers have been used to identify different production levels to formalize the driving and limiting factors determining crop production: potential (temperature, radiation and, in some cases, atmospheric

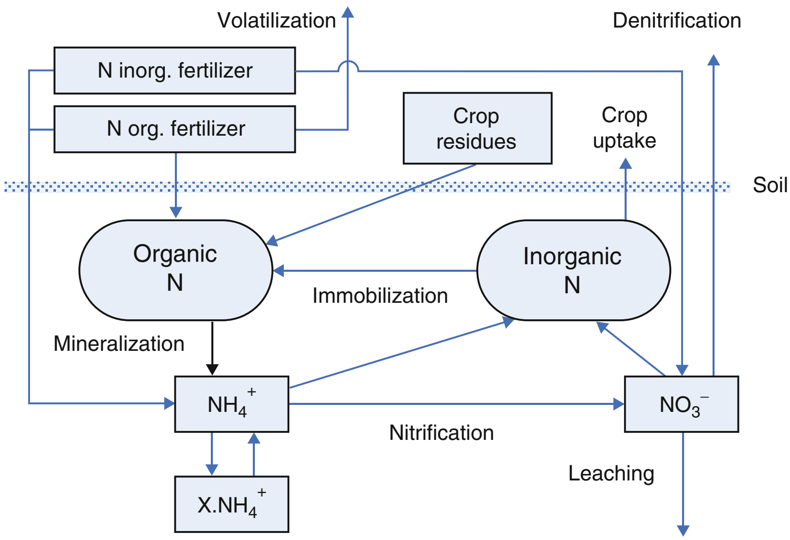


**Fig. 4.5** States and transformations in the nitrogen cycle as modelled by CropSyst

$CO_2$ concentration are basically the main driving forces), water limited (possible limitation due to water shortages are considered), and nutrients limited (practically, only the effects of insufficient nitrogen availability are considered). The other biotic (e.g., diseases) and abiotic (e.g., ozone concentration, frost events) factors affecting productions were traditionally considered as having a constant impact on the crops across the different seasons. In light of this (false) assumption, modelers deceived themselves to account for the impact of such factors by indirectly including their effects in the values of the parameters describing morphological and physiological plant features, thus avoiding the development of specific sub-models. This led (i) to develop crop parameter sets embedding factors other than crop features, thus partly degrading the process-based logic behind the model, (ii) to consequently develop site-specific sets of crop parameters, thus depriving the model of its capability of performing simulations under conditions different (in space or time) from those used during the parameters calibration. These consequences may strongly decrease crop models suitability for large-areas simulation or for evaluating the impact of climate change scenarios.

In recent years, modellers started looking at biotic and abiotic stressors with more caution: first, in spite of obvious differences in the susceptibility of different areas, the impact of such stressors within the same area is far from being constant. Second, abiotic factors are considered as driving variables since the real first steps of the modelling discipline. However, temperature, under certain conditions, can be considered as an abiotic stressors, leading even to crop failure. The role of weather variables in acting as abiotic stressors is rarely taken into account in cropping systems simulation models. As an example, air temperature is a driving variable involved in the simulation of plant development and photosynthesis limitation at any given level of intercepted solar radiation. Although with different algorithms, these processes are basically included in all crop simulation models. The reduction of leaf area index or the possible death of the plant due to frost are usually not included in crop growth models, as well as the yield losses due to pre-flowering cold or heat shocks inducing spikelet sterility, although these processes are driven by the same variable (temperature). These considerations are leading the international modeller community to approaching the development of models for the simulation of biotic and abiotic stressors in a more systematic way, although modules for the simulation of their impacts on crop yields are practically absent from almost the totality of cropping systems models.

It is possible to define abiotic stressors as "Environmental variables assuming values – or evolving with dynamics – for which a crop is not prepared (in a specific moment of its crop cycle)". Confalonieri et al. (2009b) identified six categories of damages due to abiotic stressors: temperature-shocks induced sterility, lodging, frost, ozone, salinity, and heat damages. Temperature-induced sterility damages usually occur when temperature falls below a threshold during pollen formation. A model accounting for pre-flowering sterility due to cold shocks is implemented in the WARM rice model (Confalonieri et al. 2009a), accounting for the different crop susceptibility during the period between panicle initiation and heading. A model of stem and root lodging was proposed by Baker et al. (1998), assuming wind induced bending moment at the stem base as predominant factor affecting lodging. In this model, the value of bending moment relative to the failure moment of the stem, and the failure moment of the root/soil system indicates whether or not lodging of either type will occur. A model for frost damages was proposed by Ritchie (1991) and included in the CERES-Wheat model (Jones et al. 1984). This approach calculates crown temperature (also considering the possible insulating effect of snow), hardening and de-hardening index, a killing temperature, the possible reduction in leaf area index, and evaluates if the crop has been killed by the frost. A complex model for ozone damages was proposed by Sitch et al. (2007), estimating the fractional reduction of plant production as a function of the ozone flux through the stomata and the leaf water conductance. Different approaches were proposed for salinity damages. The Ferrer-Alegre approach (Ferrer-Alegre and Stöckle 1999) is based on the calculation of plant conductance and then of a function for the estimation of salinity stress at different layers of the vegetation. The Karlberg approach (Karlberg et al. 2006) calculates the reduction of nutrients partitioned to the leaves due to salinity stress on the roots. A model for heat damages was proposed by Challinor et al. (2005). It calculates the critical temperatures, according to the sensitivity to heat stress of different groups of genotypes, the flowering distribution and the actual fraction of pods which set.

Biotic stresses to crops can be caused by weeds, diseases and pests. Algorithms for the simulation of yield losses due to the competition between crops and weeds for light, water and nutrients are practically absent from the most widely used cropping system models. However, attempts for developing specific models were carried out from the late 1980s, in some cases with satisfactory results. One of the most known approaches is INTERCOM, proposed by Kropff and Van Laar (1993). This model simulates the competition for light based on the amount of leaf area of the competing species and on how this amount is distributed within the canopy height, thus the radiation absorbed by the competing species is also a function of plant height. Soil water and nutrients are distributed between the competing species mainly as a function of their respective demands. Yield losses due to fungal diseases are estimated through the simulation of the plant-pathogen interaction, in turns needing the simulation of both the interacting species. The progress of the epidemics of fungal pathogens can be reproduced by considering the following weather-driven phases of the infection process: infection, incubation, latency, infectiousness, sporulation, and spore dispersal. The impacts on the crop (decrease in the photosynthetic efficiency, increase in leaves aging and leaves maintenance respiration) are modulated according to the disease severity. Pest induced yield losses is not addressed in most of the available cropping systems models (Aggarwal et al. 2005). In most of the cases where it is considered, the population dynamics are not explicitly reproduced, and field observations and scouting data on insect damages are given as input to the model which, then, calculate possible impact on crop growth, such as leaf area reduction or reduction of the assimilation efficiency (Boote et al. 1983). This kind of approaches demonstrated its reliability when field data are available, whereas it is intrinsically useless for large-areas simulations. The first approaches based on the simulation of single-species population dynamics were developed during the 1960s, followed by models for bitrophic interactions between pests and natural enemies. Crop and pest models – where only the impact of the pest population on the crop was simulated – were coupled during the 1980s, although the first examples of models for crop-pest interaction where pest and crops influenced each other in a dynamic way were developed during the 1990s (Chander et al. 2007). Currently, a large effort is being run to build libraries of plant disease models and coupling them to crop models, estimating the potential inoculums, the epidemiology, the plant damage, and the response to agricultural management (Bregaglio et al. 2009).

## 8 Modelling Agricultural Management

Farm management is the result of planned management for each production enterprise, and, during the growing season, of physical states at field level and resource competition at farm level. As an example, the irrigation scheme for each field at a given time leads to potential irrigations if the thresholds for irrigation set in the relevant rules are met. At farm level, such potential irrigations become quantities of water and labour required for each field, and they compete for the resources available. Rules for actions at farm level are a layer above the one at field level. Modeling agro-management at field level provides estimates of technical feasibility and performance for a production enterprise, whereas the simulation of agro-management at farm level allows estimating agro-management feasibility either in concrete farms or in farm abstractions such as farm typologies.

An agricultural activity is defined, in a modeling context, as a production enterprise such as a crop rotation, i.e., a sequence in time of crops, an orchard, etc., associated with a production system characterized in terms of outputs and inputs such as high input, high output (e.g., irrigated, high nitrogen fertilization, minimum tillage). Such an integrated system must be implemented in a way that imitates as closely as possible farmers' behavior. Limiting the drivers of the decision making process to the biophysical system implies that each action must be triggered at run time via a set of rules which can be based on the state of the system, on constraints of resource availability, or on the physical characteristics of the system. The model framework Agro-management (Donatelli et al. 2006; Donatelli et al. 2011) formalizes the decision making process in models called rules which respond to states of the biophysical system. The rule-based model is characterized by three main sections: (1) Inputs: states of the system and time, (2) Parameters specific for each, and (3) A model which returns a true/false output.

One feature of interest is that implementing the rule approach allows the formalization of what is generically referred to as "expert knowledge". For example, expert knowledge which suggests that in a specific environment, a farmer will "plant maize on a *date later than April 1st*, if it has not rained for the *last 3 days*, and when average air temperature has been above *5°C for 7 days continuously*" can be formalized and used in simulations. The italicized words are the parameters of the rule to be compared with system states/exogenous variables at run-time (e.g., the condition "no rain for the last 3 days" is tested against the values of rain at run time starting from April 1st as in this example). The possible uses of such formalization include building a consistent quantitative database of agricultural management, optimizing parameters in climate change scenarios as an adaptation strategy and using such metrics in climate change impact assessments, and improving technical management in current conditions through rule-parameter optimization.

Agro-management has been traditionally studied using the trial-error approach. The formalization of rules based on the state of the system in an extensible way opens to optimization and to a gain in the understanding of the system responses to agricultural management.

## 9 Parameterization

Parameters of biophysical models can be defined as quantities which are set during the initialization step of a simulation and either do not vary during simulation, or vary in response to events (e.g., the specific crop parameters are loaded when a crop is planted in a simulation of a rotation).

Parameters can be classified in two types, that is, parameters which either have a biological meaning and can be either measured or derived from the system, or parameters which have an empirical basis because they summarize different factors. The former group includes quantities that in some simulation approaches can be states of the systems, instead of parameters (e.g., soil bulk density). The latter group de facto sets the assumptions of using the model extrapolating from tested conditions; limits are not as strict as for fully empirical models as discussed in the paragraph *Type of models*, but represent a critical step in designing a simulation experiment.

Parameterization of biophysical models is the activity of determining the values of their parameters. Biophysical models are in fact sets of interrelated laws describing the behaviour of the modelled system, which are assumed to be valid for the general representation of a group of coherent entities, e.g., for a crop model, species or varieties with similar morphological and physiological features. The adaptation of the biophysical laws to the features of one of these entities is carried out by assigning to the parameters of the laws specific values. This can be achieved via two different strategies: measuring the values of the parameters or calibrating them by forcing model outputs to reproduce observed states (or, more rarely rates) of the real system.

The first solution, i.e., direct measurements of the model parameters, should be preferred, although (i) uncertainties in the measurements themselves, (ii) the presence of empiric parameters (i.e., parameters without a biophysical meaning), (iii) and the frequent need of parameterizing a model for a huge number of different entities often make this solution at least partially unfeasible. This is why, after defining the values of certain parameters, that is the parameters for which no significant changes are expected among the simulated entities, the other are calibrated.

Calibration consists in adapting a model to one or more sets of measured data to allow its application to similar conditions (Beck 1987). It is key that the reference data used for calibration contain sources of variation which are modelled by the model being calibrated. The use of large area statistics to calibrate the type of models described above should be approached with extreme caution, regardless of the apparently good numerical output that a times can be obtained. Although calibration is widely applied, it could be a very risky procedure (de Wit and Penning de Vries 1982). Calibration, in fact, is aimed at determining the values of uncertain parameters via the their effects. This could lead to reduce a parameterization process to a pure fitting exercise, in turns degrading a biophysical model to a totally empirical one, very similar to a regression model but without any statistical support. This because the calibration of a parameter can be affected by errors in the estimation of others, by errors in the conceptual representation of the system, by effect of correlation with other variables, etc. These risks increase exponentially along with the number of parameters under calibration.

To avoid incoherent model behaviours resulting from improper calibration procedures, there is the need of defining rigorous, low-risk criteria for each of the following steps (Acutis and Confalonieri 2006).

1. Observations representative of the process the parameters under calibration refer to should be selected. Parameters involved with the simulation of a process should be calibrated independently from the others.
2. Only unknown parameters should be calibrated. In case measured data are available, they should be used or the calibration should be carried out only within the range of the measurement error.
3. The number of parameters to be simultaneously calibrated should be kept to the minimum: advanced sensitivity analysis techniques (e.g., Confalonieri et al. 2010a) should be used to identify the parameters with the highest impact on the output(s) considered and calibration should be concentrated on them.
4. All the available information should be used to reduce the physical domain of the parameters, and the calibration must be carried out only within the parameters physical domains themselves. Unrealistic parameters values often indicate that the calibration is affected by errors in the values of other parameters.
5. Calibration results should always be tested against independent observations.
6. Objective functions (i.e., simple or complex performance metrics) should be evaluated on more variables (e.g., aboveground biomass, leaf area index). This ensures stability and coherence in simulation.

From the theoretical point of view, the *perfect* calibration allows for ‘model inversion’, where a model is used to retrieve information about an unknown (or uncertain) physical parameter by considering the model itself exactly as a laboratory analysis (Romano and Santini 2002). The criteria needed to obtain the value of a physical parameter from a model inversion are the following. (i) Uniqueness: there is only one minimum (or local minima are clearly different from the absolute one). (ii) Identifiability: different sets of parameters give a large difference in the value of the objective function. (iii) Stability: small changes in measured data do not affect significantly the parameter values. When a model inversion has an unstable and non unique solution, it is considered an ill-posed problem, that can become well-posed by increasing the quality and quantity of observations or using tighter constraints to parameters.

Since these criteria are not fulfilled in many biophysical modelling applications, different optimization algorithms were developed in the last decades. These algorithms can be classified according to the fact that they use derivatives or not. The most relevant drawback of derivative-based method is that, if analytical partial derivatives are not available for all the parameters, the computation must be carried out numerically. On the other side, algorithms not based on derivatives are useful with complex models, very easy to understand and often decidedly powerful (Smyth 2002). Further, model inversion should not be done in biophysical system given the nature and the amount of parameters involved.

Some of the most used optimization algorithms are summarized in Fig. 4.6.

Apart from automatic optimization algorithms, it is possible to carry out calibrations using the grid search criterion. This method consists in five steps: (i) select the parameters to calibrate, (ii) set a domain for each parameter,

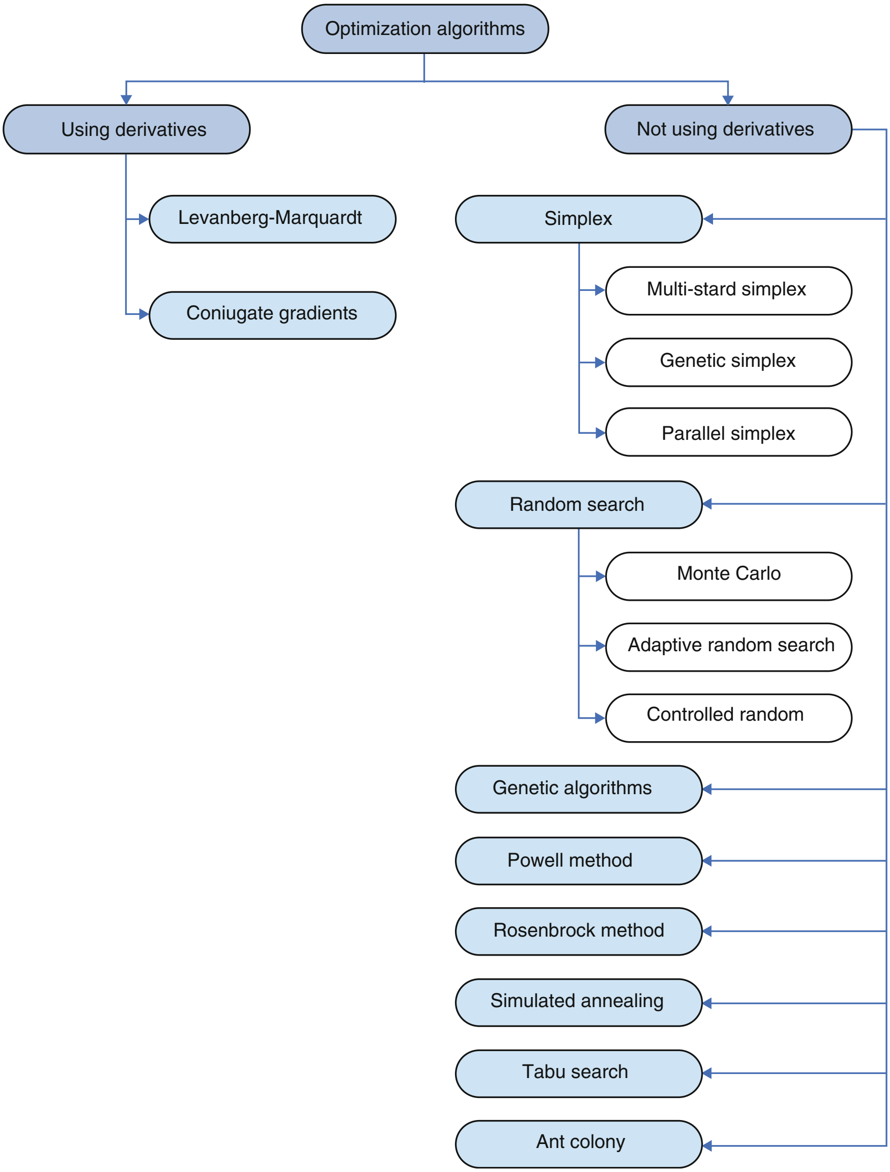


**Fig. 4.6** Flow diagram of the optimization algorithms (Redrawn from Acutis and Confalonieri 2006)

(iii) divide the domain in user-defined parts, (iv) evaluate the objective function at any node of the grid, (v) select as calibration result the node corresponding to the best value of the objective function. Grid search is undoubtedly simple and reliable but time consuming and not practical in most cases, especially for ill-posed problems. For sure, this method is useful for the exploration of the hyperspace defined by the parameters to calibrate.

Although the just mentioned technology for performing automatic calibrations, trial-and-error is still the most diffused method for calibrating the parameters of cropping systems models. The only advantage of this method is the fact that no specific software is needed. The drawbacks of trial-and-error are the time required and the risks associated to this procedure: the user changes many parameters, often simultaneously, without a strategy, hoping that a long series of small changes allows obtaining good fittings. In many cases, final results are not satisfactory: unrealistic combinations of parameters are reached or poor agreements between observations and simulated data are achieved.

Among the different optimization methods which do not use derivatives, the downhill simplex (Nelder and Mead 1965) is one of the most used when the figure of merit is "get something working quickly" (Press et al. 1992). In case of complex hyperspaces, the risk that it falls in local minima is overcome by its evolutions, like the parallel simplex (Matsumoto et al. 2002) and the evolutionary shuffled simplex (Duan et al. 1992).

Whether following correct procedures as the ones summarized above is required, they will not be able to overcome problems in the structure of the model, which may lead to noticeable autocorrelation and hence to multiple solutions, in any case instable as in a problem of multicollinearity.

## 10 Evaluation

Model evaluation is assuming the status of an autonomous discipline and it is decidedly catalyzing the attention of a relevant part of the modellers community (Rykiel 1996; Bellocchi et al. 2009; Confalonieri et al. 2010b). Recently, modellers have started preferring to use specific evaluation procedures (e.g., Bellocchi et al. 2002; Donatelli et al. 2004b) for facing with specific modelling problems with respect to looking for an ideal and unique methodology for the evaluation of biophysical models. This is explained by the variety of applications models can be involved with, which in turns leads the modellers focusing on, e.g., the level of detail used to reproduce biophysical processes, the degree of overparameterization or the model suitability for large-area simulations (Confalonieri et al. 2010c).

Regardless of the type of study for which the use of a model is required, the following steps should be followed to evaluate the available modelling approaches and to select the most suitable among them:

1. Identify clearly the aims of the study (i.e., what exactly the model should do) and the conditions of application (e.g., spatial scale, data availability);
2. Derive from step 1 a quantitative criterion to evaluate the available model on the basis of their suitability;

3. Use the quantitative criterion defined at step 2 to rank the models available;
4. Select the most suitable model according to the specific criterion derived (i.e., the most suitable model according to the aims of the study and to the specific conditions of application).

Van Ittersum et al. (2003) identified a number of possible model objectives (e.g., research, making prediction, decision making) and conditions of application, mainly related to the spatial scale and to the level of detail of the model in relation to the availability of data. Model objectives and conditions of application should be used to select a certain number of suitable, quantitative metrics for evaluating relevant (according to the specific study) model features. It is possible to find in the literature a huge number of evaluation metrics, focusing on different aspects of model accuracy, structure, and behaviour.

Model evaluation was traditionally intended as a measure of the agreement between measured and simulated data. Such an agreement can be visually analyzed using charts where measured and simulated data series are plotted against time (in case of dynamic models) or using simulated vs. measured scatter plots to highlights deviations from the 1:1 line. Although this kind of evaluation can be useful to analyze possible model inaccuracies in light of the model behavior during the simulation, it is a good practice to quantify different typologies of disagreement between observations and simulated data using specific metrics (indices of agreement) (e.g., Loague and Green 1991; Martorana and Bellocchi 1999). Among the indices proposed, the root mean square error (RMSE; Fox 1981), the relative RMSE (also named general standard deviation, GSD; Jørgensen et al. 1986), the modelling efficiency (EF, if negative, the average of observation is a better predictor than the model; Nash and Sutcliffe 1970), the correlation coefficient of the estimates versus measurements ($R$), the probability of equal means by the paired Student $t$-test ($P(t)$), and the presence of patterns in residuals (Donatelli et al. 2004a) are some of the most used for estimating model accuracy.

Although the literature about agreement metrics is flourished in the last decades, the modellers community has recently started to look at model evaluation as something going beyond the simple quantification of the agreement between measured and simulated data. The concept of model performance has been extended to account also for aspects of model structure and behaviour that, although partly or totally distinct from model accuracy, can be useful to discriminate among different approaches. Akaike (1974) proposed an index – considered as a numerical formulation of the Occam's razor – to quantify in a synthetic metric both the accuracy (via the mean square error) and the complexity of the model, with the latter estimated through the number of inputs needed by the model. Such a criterion should be considered important, since in case the increase in effort for feeding a more complex model is not counterbalanced by an actual increase in the accuracy of estimations, there is no reason to reject the simplest one in a given context. Another important aspect of model evaluation is robustness. This is one of the model features users should be more interested in, especially in case of large area applications, when users have to trust the model in conditions far from those in which the model itself was calibrated and tested (Confalonieri et al. 2010b). An indicator for quantifying model robustness was recently proposed, based on the ratio between the variability of EF and the variability of explored conditions (Confalonieri et al. 2010b). Another criterion was proposed for model balance Confalonieri (2010), intended as the model tendency to avoid concentrating most of the parameters relevance in few parameters.

The availability of a variety of evaluation metrics and the different purposes and conditions of application of modelling studies led to the development of multi-metric evaluation systems (Bellocchi et al. 2002; Confalonieri et al. 2009a). This, in turns, led to the use of advanced fuzzy-based procedures to overcome the mathematical and conceptual limits of classical aggregation procedures (i.e., summation, multiplication, or a combination of both) in managing the intrinsic subjectivity in the definition of thresholds and weights (Keeney and Raiffa 1993). A direct evidence of the usefulness of multi-metric evaluation systems, based on the aggregation of metrics quantifying different aspects of models performance, is demonstrated by the absence of correlation among the different evaluation metrics and by the different rankings obtained by a set of models when a different evaluation criterion is used. As an example, Confalonieri et al. (2010b) underlined how the most robust method (i.e., Mahmood and Hubbard 2002) for the estimation of global solar radiation among the six alternative approaches compared by Abraha and Savage (2008), was – among the same models – the less accurate.

## 11 Conclusions

Bio-physical process based models allow estimating crop yield in response to weather, agricultural management, and diseases. They also allow estimating the externalities of the system, which may have an environmental impact. Such estimates can also be expressed as probability distributions, allowing the estimate of risk. These models can be used to extrapolate to new environments and production systems, provided that a system analysis confirms the conceptual validity of the model construct, which is always a simplification of the real system.

These models are the only option available to estimate system performance in cases where running experiments on the real system can be dangerous for the system itself, too expensive, or requiring long time prior to producing results representative of the system under study. In these conditions modeling is the unique possible choice for estimating crop-soil interactions under the driving forces of climate and agricultural management. However, bio-physical models can be demanding in terms of the information required to calibrate and run them, and require a full understanding of the system being modeled and of model assumption by users.

Using models with a strong empirical basis, built using a reductionist approach, can be accepted in very limited cases, in any case restricting their use to the systems and conditions used to develop them. Evaluating the performance of innovative systems built combining response curves to single factors does not represent the behavior of the system which is governed by complex non-linear relationship which cannot be represented by a single relationship to be optimized analytically.

Although biophysical models are already an effective tool for system analysis, a large effort still needs to be acted on to allow exploiting their full potential. This aiming at addressing the multiple and moving targets required by integrated analyses, in which bio-physical modeling plays the role of data provider to the following steps of the modeling chain. The work to be done can be identified as further improvement of models in terms of integration of modeling approaches, but primarily to building database of reference data to be used as benchmark for model evaluation, and database allowing the use of models beyond case studies.

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# Part II

# Chapter 5
# Incorporating Yield Information from a Biogeochemical Model into an Agricultural Production Model to Infer Adoption of a New Bioenergy Crop

**P. Mérel, F. Yi, S. Bucaram, J. Lee, R. Howitt, and J. Six**

Positive Mathematical Programming (PMP) models of agricultural supply have been popularised by Howitt (1995b) and used extensively in policy analysis, to predict the response of agricultural systems facing resource, technology and policy constraints to exogenous shocks. The models are typically calibrated against observed regional—e.g., in the U.S., county-level or state-level—cropping patterns and input allocations, under the maintained assumption of profit-maximising behavior. To calibrate regional models of agricultural supply, model parameters are chosen so that the first-order conditions to the economic optimisation program are satisfied at the observed base-year allocation. This is made possible by specifying a non-linear objective function, to avoid overspecialisation. In the canonical example of Howitt (1995b), a quadratic term is added to the net revenue of profitable activities so that yields are linearly decreasing in acreage, but other specification rules may be used (Heckelei and Britz 2005; Heckelei and Wolff 2003). The profit-maximising assumption allows the analyst to model the outcome of the production decisions of atomised farmers, facing the same input and output prices, as the result of the optimisation of aggregate farm returns subject to regional resource and/or technical constraints. As such, the calibrated production functions obtained from PMP reflect technology and resource limitations at the regional level.

More recently, the literature on mathematical programming has developed methodologies to also force regional programming models to replicate an exogenous supply response pattern, through the use of prior information—typically in the form of econometric estimates—on supply elasticities (Heckelei 2002; Heckelei

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and Wolff 2003; Jansson and Heckelei 2008; Mérel and Bucaram 2010). The idea is to avoid constructing models that display unreasonable supply responses to price shocks. More fundamentally, the response of programming models to policy is dominated by supply elasticities, and any reasonable calibration algorithm should control, one way or another, for their magnitude. This additional demand on the part of PMP models is theoretically possible to satisfy, due to the typical underdeterminacy of such models. Said differently, PMP objective functions typically have enough calibrating parameters to allow replication of both the observed allocation and a set of supply elasticities at the base-year conditions. As such, the use of prior information on supply elasticities can also be viewed as a way to mitigate the underdeterminacy problem.

Crop allocation models calibrated on economic information regarding observed cropping patterns and estimated supply responses, when linked to agronomic models, also constitute a valuable tool to assess the effects of environmental policies. For instance, the impact of a nitrogen tax on nitrogen losses through leaching and gaseous emissions at the regional level can be inferred by linking a calibrated PMP model of crop allocation that integrates the economic incentives facing farmers to a calibrated biogeochemical model that predicts nitrogen losses from various agricultural activities under given regional conditions. In the same vein, a bio-economic model that incorporates information on soil processes can be used to infer the carbon sequestration potential of various management practices, for instance reduced tillage (Howitt et al. 2009).

In addition to providing information on the quantity of environmental externalities arising from agricultural activities, a biogeochemical model can provide critical "engineering" information on production processes that may be used to complement the "economic" information on technology contained in the observed allocation. For instance, if prior information on supply elasticities is not available to calibrate non-linear terms in the objective function, yield variation at the regional level—a piece of information typically available from highly disaggregated biogeochemical models of plant growth—may be used to construct such terms and thus ensure the smoothness of the model's response (Howitt 1995b). Such information is particularly needed for newly introduced activities, for which there is no historical acreage allocation and for which no reasonable supply elasticity estimate is available. In this chapter, we focus our attention on the use of such information to derive regional supply curves for a newly introduced activity, namely switchgrass for the production of bioenergy. The analysis provides information about the extent and location of potential switchgrass production in California and has direct policy implications regarding the economic viability of switchgrass-based biofuel production and the optimal location of processing plants.

This chapter is organised as follows. First, we review recent developments in the field of PMP calibration, with a focus on the incorporation of prior information on own-price supply elasticities in calibrated models. This provides the methodological basis for calibrating a regionalised model of California agriculture with land and water constraints. Second, we explain how information obtained through simulation of crop cycles within the biogeochemical model DAYCENT

(Del Grosso et al. 2008) can be used to introduce a new crop into the calibrated PMP model. Third, we present preliminary results from an application of this approach to switchgrass adoption in California. In the conclusion, we briefly discuss limitations and extensions.

# 1 First- and Second-Order PMP Calibration of a Regionalised California Agriculture Model

## *1.1 Calibration Against Observed Acreage*

Calibration of a PMP model to observed input allocation and output levels (hereafter called first-order calibration) is standard and described in Howitt (1995a,b). The allocation data typically consists of a single observation on market conditions (prices of outputs and inputs, resource availabilities) and observed economic behavior (input allocations and output levels). In applications, the reference year allocation may be obtained as the average of a small number of observations. This is particularly useful when the data comes from different sources, as with models of explicit input allocation such as constant-elasticity-of-substitution (CES) models, because input allocation data typically comes from accounting surveys that are conducted at different dates and/or frequencies than available data on prices, acreage and yields.[1]

Given the base year allocation data, first-order calibration is achieved by (i) assuming profit maximisation by farmers, (ii) specifying a nonlinear functional form for the regional profit function and (iii) recovering values for the shadow prices of constrained resources. The resulting calibrated model is therefore only as good as these three elements are in describing farmers' choices.

Out of the three types of assumptions above, assumption (iii) has certainly been the most controversial. One of the main contributions of Howitt (1995b) was to show that values for the shadow prices of constrained resources (or policy constraints) can be recovered in a first-stage linear programming model, subject to calibration constraints. Calibration constraints force the optimised cropping pattern at the base year conditions to coincide with the reference allocation, but are decoupled from the binding resource constraints by adding an arbitrarily small disturbance term. Shadow values on resource constraints obtained from this first step can then be used in a second step to construct the nonlinear profit function, so that its optimisation at the base year conditions exactly results in the observed input and output allocation (up to the small disturbance), and optimised

[1] For instance, in our application to the California SWAP model, data on acreage allocation, output prices and yields comes from the California Department of Water Resources, while input use data comes principally from the University of California Cost and Return Studies.

values of the shadow prices of constrained resources coincide with the values obtained in the first stage.

Recent literature has highlighted that using the first stage of PMP to recover values for the shadow prices of constrained resources is arbitrary, and thus questionable (Buysse et al. 2007; Heckelei and Wolff 2003). As noted by Buysse et al. (2007), the traditional two-stage PMP approach mechanically assigns the highest possible values to the dual variables of the resource constraints that still allow for calibration. Several authors suggest using exogenous information on the prices of constrained resources (such as land rent) as an alternative to the first stage (Gohin and Chantreuil 1999; Heckelei and Britz 2005; Kanellopoulos et al. 2010). Heckelei and Wolff (2003) and Heckelei and Britz (2005) argue, in the context of estimation models that rely on more than one observation, that the use of shadow values obtained from a linear programming model is fundamentally inconsistent with the final, nonlinear programming model used for policy analysis, and propose to estimate the shadow values as part of a generalised maximum entropy (GME) optimisation program. It is not the purpose of this chapter to discuss these issues, particularly since our application relies on a calibration model with one single observation. We refer the reader to the above referenced literature for a more detailed overview.

## *1.2 Calibration Against Own-Price Elasticities*

The use of prior information on supply elasticities to calibrate PMP models of agricultural supply has been advocated repeatedly in the recent literature (Heckelei and Britz 2005; Mérel and Bucaram 2010). The reason is twofold: first, PMP models are typically underdetermined, that is, the information on the observed cropping pattern and input allocation is not sufficient to recover the entire set of model parameters. The literature has dealt with this underdeterminacy problem by either imposing a priori restrictions—in quadratic models for instance, setting off-diagonal elements to zero is a popular modelling choice—or, more recently, by using generalised maximum entropy algorithms to recover the entire set of model parameters (Paris and Howitt 1998). The use of prior information on crop supply elasticities as a second source of information to recover model parameters has the ability to mitigate the underdeterminacy problem. Second, whether arbitrary restrictions or GME algorithms are used, traditional PMP algorithms are not always geared towards ensuring consistency of the model's implied supply responses with econometric priors regarding the value of supply elasticities. Although any PMP model exactly replicates the observed cropping pattern, different calibration rules imply different—and sometimes unrealistic—supply response patterns (Heckelei and Britz 2005).

An early solution to this problem has been to use "myopic" calibration rules. Such rules ignore the change in the shadow prices of constrained resources (in particular, land) that are induced by the change in crop prices, and therefore allow each activity to be calibrated separately from all others. However, they provide an acceptable calibration rule only when changes in shadow prices are negligible. Mérel and Bucaram (2010) recently provided an *ex ante* test to determine, within quadratic models of crop allocation, whether the use of a myopic calibration rule is defendable in practice. In essence, the base allocation must have a sufficiently large number of positive activities, and no activity can have a desired acreage response that dominates all others.

When the use of "myopic" calibration rules cannot be justified, one must take account of the fact that the implied supply elasticity of each crop depends on all model parameters, and it is no longer possible to calibrate each activity independently. The modeller then needs to solve a system of (typically) nonlinear equations that is not guaranteed to have an acceptable solution, that is, a solution that preserves the concavity properties of the economic optimisation program.

Recent research in the area of *exact* calibration of PMP models has focused on the following questions: (i) How to recover the supply elasticities implied by a given model specification, as a function of the model parameters to be calibrated? (ii) Given a system of nonlinear calibrating equations, under which conditions can the analyst recover an acceptable solution? and (iii) Under which conditions is the solution to the calibrating system unique?

Mérel and Bucaram (2010) have provided a general answer to question (i). Questions (ii) and (iii) cannot be answered generally and instead are model-specific. This is because the general form of the calibration system depends upon the form of the objective function. Two popular PMP models specified in primal form are the CES-quadratic model (Howitt 1995a; Graindorge et al. 2001; Mérel and Bucaram 2010) and the generalised CES model (Heckelei and Wolff 2003; Mérel et al. 2010). Mérel et al. (2010), in addition to deriving the essential conditions for calibration of the generalised CES model, argue that it is preferable to the CES-quadratic model from a theoretical standpoint, and we therefore adopt the generalised CES model in the present chapter.[2]

## 1.3 An Application to the California SWAP Model

Our model of California agriculture is built as an extension to the existing statewide water and agricultural production (SWAP) model developed by R. Howitt (Jenkins et al. 2001). The SWAP model divides California into $G = 27$ regions based mostly on water transferability. These regions are shown in Fig. 5.1 and described in Table 5.1. There are four water sources in California: the Central Valley Project

[2] These authors also show that, subject to myopic calibration being feasible, the generalised CES model is more flexible than the CES-quadratic model, in the sense that it can accommodate larger sets of supply elasticities. This argument is not crucial to our choice to use a generalised CES specification, since the conditions for myopic calibration are not always satisfied in our application.

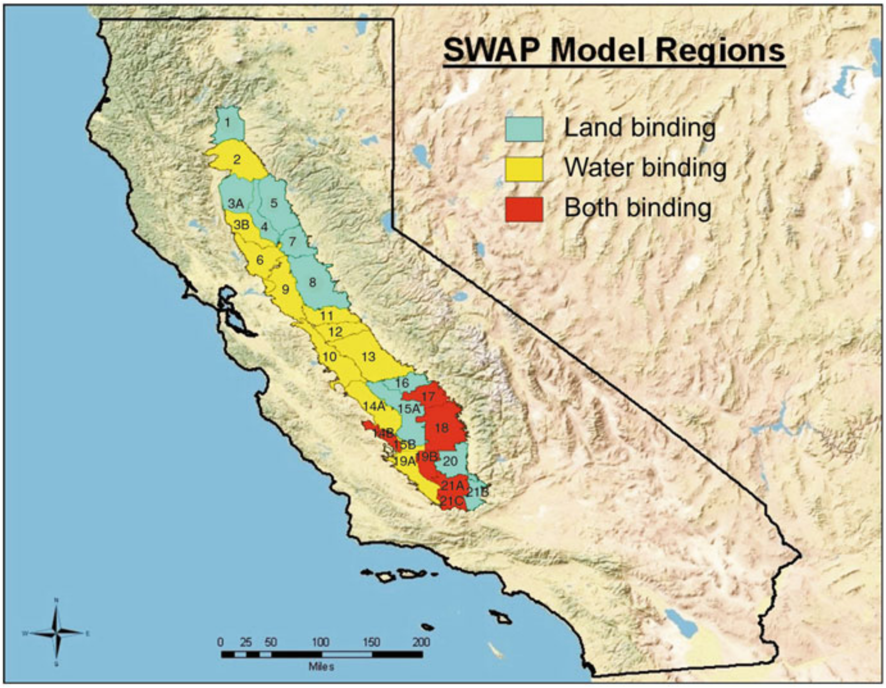


**Fig. 5.1** The SWAP agricultural regions

(CVP), the State Water Project (SWP), local surface water and ground water. Water use from these sources is either based on long-term contracts (e.g., CVP and SWP), or water cannot be transferred between regions because of existing law (e.g., ground water). In addition, local water agencies only provide water to specific regions. As a result, each of the 27 SWAP regions can be considered independent in terms of water allocation. In addition, land cannot be moved across regions. Thus, there are two constrained resources in each SWAP region: land and water. (Both constraints need not be binding, see Fig. 5.1.)

The statewide economic optimisation model is defined as follows:

$$\max_{q_{gi}\geq 0, x_{gij}\geq 0} \sum\nolimits_g \sum\nolimits_i p_{gi}q_{gi} - \left[(c_{gi1}+\lambda_{gi1})x_{gi1} + (c_{g2}+\lambda_{gi2})x_{gi2} + c_{g3}x_{gi3}\right]$$

subject to

$$\begin{cases} \sum_{i=1}^{I} x_{gi1} \leq b_{g1} & \forall g \in [|1,G|] \\ \sum_{i=1}^{I} x_{gi2} \leq b_{g2} & \forall g \in [|1,G|] \\ q_{gi} = \mu_{gi}\left[\sum_{j=1}^{3} \beta_{gij} x_{gij}^{\rho_{gi}}\right]^{\frac{\delta_{gi}}{\rho_{gi}}} & \forall (g,i) \in [|1,G|] \times [|1,I|] \end{cases} \tag{5.1}$$

**Table 5.1** Description of the SWAP regions

| SWAP region | Counties |
|---|---|
| 1 | Tehema, Shasta |
| 2 | Butte, Glenn, Tehema |
| 3A | Colusa, Glenn, Yolo |
| 3B | Colusa, Glenn, Yolo |
| 4 | Butte, Colusa, Glenn, Sutter, Yolo |
| 5 | Butte, Glenn, Sutter, Yuba |
| 6 | Solano, Yolo |
| 7 | El Dorado, Placer, Sacramento, Sutter |
| 8 | Amador, Claveras, Sacramento, San Joaquin, Stanislaus |
| 9 | Alameda, Contra Costa, Sacramento, San Joaquin, Solano, Yolo |
| 10 | Fresno, Merced, San Benito, San Joaquin, Stanislaus |
| 11 | San Joaquin, Stanislaus |
| 12 | Merced, Stanislaus |
| 13 | Madera, Mariposa, Merced |
| 14A | Fresno, Kings |
| 14B | Fresno, Kern, Kings |
| 15A | Fresno, Kings, Tulare |
| 15B | Kings |
| 16 | Fresno |
| 17 | Fresno, Kings, Tulare |
| 18 | Kings, Tulare |
| 19A | Kern |
| 19B | Kern |
| 20 | Kern, Tulare |
| 21A | Kern |
| 21B | Kern |
| 21C | Kern |

where $p_{gi}$ is the price of crop $i$ in region $g$ and $c_{gj}$ is the price of input $j$ ($j = 1, 2, 3$) in region $g$. The choice variables $x_{gij}$ represent the amount of input $j$ used in the production of crop $i$ in region $g$, and $q_{gi}$ the output level, related to the input employments in a generalised CES production function with parameters $\mu_{gi}$, $\beta_{gij}$ and $\delta_{gi}$, satisfying $\mu_{gi} > 0$, $\beta_{gij} > 0$, $\sum_j \beta_{gij} = 1$ and $\delta_{gi} \in (0, 1)$. There are three explicitly modeled inputs in our model. The indices $j = 1$ and $j = 2$ denote land and water, respectively. The third explicit input is labour, assumed to be supplied in a perfectly elastic fashion to the farm sector. For the purpose of this study, all other inputs (such as pesticides, fertiliser, custom operations etc.) are assumed to be employed in fixed proportion with land, and therefore their respective cost is included in the price of land, $c_{gi1}$.[3] The parameters $b_{g1}$ and $b_{g2}$ represent the limited

[3] Since different activities require different proportions of these other inputs, the "cost" of land is therefore crop-specific in our model. Note that this cost represents the cost of these inputs and does not reflect the scarcity of land, which is embedded in the shadow value of the land constraint.

land and water resources in each region. Following common PMP practice, calibration parameters $\lambda_{gi1}$ or $\lambda_{gi2}$ are added to the land and water cost terms to allow for calibration against the reference allocation (otherwise, the model does not have enough parameters to calibrate). For each crop, at most one of these parameters is nonzero, so that the calibration problem is not underdetermined. The choice of whether $\lambda_{gi1} = 0$ or $\lambda_{gi2} = 0$ is driven by ease of calibration against the supply elasticity. The analysis by Mérel et al. (2010) suggests that larger sets of supply elasticities will be replicable if the cost adjustment parameter is added to the input with the largest cost share in the reference allocation (including the shadow price of any limited input). Therefore, we choose to add the cost adjustment to whichever input has the largest cost share, here either land or water.[4]

The calibration phase consists of recovering the set of unknown parameters ($\mu_{gi}$, $\beta_{gi}$, $\delta_{gi}$, $\lambda_{gi1}$, $\lambda_{gi2}$), given the reference allocation and a set of supply elasticities. The parameter $\rho_{gi}$ is a pure substitution parameter and is given by $\rho_{gi} = \frac{\sigma_{gi}-1}{\sigma_{gi}}$, where $\sigma_{gi}$ is the elasticity of substitution among inputs. In the absence of reliable prior information on the value of substitution elasticities, we choose to set $\sigma_{gi} = 0.5$.[5]

We calibrate model (5.1) against the observed cropping pattern and supply elasticities using the latest PMP methodology developed by Mérel et al. (2010).[6] The starting point is a set of exogenous state elasticity priors from SWAP (see Table 5.2).[7] Mérel et al. (2010) have derived the necessary and sufficient conditions for calibration of the generalised CES model. Unfortunately, here these conditions are often violated when the available statewide elasticities are used at the regional level, even once the cost increment parameters are chosen optimally to allow for maximum flexibility. Our approach is therefore to construct region-specific supply elasticities using a generalised maximum entropy algorithm that disaggregates the econometric elasticity estimate by minimising the information cost from deviating, in each region *and* at the state level, from the state elasticity prior, while allowing for exact calibration. State level elasticities are calculated as weighted averages of the regional supply elasticities, using output shares as the weights, and support intervals are wider for regional elasticities than for state level elasticities. This algorithm enables one to recover region-specific elasticities while allowing for calibration of the model against a set of elasticities that departs from the prior minimally—in a maximum entropy sense. The resulting regional variation in supply elasticities, certainly desirable from a modelling perspective, is driven by

[4] Mérel et al. (2010)'s analysis applies for regions with one binding constraint only. In regions where two of the constraints are binding we apply the increment to the land cost systematically ($\lambda_{gi2} = 0$).

[5] Howitt (1995a) uses $\sigma = 0.7$ for a similar breakdown of farm inputs. Hatchett (1997) estimates elasticities of substitution between water and capital for various crops in California. His estimates range around 0.6–0.8.

[6] A standard PMP approach is used to infer the dual values of constrained resources in the reference allocation.

[7] Most of these elasticities come from the study by Russo et al. (2008).

**Table 5.2** Statewide supply elasticities

| Crop | Prior elasticity | Calibrated elasticity | Regional variation |
|---|---|---|---|
| Almond and Pistachio | 0.03 | 0.23 | Yes |
| Alfalfa | 0.24 | 0.37 | Yes |
| Corn | 0.21 | 0.50 | Yes |
| Cotton | 0.36 | 0.44 | Yes |
| Dried Bean | 0.13 | 0.51 | Yes |
| Fresh Tomato | 0.16 | 0.16 | No |
| Wheat | 0.36 | 0.43 | Yes |
| Onion and Garlic | 0.11 | 0.80 | Yes |
| Other deciduous fruits and nuts | 0.03 | 0.53 | Yes |
| Other field crops | 0.63 | 0.77 | Yes |
| Other truck crops | 0.11 | 0.12 | Yes |
| Potato | 0.11 | 0.12 | Yes |
| Processing tomato | 0.15 | 0.21 | Yes |
| Rice | 0.96 | 0.96 | No |
| Safflower | 0.34 | 0.34 | Yes |
| Sugar Beet | 0.11 | 0.50 | Yes |
| Citrus | 0.03 | 0.10 | Yes |
| Grape Vine | 0.05 | 0.60 | Yes |

Note: Corn includes grain and silage. Other deciduous fruits and nuts includeapples, apricots, cherries, plums, walnuts, etc. Other field crops includegrain sorghum, sudan grass, sunflower, etc. Other truck crops includeartichokes, asparagus, green beans, carrots, celery, lettuce, flowers, berries,peppers, cabbage, etc. Grape Vine includes wine grapes, table grapes and raisins

observed input and output allocation patterns, the choice of functional form for the crop-specific profit functions, and the necessity to calibrate as closely as technically feasible to the initial prior.[8] In our application, a majority of 16 crops (out of 18) display regional variation in their supply elasticities. The resulting statewide elasticities are reported in Table 5.2. While a handful of elasticities seem to differ widely from the initial prior, our choice of regional and state elasticities is optimal in the sense that the state elasticities are as close as possible to the prior values while allowing the model to calibrate against regional elasticities.[9]

[8] Note that regional variation in supply elasticities will only obtain if the calibration conditions are violated in at least one region. In that case, elasticities may differ from the prior even in regions for which the calibration conditions would hold, in order to make the state level elasticity as close as possible to the prior.

[9] Since a calibration criterion was not available for models with two binding constraints at the time when this chapter was written, the state averages reported in Table 5.2 only take into account regions where either land or water is binding. Out of the 27 regions, 21 have only one binding constraint. We note that crops for which calibrated statewide elasticities differ widely from the prior often correspond to "grouped crops": Almond and Pistachio, Corn, Onion and Garlic, Other Deciduous Fruits and Nuts, Grape Vine. Our inability to calibrate these crops close to the elasticity prior is likely due to the aggregation process.

## 2 Calibrating Regional Production Functions for a New Crop

To analyse how the introduction of a new activity affects the input allocation decisions in each region, we need to calibrate a production function for switchgrass. Since technology parameters for switchgrass cannot be recovered from observed economic behavior, we rely on information obtained from the calibrated biogeochemical crop simulation model DAYCENT (Lee et al. 2010) to identify the essential relationship between input intensity and output. We here present a simple application where the production function for switchgrass essentially consists of a relationship between acreage and output, namely

$$q_{gs} = \mu_{gs} x_{gs}^{\delta_{gs}}$$

where $x_{gs}$ is acreage of switchgrass in region $g$, $q_{gs}$ is output, and $\mu_{gs}$ and $\delta_{gs}$ are unknown technology parameters satisfying $\mu_{gs} > 0$ and $\delta_{gs} \in (0, 1)$. This production function is simply the fixed-proportion variant of the generalised CES specification used for other crops in the model.

Simulation data from DAYCENT is used to construct, for each SWAP region, estimates of switchgrass expected yields, given specific water and fertiliser application rates. The water and fertiliser application rates that are used for each region correspond to "optimal" rates from a purely agronomic perspective. Predicted yield, conditional on water and fertiliser rates, is obtained from DAYCENT at a given geographical "point". The DAYCENT yield prediction depends on local conditions at that point: temperature, soil characteristics, weather, etc. GIS information on these local conditions can be exploited to obtain region-specific yield estimates. The California Central Valley was divided into $12 \times 12$ km$^2$ (cells) that were treated as being homogenous in terms of local conditions. The DAYCENT model was run to obtain yield estimates for each cell. In a given agricultural region $g$ covering multiple cells, we used the highest and second-highest yields to calibrate the production function for switchgrass. Let us denote by $y_{gs}^{\max}$ the highest yield in region $g$, covering an area $a_{gs}^{max}$, and by $y_{gs}^{second}$ the second-highest yield. These yields are reported for each agricultural region in Table 5.3.

The highest yield $y_{gs}^{max}$ represents the average yield over the agricultural area $a_{gs}^{max}$, and therefore

$$y_{gs}^{max} = \mu_{gs}\left(a_{gs}^{max}\right)^{\delta_{gs}-1}. \tag{5.2}$$

We interpreted the second-highest yield as the "marginal yield" at the acreage $a_{gs}^{max}$, that is,

$$y_{gs}^{second} = \mu_{gs}\delta_{gs}\left(a_{gs}^{max}\right)^{\delta_{gs}-1}. \tag{5.3}$$

**Table 5.3** Regional yields and input application rates

| Region | | Yield $y^{max}$ | Yield $y^{second}$ | Water | Labour | Variable cost |
|---|---|---|---|---|---|---|
| Name | Acres | (US ton/ac) | (US ton/ac) | (ac-ft/ac) | (h/ac) | ($/ac) |
| 1 | 6,300 | 13.30 | 13.24 | 2.72 | 3 | 524.61 |
| 2 | 141,600 | 13.61 | 13.54 | 2.74 | 3 | 593.91 |
| 3A | 255,456 | 13.53 | 13.39 | 3.00 | 3 | 597.33 |
| 3B | 88,444 | 13.53 | 13.39 | 3.00 | 3 | 597.33 |
| 4 | 241,140 | 13.39 | 13.38 | 2.96 | 3 | 550.84 |
| 5 | 338,120 | 13.58 | 13.47 | 2.99 | 3 | 531.28 |
| 6 | 211,790 | 13.38 | 13.34 | 3.52 | 3 | 573.26 |
| 7 | 86,530 | 14.09 | 13.63 | 2.64 | 3 | 545.28 |
| 8 | 274,360 | 14.09 | 13.68 | 2.61 | 3 | 575.60 |
| 9 | 356,250 | 13.68 | 13.38 | 4.06 | 3 | 569.58 |
| 10 | 394,270 | 12.83 | 12.75 | 4.67 | 3 | 661.62 |
| 11 | 185,370 | 12.22 | 12.19 | 2.60 | 3 | 504.56 |
| 12 | 231,300 | 12.79 | 12.70 | 2.66 | 3 | 520.29 |
| 13 | 519,400 | 13.77 | 13.73 | 3.01 | 3 | 555.65 |
| 14A | 457,100 | 13.33 | 13.32 | 4.54 | 3 | 800.72 |
| 14B | 37,900 | 14.07 | 14.00 | 3.28 | 3 | 717.06 |
| 15A | 619,300 | 14.22 | 13.88 | 4.10 | 3 | 694.04 |
| 15B | 19,000 | 14.22 | 14.00 | 3.23 | 3 | 646.94 |
| 16 | 146,000 | 13.93 | 13.61 | 2.96 | 3 | 534.96 |
| 17 | 256,400 | 14.39 | 14.21 | 2.85 | 3 | 571.35 |
| 18 | 710,300 | 14.39 | 14.38 | 3.04 | 3 | 558.11 |
| 19A | 83,500 | 13.70 | 13.56 | 4.65 | 3 | 708.86 |
| 19B | 165,900 | 13.95 | 13.93 | 4.03 | 3 | 645.10 |
| 20 | 209,500 | 14.23 | 14.17 | 3.16 | 3 | 631.94 |
| 21A | 192,800 | 13.95 | 13.93 | 2.96 | 3 | 651.06 |
| 21B | 100,600 | 13.51 | 13.12 | 2.87 | 3 | 648.40 |
| 21C | 65,700 | 13.51 | 13.14 | 2.92 | 3 | 649.98 |

Note: The fertiliser application rate was set at 200 lbs N/ac in all regions

Equations (5.2) and (5.3) constitute a system of two equations from which the technology parameters $\mu_{gs}$ and $\delta_{gs}$ can be recovered. Other calibration rules can be implemented, that rely on a different information set.[10] The advantage of interpreting the second-best yield as a marginal yield is that it guarantees that the returns to scale parameter $\delta_{gs}$ will lie between zero and one, no matter what the empirical values of $y_{gs}^{max}$ and $y_{gs}^{second}$ turn out to be. A graphical interpretation of the calibration procedure is given in Fig. 5.2.

[10] As an alternative to this approach, we calibrated the production function using the maximum yield $y_{gs}^{max}$ and the average yield $y_{gs}^{av}$ reported by DAYCENT. The maximum yield was interpreted as the average yield over the acreage $a_{gs}^{max}$, while $y_{gs}^{av}$ was interpreted as the average yield over the entire agricultural area in the region, $b_{g1}$. Adoption patterns were extremely close to those reported here.

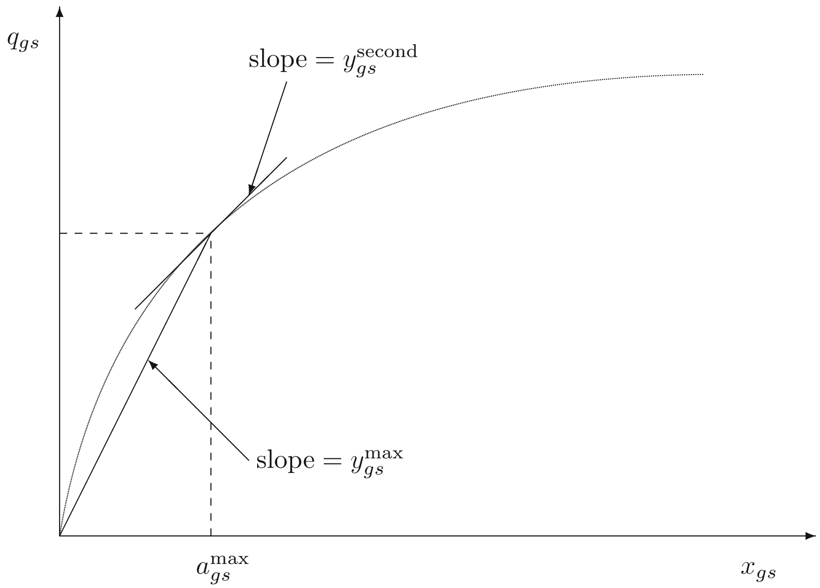


**Fig. 5.2** Calibration of the switchgrass production function

The regionalised economic optimisation model once switchgrass is introduced has the form

$$\max_{\substack{q_{gi}\geq 0, x_{gij}\geq 0\\ q_{gs}\geq 0, x_{gs}\geq 0}} \sum_g \sum_i p_{gi}q_{gi} - \left[(c_{gi1}+\lambda_{gi1})x_{gi1} + (c_{g2}+\lambda_{gi2})x_{gi2} + c_{g3}x_{gi3}\right]$$
$$+ p_{gs}q_{gs} - C_{gs}x_{gs}$$

subject to

$$\begin{cases} \sum_{i=1}^{I} x_{gi1} + x_{gs} \leq b_{g1} & \forall g \in [|1, G|] \\ \sum_{i=1}^{I} x_{gi2} + x_{gs}w_g \leq b_{g2} & \forall g \in [|1, G|] \\ q_{gi} = \mu_{gi}\left[\sum_{j=1}^{3} \beta_{gij}x_{gij}^{\rho_{gi}}\right]^{\frac{\delta_{gi}}{\rho_{gi}}} & \forall (g,i) \in [|1, G|] \times [|1, I|] \\ q_{gs} = \mu_{gs}x_{gs}^{\delta_{gs}} & \forall g \in [|1, G|] \end{cases} \tag{5.4}$$

where $q_{gs}$ is the regional quantity of switchgrass produced and $x_{gs}$ the corresponding acreage. In the water availability constraint of program (5.4), the parameter $w_g$ denotes the regional water application rate for switchgrass. The variable $C_{gs}$ represents an estimate of the variable per-acre cost, based on the water and fertiliser application rates used to obtain the regional yield estimates, combined with the local prices of water $c_{g2}$ and the price of fertiliser, as well as an exogenous estimate of other operating costs, including labour. Input intensities and resulting variable costs are reported in Table 5.3. Although not the focus of this particular study, sensitivity analysis on $C_{gs}$ could be conducted to test the robustness of our results to this exogenous information.

The calibrated model can then be used for policy analysis. The question we address here is the extent and location of switchgrass production at various hypothetical switchgrass prices, that is, the derivation of the regional and statewide supply curves for switchgrass. Such information is likely to be of interest to policy makers and entrants in the biofuel industry alike. The use of a regionalised model is of critical importance since biofuel feedstock is usually expensive to transport. It is, therefore, pertinent to know where potential biofuel production would be located. This particular question clearly illustrates the need for a pluridisciplinary approach that can combine technical information regarding regional yield possibilities and input intensities for the new crop (information that is typically not available to the econometrician) and economic information regarding the opportunity cost of growing switchgrass in each agricultural region, taking full account of the limited availability of some inputs, the existing technology set and observed market conditions.

## 3 A Pattern of Switchgrass Adoption

To illustrate the possibilities offered by our approach, we derived regional supply patterns and the statewide supply curve for switchgrass in California using the fully calibrated SWAP model (5.4). To this end, we simply solved program (5.4) iteratively for $p_{gs} \in [0, \$70/\text{ton}]$. We conducted this experiment under two market scenarios: (i) exogenous output prices and (ii) endogenous output prices. In scenario (i), model (5.4) was run as is. Scenario (ii) followed the assumptions of SWAP. State-level demand functions were added to the model, for all crops other than switchgrass. The initial state-level prices were calculated as $P_i = \frac{\sum_g p_{gi} q_{gi}}{\sum_g q_{gi}}$, and linear demand functions were fitted through the initial point using exogenous estimates of residual demand elasticities.[11] The difference between the initial regional price $p_{gi}$ and the state-level price $P_i$ can be interpreted as a regional marketing cost (which is negative for some regions by construction). This regional price difference reflects differences in transportation costs among regions and was assumed to be constant per unit of output. The market equilibrium was found by maximising total economic surplus, including consumer surplus and taking account of regional marketing costs.

[11] Most of these elasticities came from Russo et al. (2008).

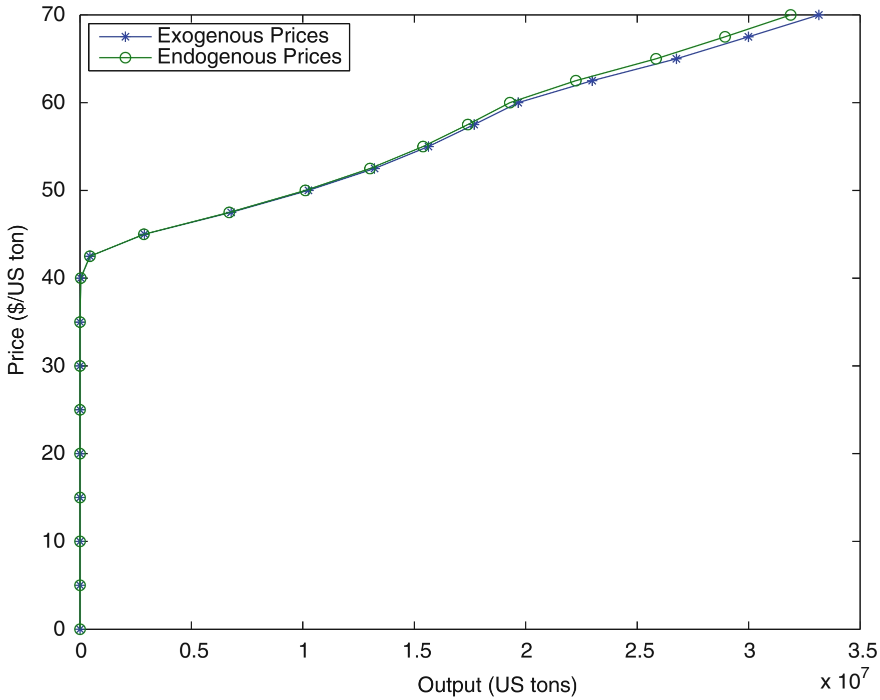


**Fig. 5.3** State supply curves for switchgrass

Figure 5.3 depicts the state-level supply curve for switchgrass under the fixed and endogenous prices scenarios. The two curves are extremely close, reflecting the fact that California faces a highly elastic demand for the included crops.[12] The supply curve corresponding to scenario (i) (fixed crop prices) lies to the right of the supply curve for scenario (ii) (endogenous crop prices). This is expected, since, as switchgrass enters the cropping pattern, fewer resources are allocated to other crops. When crop prices are endogenous, the prices of other crops therefore increase as switchgrass expands at their expense, which tends to mitigate their decline compared to the situation where crop prices do not change.

Since our model is regionalised, we can also derive regional supply curves for switchgrass. This type of information is particularly relevant when deciding where to locate processing plants, in order to minimise transportation costs. Figure 5.4 depicts switchgrass output for each agricultural region at four different price levels. Figure 5.5 depicts the corresponding acreage allocated to switchgrass as a percentage of total regional acreage. These figures are derived under scenario (ii) (endogenous crop prices).

[12] The statewide supply curve is "bumpy" because it is the horizontal sum of the regional supply curves, which all have an inflection point.

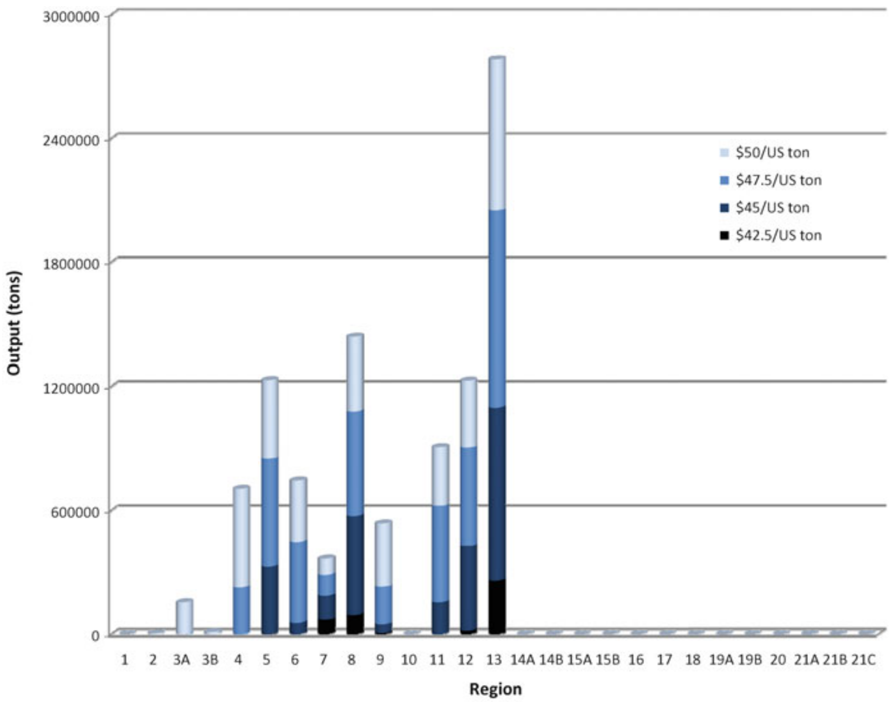


**Fig. 5.4** Regional switchgrass output at various prices

Figures 5.4 and 5.5 show that adoption of switchgrass is far from being uniform across regions, justifying *ex post* the use of a regionalised agricultural model. Figure 5.4 suggests that processing plants should primarily be located in or near regions 12 and 13, corresponding to the counties of Madera, Mariposa, Merced and Stanislaus, because these regions appear to be early and massive adopters of switchgrass. Taken together, these two contiguous regions cover a significant acreage, and a large share of their agricultural land is predicted to be allocated to switchgrass at prices above $45/ton.[13] In contrast, some regions appear as late and/or insignificant adopters, in particular those located in the Southern San Joaquin Valley (all regions with indices 14 and larger). This is not surprising, as switchgrass is modeled as a water-intensive crop, and water is relatively more expensive in this part of the Central Valley. Note that our finding that region 13 is the most significant region

[13] The output and energy levels inferred from our model for prices at the upper end of the range seem to be consistent with the feedstock requirements of cellulosic ethanol plants. Perrin and Williams (2008) report that 80 gal of ethanol can be extracted per ton of switchgrass. Grooms (2009) reports that a US company has begun constructing a commercial-scale cellulosic ethanol facility in Emmetsburg, IA. The capacity of this facility is 25 million gal per year. At a price of $45/ton, region 13 is predicted to supply a little more than one million ton, an equivalent of about 80 million gal of ethanol, and could thus supply several such facilities.

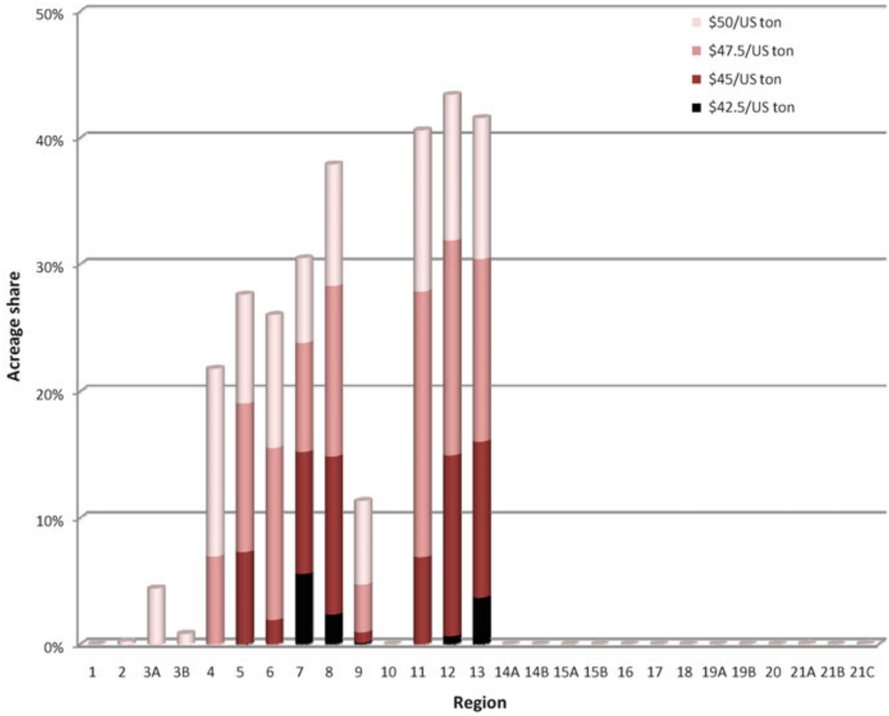


**Fig. 5.5** Regional switchgrass adoption at various prices

in terms of switchgrass output is not a mere consequence of its relatively large size. Indeed, regions 15A and 18 are both larger than region 13, yet they do not adopt switchgrass at any of the price levels considered here (less than $50/ton).

The calibrated bio-economic model can also be used to predict the contraction of crops that are competing with switchgrass for limited resources. Table 5.4 shows the percentage reduction in acreage for existing crops at the state level, for various switchgrass prices. At a price of $45/ton, all competing crops experience acreage contractions, though crops that are considered "specialty crops" in California seem to experience relatively smaller contractions. The crops that are the least affected by the introduction of switchgrass at this price level are Potato, Citrus, Cotton, Fresh Tomato, and Onion and Garlic. The most affected crops are Dried Bean and Corn, and, perhaps surprisingly, Other Deciduous Fruits and Nuts and Grape Vine.

Table 5.5 shows the reduction in the acreages of competing crops for the early and large switchgrass adopter, namely region 13. In this region, at the price of $45/ton, where switchgrass is predicted to take over about 15% of acreage, the acreage contraction exceeds 20% for Dried Bean, Onion and Garlic, Sugar Beet, Corn, Other Deciduous Fruits and Nuts and Wheat. Except for Other Deciduous Fruits and Nuts, all these crops either represent a relatively small share of initial acres or are low value. In contrast, crops that are high value (Fresh Tomato, Rice,

**Table 5.4** Statewide acreage reduction for existing crops

| | | % change in acreage | |
|---|---|---|---|
| | | switchgrass price | |
| Crops | Initial acreage (%) | $p_s = 45$ | $p_s = 50$ |
| Almond and Pistachio | 12.57 | −3.11 | −12.53 |
| Alfalfa | 10.94 | −2.64 | −10.89 |
| Corn | 10.46 | −7.20 | −19.78 |
| Cotton | 10.38 | −0.60 | −2.13 |
| Dried Bean | 0.98 | −7.36 | −29.37 |
| Fresh Tomato | 0.60 | −0.71 | −3.17 |
| Wheat | 5.70 | −3.59 | −16.21 |
| Onion and Garlic | 0.71 | −0.94 | −3.11 |
| Other Deciduous Fruits and Nuts | 9.40 | −5.57 | −18.25 |
| Other Field Crops | 6.79 | −3.64 | −14.53 |
| Other Truck Crops | 3.28 | −1.70 | −7.09 |
| Potato | 0.40 | −0.23 | −1.25 |
| Processing Tomato | 4.73 | −1.86 | −9.96 |
| Rice | 8.83 | −1.92 | −10.16 |
| Safflower | 0.77 | −2.30 | −15.65 |
| Sugar Beet | 0.33 | −4.28 | −8.64 |
| Citrus | 3.88 | −0.52 | −1.73 |
| Grape Vine | 9.25 | −5.05 | −13.96 |

**Table 5.5** Acreage reduction for existing crops in region 13

| | | % change in acreage | |
|---|---|---|---|
| | | switchgrass price | |
| Crops | Initial acreage (%) | $p_s = 45$ | $p_s = 50$ |
| Almond and Pistachio | 26.22 | −8.15 | −26.71 |
| Alfalfa | 13.52 | −16.15 | −44.68 |
| Corn | 12.24 | −28.98 | −62.16 |
| Cotton | 6.08 | −10.54 | −33.52 |
| Dried Bean | 0.12 | −45.24 | −73.94 |
| Fresh Tomato | 1.27 | −2.03 | −7.82 |
| Wheat | 2.84 | −23.40 | −58.24 |
| Onion and Garlic | 0.08 | −42.43 | −60.98 |
| Other deciduous fruits and nuts | 4.74 | −27.44 | −56.68 |
| Other field crops | 8.36 | −15.96 | −46.21 |
| Other truck crops | 2.79 | −9.88 | −31.49 |
| Processing tomato | 2.16 | −8.01 | −26.96 |
| Rice | 0.65 | −4.51 | −17.11 |
| Safflower | 0.06 | −17.24 | −48.14 |
| Sugar Beet | 0.48 | −34.91 | −67.15 |
| Citrus | 1.08 | −10.79 | −33.19 |
| Grape Vine | 17.31 | −19.42 | −47.12 |

Processing Tomato, Almond and Pistachio, Other Truck Crops) experience the smallest contractions. At the higher price of $50/ton, almost all crops in this region (except, maybe, Fresh Tomato) experience a significant reduction in acreage.

## 4 Conclusion

This chapter has demonstrated the usefulness of combining information obtained from observed economic behavior (regionalised input and output allocation, econometrically estimated crop supply elasticities) with information simulated using a biogeochemical model (regionalised yield estimates) to infer the pattern of adoption of a new crop in a diverse agricultural region.

The innovative features of our approach are the use of prior information on state elasticities to derive region-specific supply elasticities against which the model can be exactly calibrated, as well as the recovery of region-specific technology parameters for the new crop using information on average yields and yield variability obtained from a biogeochemical model of plant growth.

Once calibrated, our model was used to infer the pattern of adoption of a new energy crop, switchgrass in California. The use of regionalised economic information combined with regionalised yield estimates allowed for the derivation of a spatially explicit supply pattern. Our results suggest that adoption rates differ widely among California SWAP regions, meaning that the location of processing plants may be an important issue. They also suggest that switchgrass adoption, though plausible at the price levels considered here, is not likely to displace specialty crops by much statewide.

Although our approach may represent a significant step forward in terms of the sophistication of the calibration methodology used, it is not exempt from limitations. First of all, the set of regional supply elasticities recovered using the exact calibration conditions derived in Mérel et al. (2010)—conditional on functional form—yield state elasticities that are, at least for some crops, far from the initial prior. This finding may reflect the fact that we are imposing a lot of structure on the functional relationship between inputs, output and profit. Adopting a more flexible relationship would certainly help mitigate this problem.

Second, even though our model is regionalised, the level of disaggregation (27 regions) is not commensurate with the possibilities offered by DAYCENT in terms of predicted yields. This aspect can be overcome by obtaining more disaggregated economic data, but the cost of doing so is likely high.

Finally, one can regret that the technology specified for the new crop in our application is less flexible than that of existing crops, in the sense that it does not allow for substitution between factors. Indeed, agronomic process models such as DAYCENT are specified and calibrated to accurately reflect the effects of changes in levels of inputs such as fertiliser or water, which generally can be considered as the intensive marginal adjustment. Multi-product economic models with fixed proportions production functions can only represent crop switching at the extensive margin. To avoid losing information from the underlying biogeochemical process models, bio-economic models should certainly take the form of interdependent multi-input production functions, which are able to reflect rational economic adjustment at both the extensive and intensive margins, as well as externalities from specific inputs such as nitrogen or water. This, in our opinion, represents an essential step in fully linking the biogeochemical and economic models.

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# Chapter 6
# Agri-Environmental Nitrogen Indicators for EU27

**A. Leip, F. Weiss, and W. Britz**

## 1 Introduction

Nitrogen is a key element to ensure modern agriculture's output, sustaining global food, feed, fibre and now bio-energy production. But it also accounts also for, or at least contributes to, key environmental problems that challenge the well functioning of today's societies (Sutton et al. 2011). One molecule of nitrogen can contribute to one or many environmental problems, including eutrophication, groundwater pollution via leaching and run-off of nitrates and organic nitrogen, climate change via $N_2O$ emissions, acidification via ammonia emissions and may affect human health via ozone formation or biodiversity via nitrogen deposition on natural areas. This multiple impact of nitrogen is often referred to as the "nitrogen cascade" (Galloway et al. 2003).

Accordingly, agri-environmental indicator frameworks typically feature several indicators related to nitrogen such as ammonia emissions, use of nitrogen fertilisers, gross N surplus, nitrates in water or GHG emissions (EEA 2005). Often, however, these indicators are calculated independently from each other based on sometimes contradicting data sources, methodologies or assumptions (see e.g. Grizzetti et al. 2007). This includes also the first overview of the "European Nitrogen Case" that was presented by van Egmond et al. (2002) at the second International Nitrogen Conference held in Potomac (USA). Thus, a system that calculates the detailed nitrogen balance and the related indicators for agriculture in Europe on the basis of consistent data sets and advanced methodologies is highly desirable.

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A closed balance of nitrogen is calculated in the CAPRI (Common Agricultural Policy Regionalized Impact) model, i.e., next to monetary values and product balances, also the nutrient fluxes are in accordance with the law of mass-conservation (Britz et al. 2007). This has been exploited by Leip et al. (2011b) to develop nitrogen budgets for the system boundaries of the soil, land, and the farm. The authors provide for the first time mutually consistent calculations of farm, land and soil N-budgets for all member states of the European Union and quantify the two major indicators, namely the nitrogen use efficiency and the nitrogen surplus for each of the N-budgets. The data showed that the nitrogen surplus increases for the soil < land < farm budget, while the nitrogen use efficiency decreases analogically for soil > land > farm budgets. The farm N-budget appeared to be the most relevant one giving a picture of the overall N management of agriculture and is accordingly recommended for integrative studies assessing the "nitrogen footprint" of society.

Based on the work of Leip et al. (2011b), we propose in this chapter three additional nitrogen indicators focusing even more on the use society in European countries makes of their productive land.

## 2 Methods

### *2.1 The CAPRI Model*

The core of the CAPRI (Common Agricultural Policy Regional Impact; Britz et al. 2007; Britz and Witzke 2008) model is an economic agricultural sector model aiming to analyse impacts of changes in the European Union (EU) (or international) agricultural policies on European agriculture and global agricultural commodity markets, typically in a forward looking analysis (8–10 years ahead). Technically, that economic core is a static, partial equilibrium model linking by sequential calibration a supply and a market module. The supply module of CAPRI consists of non-linear programming (NLP) models comprising about 50 crop and animal activities for each of about 280 NUTS-2-regions (EU 27, Norway, Turkey and Western Balkans) according to the Nomenclature des Unités Territoriales Statistiques (or short 'NUTS'). Each regional model maximises agricultural income at given prices and subsidies subject to resource constraints on land, feed and nutrient requirements as well as policy restrictions (box 'Supply Models' in Fig. 6.1). The global market module (Britz et al. 2007 page 96ff, see box 'Market Model' in Fig. 6.1) is formulated as a spatial multi-commodity model allowing for bilateral trade flows based on a "product differentiation by origin" assumption (Armington 1969). It explicitly represents major agricultural trade policy instruments such as (bilateral) tariffs, tariff quotas, and subsidised exports, and ensures price feedback in the overall model. The supply model in CAPRI iterates not only with the market model, but also with a young animal market model and a policy model in order to keep consistency between supply and the dynamics of animal herds as well as with premium calculations. Finally, an exploitation module calculates environmental and economic indicators describing the status of current and/or the impact of a changing policy.

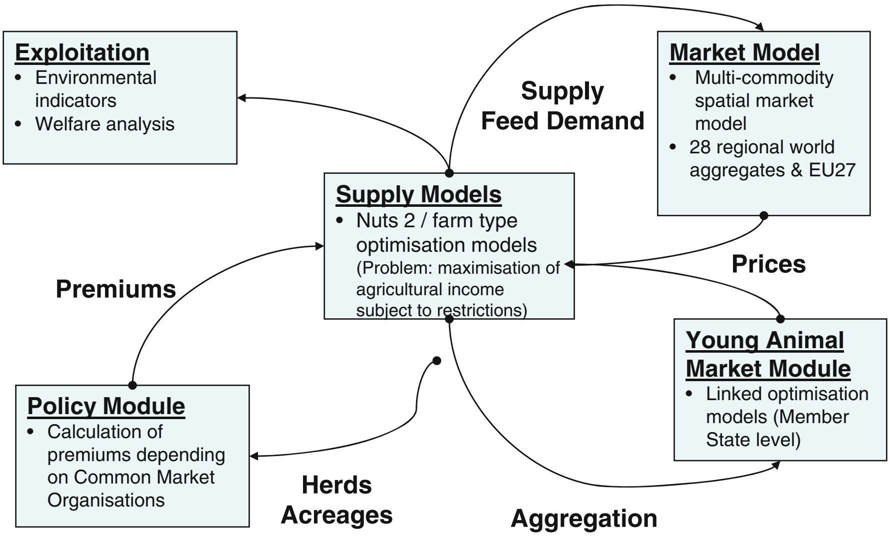


**Fig. 6.1** The CAPRI model chain (source: CAPRI Model Documentation, Britz et al. 2007)

For the application at hand, we are exploiting the CAPRI data base with a focus on N-mass balancing rather then making use of the simulation model. It is however important to note that appropriate modules ensure that the necessary elements for the nitrogen cascade calculations are available from the so-called baseline – an ex-ante outlook for agricultural market and farming practise – and counterfactual scenarios ex-post and ex-ante as well. The data base relies to the largest extent on harmonised, officially available data sets, mainly from Eurostat. Of specific interest in the following are land use, crop production, slaughter, herd size and market balance statistics. The data base build-up process ensures consistency (1) between output coefficients (crop and milk yields, slaughter weights), acreages respectively herd size and production, (2) between production and further positions in market balances, and (3) across regional scales. It also generates feed input coefficients for the different animal production processes which exhaust available feeding stuff quantities, and distributes manure and mineral fertilizer to crops. It requires to a certain extent corrections to officially reported data which are stemming from different domains and are based on different collection methodologies. Equally, missing values are replaced by estimates and obvious outliers corrected.

## *2.2 Nitrogen Balances in CAPRI*

In CAPRI, different parts of the agricultural sector are linked by the flow of (mass and) nitrogen: the crop sector receives manure nitrogen from the livestock sector in the exchange of animal feed; the animal sector receives feed and concentrates also from the agricultural market and sells products for processing and consumption; the industry produces synthetic fertilizer as major nitrogen input to agricultural soils that produce food, feed, fibre and energy for societal use. Nitrogen losses occur both in the livestock production system and from agricultural soils. The nitrogen balance of the livestock sector is closed by estimating manure nitrogen excretion as the difference from nitrogen intake with feed and nitrogen output (or retention) in products (see Eq. 6.1). Excreted manure can be managed or unmanaged. In the latter case manure is deposited on pasture or grassland by grazing animals. CAPRI considers only intentional application of manure to agricultural land as possible management option, ignoring other options such as the use of dried manure as fuel. However before application the manure is usually kept for varying time periods in animal stables and in manure management systems, where losses of nitrogen gases can occur. Thus, the manure balance (see Eq. 6.2) tracks the fate of the excreted manure for each region on the basis of available statistics on grazing pattern and existing manure management systems. The soil nitrogen balance (see Eq. 6.3) is closed by estimating soil nitrogen surplus from total nitrogen input and quantified nitrogen output, split into nitrate leaching, N accumulation in soils and denitrification ($N_2$ emissions) using emission factors. Both manure excretion and nitrogen surplus are cross-checked by independent data sources. The calculation of the N-cycle in CAPRI follows a mass-flow approach developed for the integrated nitrogen model MITERRA-EUROPE (Velthof et al. 2009). The model keeps track of the nitrogen available at each step – net of all emissions that occurred at an earlier step – and uses this as the basis for the estimation of emissions of $N_2O$, $NH_3$, $NO_x$, and $N_2$.

Thus, the nitrogen balance is determined by three equations, i.e. the animal balance, the manure balance, and the soil balance. These equations are closely linked to the overall market balance in CAPRI ensuring a full accounting of agricultural products (see Fig. 6.2).

**Animal-balance**

$$N_{man,ex} = \frac{N_{protein,req}}{6} - N_{animalproducts} - N_{animalwaste} \tag{6.1}$$

**Manure-balance**

$$N_{man,ex} = N_{grazing} + N_{application} + N_{gas,housto} + N_{runoff,housto} \tag{6.2}$$

**Soil-balance**

$$\begin{aligned} & N_{grazing} + N_{application} + N_{\min fert} + N_{crop\_residues} + N_{biofix} + N_{atmdep} \\ & = N_{crop} + N_{crop\_residues} + N_{gas.soil} + N_{runoff,soil} + N_{leach,soil} + N_{N2} + N_{accum,soil} \end{aligned} \tag{6.3}$$

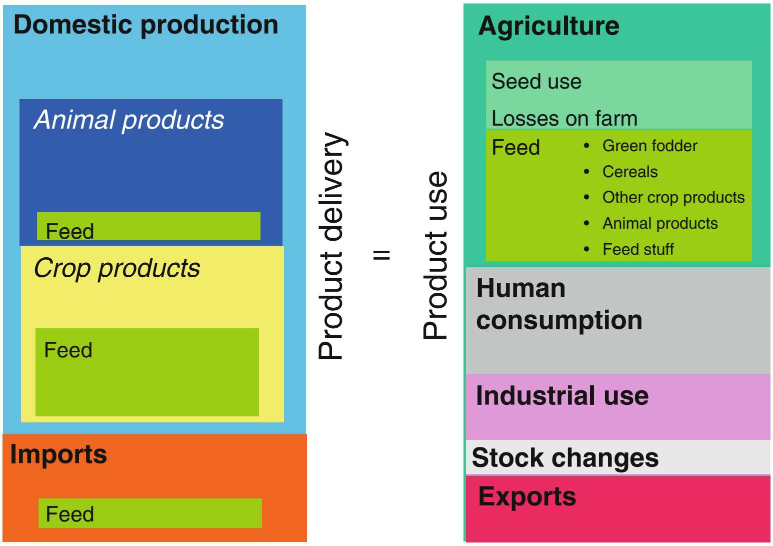


**Fig. 6.2** The CAPRI market balance ensures a full accounting of agricultural products for both dry biomass and nitrogen

where

$N_{man,ex}$: Excretion of manure by animals

$N_{protein,req}$: Protein requirement of the animals, assuming a nitrogen content in proteins of 1:6; the protein requirement should be linked to the feed use in the market balance (see discussion).

$N_{animalproducts}$: Nitrogen in animal products, obtained from the market balance

$N_{animalwaste}$: Nitrogen in animal waste (not manure)

$N_{grazing}$: Nitrogen deposition by grazing animals on pasture, range and paddock

$N_{application}$: Nitrogen in manure intentionally applied to agricultural land. Note that manure export and other uses of manure are not considered

$N_{gas,housto}$: Gaseous losses ($NH_3$, $NO_x$, $N_2O$) from animal housing and manure storage systems

$N_{gas,soil}$: Gaseous losses ($NH_3$, $NO_x$, $N_2O$) from soils

$N_{runoff,housto}$: Losses of nitrogen through runoff from animal housing and manure storage systems

$N_{runoff,soil}$: Losses of nitrogen through runoff from soils

$N_{crop\_residues}$: Crops residues returned to the soil. Crop residues are assumed to be the same at the start of the vegetation period (from last year's crop as input) and at the end of the vegetation period (from current year's crop as output) and thus does not need to be estimated in the soil-balance

$N_{crop}$: Nitrogen exported from the soil/vegetation system with harvest. $N_{crop}$ is obtained from the market-balance

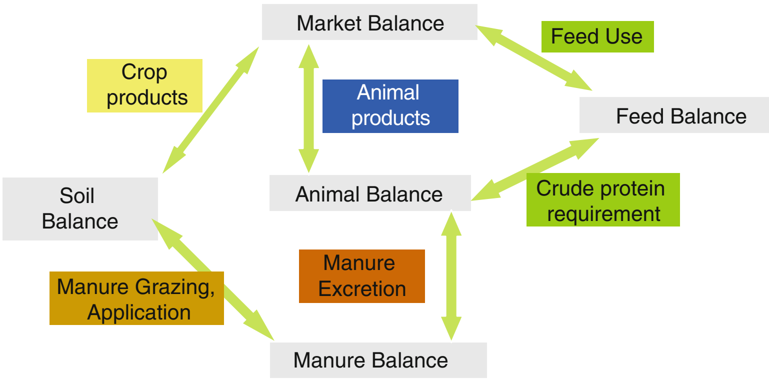


**Fig. 6.3** Linkage between the nitrogen balances in CAPRI

$N_{minfert}$: Application of mineral fertilizer nitrogen to agricultural soil
$N_{biofix}$: Nitrogen input through biological nitrogen fixation by leguminous plants
$N_{atmdep}$: Atmospheric deposition of nitrogen
$N_{leach,soil}$: Leaching of nitrogen from the soil to the below-root zone
$N_{N2,soil}$: Fluxes of $N_2$ (denitrification) from the soil
$N_{accum,soil}$: Accumulation (or depletion) of nitrogen in the soil (change of nitrogen stock)

The linkage between these balances is shown in Fig. 6.3. Each pair of N-balances is linked through a nitrogen flux, i.e. the manure balance links to the soil balance through manure deposition on grazing land and manure application, the animal and the soil balances link to the market balance through delivery of products and the animal balance links to the manure balance through manure excretion. In addition to the three balances described above, Fig. 6.3 shows also the feed-balance, which assures that each animal receives the protein it requires under the given performance (e.g. animal weight, milk yield etc.).

## 2.3 *Nitrogen Indicators*

We derive three nitrogen indicators on the basis of our calculations that deliver interesting insight into the efficiency and performance of agricultural systems in European countries. These indicators are to be seen as ancillary information to the main nitrogen indicators used, i.e. the nitrogen use efficiency and the nitrogen surplus. These indicators can be defined using soils, land and farm as system boundaries an related N budgets and are discussed in detail elsewhere

(Leip et al. 2011b). In this paper we focus on three additional indictors addressing specific questions:

– What is the fraction of annually fresh nitrogen added to the agricultural systems e.g. by mineral fertilizer and import of feed that is finally used for human consumption (New Nitrogen Conversion, *NNC*)?
– What is the fraction of total crop output that is used to feed the countries livestock, excluding or including grassland (Feed-Arable Ratio, *FAR*, and Feed-Crop-Ratio, *FCR,* respectively*)*?
– How much does agricultural production in a country depend on the import of products (Domestic share on N in Product, *DNP*)?

The indicators are calculated according to Eqs. 6.4–6.7. For *NNC,* the amount of new nitrogen is calculated using a broad interpretation, not restricting to recently generated reactive nitrogen through industrial or biological fixation and atmospheric deposition, but including also net import of nitrogen into the country. *FAR* and *FCR* calculate the ratio of crop output not returned to the soils (for example as crop residues), which is used for animal feeding versus its total output excluding N returned to the soil. Nitrogen from grassland, $N_{\mathrm{feed,grass}}$ is included in the calculation of *FCR*, but excluded in the calculation of *FAR*. This is because grassland is often not suitable for the production of arable crops. *DNP* calculates the ratio of nitrogen in crops, animal products and processed products imported to the country versus the total output of the agricultural sector, both from soils (including crop residues) and animal products.

**New Nitrogen Conversion**

$$NNC = \frac{N_{human}}{N_{biofix} + N_{atmdep} + N_{\mathrm{min}\,fert} + N_{import} - N_{export}} \tag{6.4}$$

**Feed-Arable Ratio**

$$FAR = \frac{N_{feed,cereals} + N_{feed,noncereals}}{N_{feed,cereals} + N_{feed,noncereals} + N_{crop}} \tag{6.5}$$

**Feed-Crop Ratio**

$$FCR = \frac{N_{feed,grass} + N_{feed,cereals} + N_{feed,noncereals}}{N_{feed,grass} + N_{feed,cereals} + N_{feed,noncereals} + N_{crop}} \tag{6.6}$$

**Domestic share on N in Products**

$$DNP = \frac{N_{crop} + N_{animalproducts}}{N_{import} - N_{export} + N_{crop} + N_{animalproducts}} \tag{6.7}$$

where

NNP: New Nitrogen Conversion, share of new nitrogen used in the agricultural systems of a country that is converted to products consumed by the human population
FAR: Feed-Arable Ratio, share of productivity on arable soils, excluding crop residues and permanent grassland, that is used to feed livestock
FCR: Feed-Crop Ratio, share of productivity on agricultural land, excluding crop residues but including permanent grassland that is used to feed livestock
DNP: Domestic share of N in Products, share of nitrogen consumed in products by humans or livestock that is produced domestically
$N_{human}$: Nitrogen consumed by human population, including pets-food, ornamental plants etc.
$N_{import}$: Nitrogen imported in agricultural products
$N_{export}$: Nitrogen exported in agricultural products
$N_{feed,cereals}$: Nitrogen in domestically produced cereals fed to animals
$N_{feed,noncereals}$: Nitrogen in domestically produced non-cereal crops (including fodder crops, but excluding grass)
$N_{feed,grass}$: Nitrogen in grass consumed by livestock

## 3 Results and Discussion

### *3.1 Regional N-Fluxes in Agriculture*

Examples for regional N-budgets for agriculture are presented in Fig. 6.4. We selected three countries with very different agricultural systems: (a) Ireland, which is characterized by extensive livestock production, predominantly fed by grazing and the production of meat; (b) Netherlands, which is characterized by intensive crop and dairy livestock production and (c) Romania, which is characterized by extensive crop production.

Irish agriculture is primarily a grass-based industry. Figure 6.4a shows that the production of green fodder provides three quarter of the animals feed, while only 15% of the ration is provided as feed concentrates. Ireland is the country where meat from cattle has the highest share of agricultural goods output in 2007, with 26.3% (Eurostat 2010; Olsen 2010). This is almost three times the average in EU27 of 9%. In the year 2002, for which we present the data, cattle accounted for 33% of the value of agricultural goods, in addition of 25% for milk. The three largest fluxes of nitrogen shown in Fig. 6.4a are the input of mineral fertiliser, and the cycling of fodder and manure between crop and livestock production, all about the same order of magnitude. Numerically, total soil N-surplus of 387 kt N year$^{-1}$ is almost equal to the annual input of nitrogen with manure, and by about 20 kt N year$^{-1}$ higher than the annual input of nitrogen with mineral fertiliser (see Leip et al. 2011b). Consumption of crop products in Ireland is estimated to amount to 16.8 kt N year$^{-1}$,

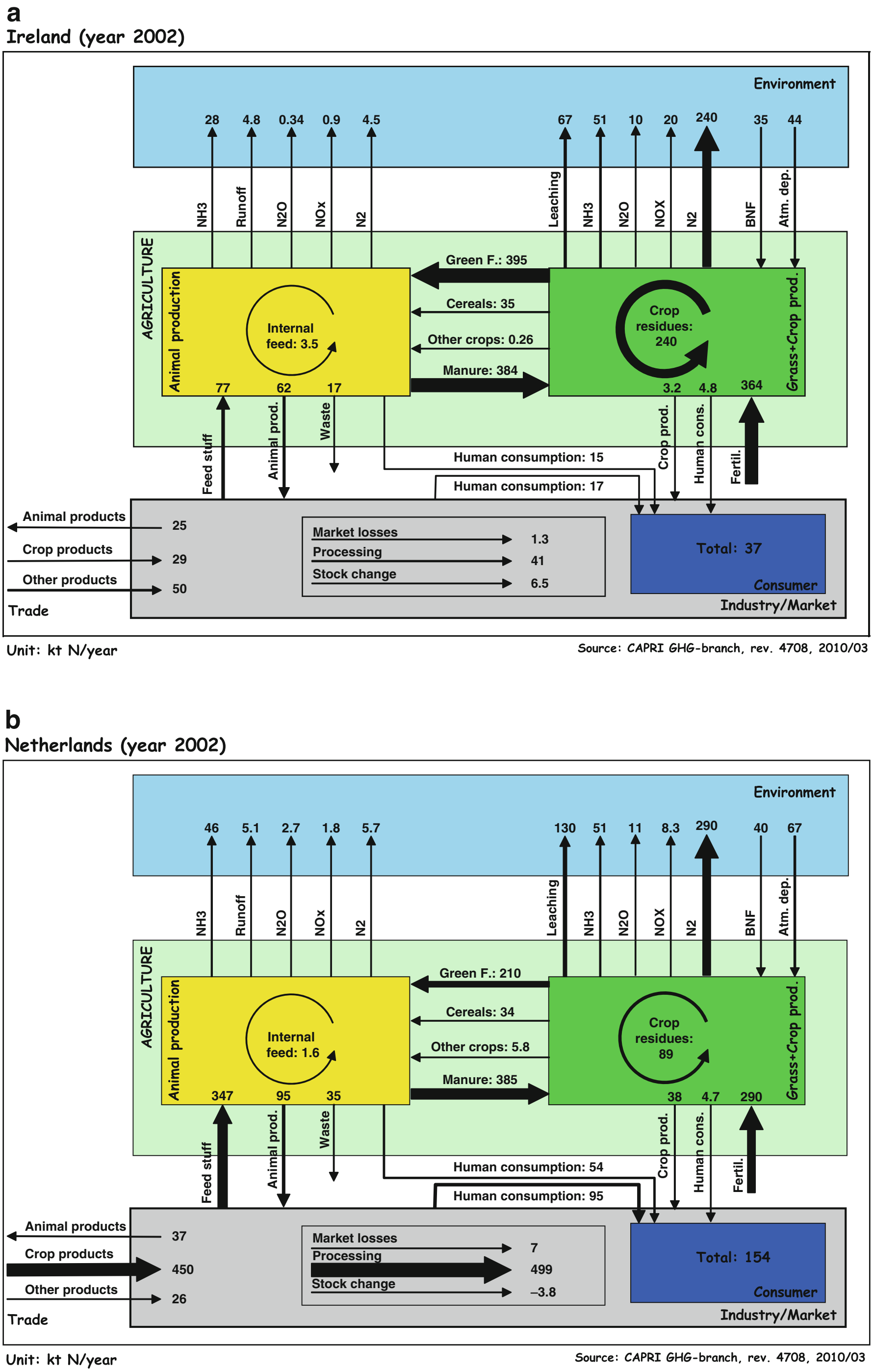


**Fig. 6.4** Regional N-budgets for agriculture in (**a**) Ireland, (**b**) the Netherlands and (**c**) Romania

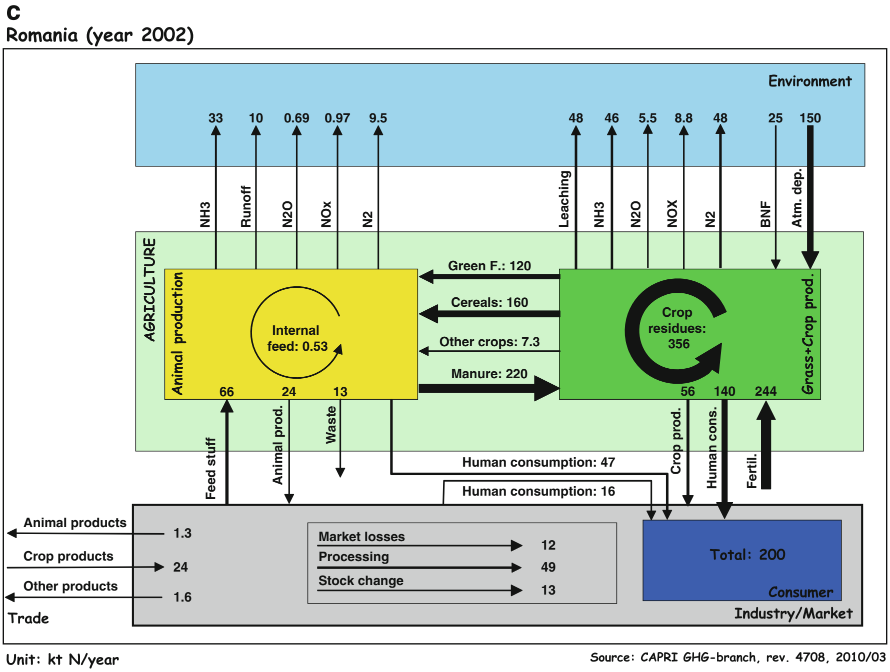


**Fig. 6.4** (continued)

out of which only about one quarter is domestically produced crop products, while the major part is imported.

Input of manure to soils in the Netherland is about the same as in Ireland (Fig. 6.4b). However, this is not matched by the same crop output and, therefore, the additional input of mineral fertiliser is only about 75% and total soil N-surplus is with 490 kt N year$^{-1}$ 25% higher than in Ireland. Instead, animal production in the Netherlands is based to a larger extent on imported feedstuff. Milk accounts for 20% of the value of agricultural goods produced in the Netherlands, about three times the value of cattle meat, but still less than the value of the produce of plant and flowers (30%, Eurostat 2010). This is reflected in the share of crop products marketed for direct human consumption (4.7 kt N year$^{-1}$) which accounts for just above 10% of the total N in crop output. Most of the crop products consumed are imported (60.6 kt N year$^{-1}$).

In Romania, crop production is extensive and more than 40% of the nitrogen in the crop output is recycled to the soil (Fig. 6.4c). Livestock production is a comparably small sector so that only one third of nitrogen input to soils comes from manure (220 kt N year$^{-1}$), less than mineral fertiliser (244 kt N year$^{-1}$), while one quarter of the total N-input to the soils is from atmospheric deposition. Nitrogen deposition is less than 10% for Ireland and the Netherland, and 10% for EU27. Animals are mainly fed by cereals (46%), with additions of green fodder (grass, silage maize and fodder beet) of 35% and feed concentrates of 19%. Also human nutrition in Romania is crop-product based, about 94% of which domestically produced and only 6% imported, while only 27% of the diet is based on animal or processed products.

### *3.2 New Nitrogen Conversion (NNC)*

In EU27, every year about 18 Tg N year$^{-1}$ are added to the agricultural systems. Most of it is added intentionally through the application of mineral fertilizer and manure. Over-supply of nitrogen in manure leads for some countries with intensive livestock production to the situation that manure application is not targeted at delivering nutrient to the soil, but rather for the disposal of the manure. From these 18 Tg N year$^{-1}$, 27% or 4.8 Tg N year$^{-1}$ are extractred from the agricultural system for human consumption. In CAPRI, human consumption includes also the delivery of ornamental plants, christmas trees, food for pets, etc. The range of NNC estimated with CAPRI for the year 2002 is between 7% for Ireland and 46% for Romania and 58% for Malta (Fig. 6.5). Agricultural production is very small in Malta, and thus most of the consumption is based on imports, which can directly be consumed with very little losses.

The NNC is an important indicator and very similar to the farm-NUE (Nitrogen Use Efficiency) defined by Leip et al. (2011b). The difference between both indicators is that the farm-NUE looks at the conversion of nitrogen input to agriculture to useful products, regardless whether these products are consumed within the country or exported. It is thus an overall performance indicator of the national farm. The NNC on the other hand looks how much nitrogen is needed for a society to satisfy consumer's demand. As the net import of nitrogen is regarded as new nitrogen, too, this is a good measure for the nitrogen efficiency of a society. The NNC however should not be regarded as an environmental indicator, as emission leakage effects are not addressed, as has been observed for Malta.

### *3.3 Feed-Arable Ratio (FAR) and Feed-Crop Ratio (FCR)*

A quantity of protein grown on soils can be used as food or it can be fed to animals to produce 'luxury' proteins (see e.g. Steinfeld et al. 2006) whereby only about 15% (global average, Galloway et al. 2010) to 25% (EU27, our data) of the proteins will be retained in animal products for consumption. Not all proteins, however, can directly be consumed by humans. Crops with high fibre content such as grass are made available to human consumption by feeding it to ruminants.

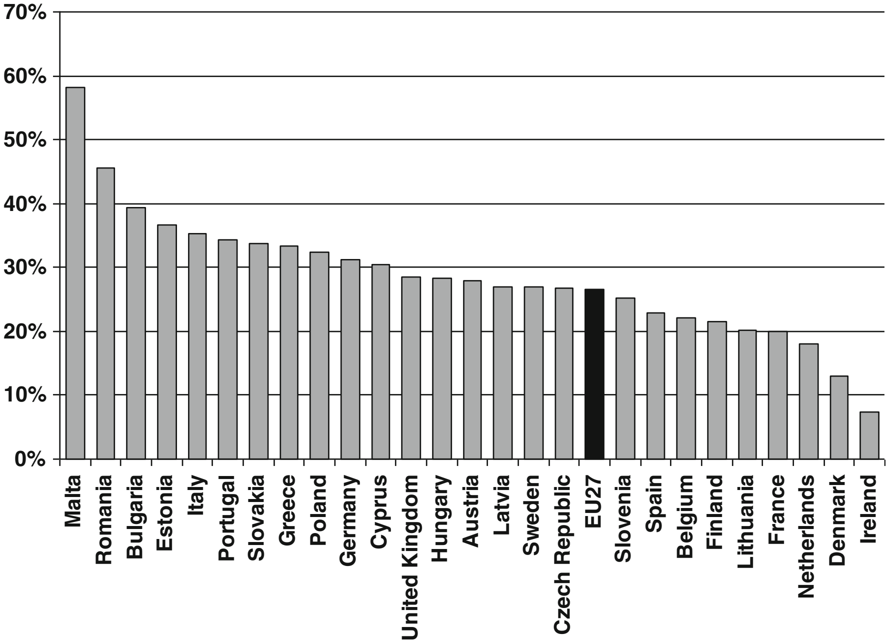


**Fig. 6.5** New nitrogen conversion (NNC) for EU27 countries

Equally, grazing might be a resource efficient way to transfer biomass into food especially where biomass generation per area is very low. The Feed-Arable Ratio (see Fig. 6.6), therefore, calculates the nitrogen used for feed only if it is grown on arable land and the product could have potentially be used directly by the consumer. Countries such as Ireland, Denmark, Belgium and Slovenia invest more than 80% of the nitrogen grown on its arable land to produce animals. Considering also proteins from grasslands (see light-grey bars in Fig. 6.6), this share would increase to almost 100%. Only few countries dedicate less than 40% of their proteins for animal production. On the average, close to 50% N comprised in arable crops is dedicated to livestock feeding or 70% of total N crop. Still this does not suffice to maintain the share of animal proteins in the diet of European citizen of more than 40% (Leip et al. 2011a) and must be supplemented by the import of feed concentrates from non-European countries. Again, these indicators do not quantify the overall environmental performance as imports of feed are not considered.

### 3.4 *Domestic Share of N in Products (DNP)*

Figure 6.5 suggests that EU27 is almost self-sufficient regarding N consumption, and need to supplement only a smaller part of the proteins consumed through imports.

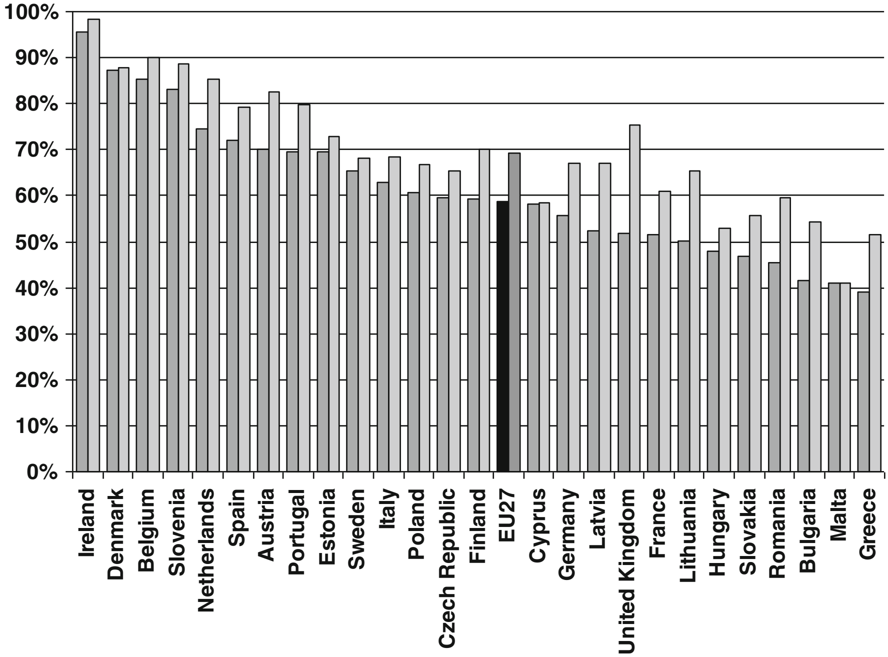


**Fig. 6.6** Ratio of crop products used for animal feeding and total crop output (Crop-Feed Ratio, CRF) for EU27 countries. Crop residues are not included. *Dark grey*: grassland included in value for feeding and total output; *light grey*: grassland excluded

Figure 6.6 shows that indeed 85% of the nitrogen that is consumed in EU27, processed or exported, is domestically produced, while the remaining 15% are imported (net import). A few countries are net exporting countries and have therefore a DNP of higher than 100%: Hungary, Czech Republic, Bulgaria and France. These are in particular those countries with a high share of crop production in their agricultural output. The countries specialised in livestock production, on the other hand, like Portugal, Slovenia, Netherlands and Belgium, and those countries with insufficient own agricultural production such as Malta, are characterised by a DNP of 60% or less.

## 4 Conclusion

Europe, thanks to in average favourite condition characterized by fertile soils, a rainfall amount and pattern allowing for high yields even under rainfed conditions and moderate temperatures, has the potential of very high agricultural productivity which allows provide sufficient food to its population. The diet of

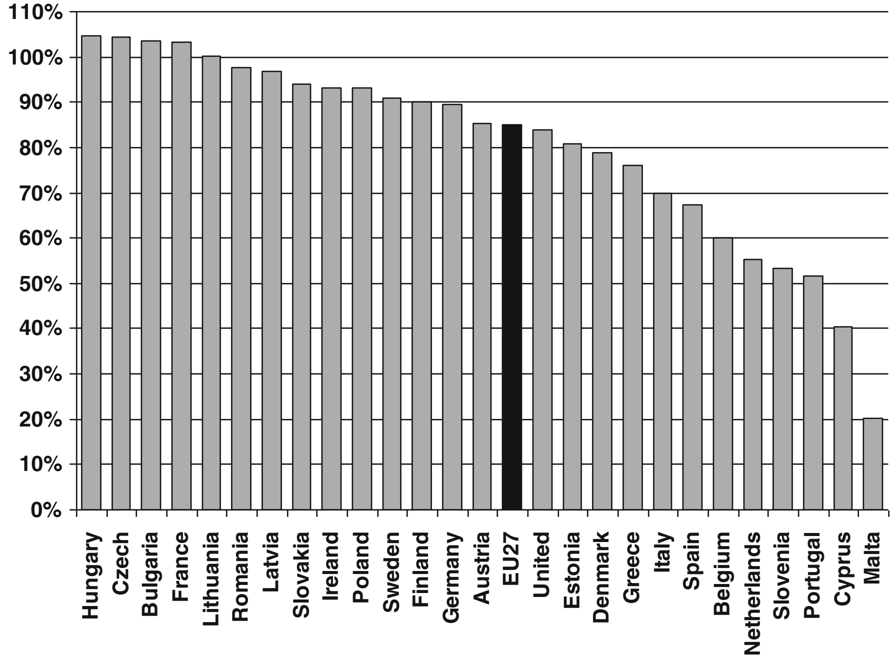


**Fig. 6.7** Ratio of nitrogen in imported versus domestically produced agricultural products (DNP) for EU27 countries

European citizens, however, is based to a large degree on 'luxury' proteins from animal products. Due to very high conversion losses in animal production, these animal proteins require a soil productivity at least four times what would be needed if the same amount of proteins were consumed in crop products. However, not all of the land's productivity can be exploited to grow crops that can be directly consumed. In marginal areas or mountainous regions ruminants are able to make land productivity available for human consumption. Still, about 60% of the productive land in Europe is used to grow animal feed.

The nitrogen performance of agriculture is mostly described by the two indicators of 'nitrogen surplus' and 'nitrogen use efficiency'. We proposed additional indicators focusing more on the use which the society in European countries makes of their productive land: the New Nitrogen Conversion (NNC), i.e. the share of new nitrogen used in the agricultural systems of a country that is converted to products consumed by the human population; the Feed-Arable Ratio (FAR) and Feed-Crop Ratio (FCR), i.e. the share of N produced on arable soils, excluding crop residues that is used to feed livestock, excluding or including permanent grassland, respectively; and the Domestic share of N in Products (DNP), i.e. the share of nitrogen consumed in products by humans or livestock that is produced domestically.

This short selection of examples shows that the CAPRI modeling system is a powerful tool for deriving nitrogen related agri-environmental indicators. These become even stronger if downscaled (Britz and Leip 2009; Leip et al. 2008) to the regional or watershed level.

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# Chapter 7
# Modelling Nitrogen Balance for a Regional Scale Livestock-Pasture System as a Discussion Support Tool

**U.B. Nidumolu, M. Lubbers, V. Alary, P. Lecomte, and H. van Keulen**

## 1 Introduction

Since the end of the 1980s, Réunion Island (located about 800 km east of Madagascar in the Indian Ocean), has developed intensive livestock farming in order to increase its self-sufficiency in food and to preserve agricultural employment. Although these objectives have been reached, considerable amounts of livestock effluent are produced. Because of the shortage of land suitable for spreading manure and the mismatch between the types of the manure produced and the needs of existing crops, livestock enterprises generate increasing risks of pollution. These include emerging conflicts with other activities, such as tourism, due to bad odour. Mastering the management of livestock wastes is therefore deemed necessary by local authorities (Aubry et al. 2006). As argued by Flamant et al. (1999), the sustainability of livestock farming should be assessed primarily in relation to local conditions, as representations of visible or potential crises originating from conflicting interests of animal husbandry and other local activities with respect to land use. According to Thornton and Herrero (2001) and quoted by Aubry et al. (2006), the likely trends of smallholder crop-livestock systems development within

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the next 20–30 years will require models to enable analysis of these complex systems, assess their impacts, and help farmers improve their performances. Currently, there is a major concern regarding agro-environmental issues. Farmers are viewed not only as food suppliers but also as the custodians of the countryside. This role of farmers has been officially acknowledged in the EU Common Agricultural Policy (CAP) through a number of regulations that enforce agri-environment schemes and cross-compliance (Pacini et al. 2004). A detailed study on the Nitrogen issues in livestock is given in the FAO publication (Steinfield et al. 2006).

In keeping tune with above trend, the objective in the development of the regional model was to calculate N as an environmental indicator which is introduced by two sources (i) by way of fertiliser for fodder requirements and (ii) by the manure excreted by the dairy animals. The objective is not model excess Nitrogen from the dairy sector but to calculate Nitrogen inputs and then discuss management options for Nitrogen utilisation with reference to environmental goals. The Nitrogen emanating from the dairy sector in this modelling project is seen more a management issue than a modelling problem.

## 2 Study Area

Réunion, one of the Overseas Departments of France is a volcanic tropical island about 800 km east of Madagascar. It covers an area of 2,512 km$^2$, of which about 1,004 km$^2$ is located above an altitude of 1,000 m above sea level (asl). As per the 2008 statistics (INSEE Reunion 2008), agricultural area was about *c*. 0.19 of total land area available (47,479 ha). Sugarcane is the main crop with an area of 24,528 ha in the lowlands (<800 m asl), permanent grassland covers 11,150 ha in the highlands and the remaining 11,801 ha are used for diverse cropping systems. On the island, four agro-climatological zones are distinguished: Plaine des Cafres, Plaine des Palmistes, Hauts de St. Joseph (consisting of Plaine des Grègues, Jean Petit, Grand Coude and La Créte) and Hauts de l'Ouest. In this paper, these areas are referred to as the Cafre, Palm, Joseph and Ouest sub-regions, respectively (Fig. 7.1). The main differences between these sub-regions in relation to the dairy industry are fodder types, fodder yields and fertiliser application. For example, temperate forages, such as ryegrass (*Lolium perenne*) are grown at altitudes above 1,000 m asl while tropical grasses such as chloris (*Chloris gayana*) and fodder sugarcane (*Pennisetum purpureum*) dominating at lower altitudes. One tropical species, kikuyu grass (*Pennisetum clandestinum*), covers a wide range of altitudes (from 800 to 1,500 m asl). These agroclimatic factors are incorporated in the model through relevant co-efficient matrices (for details please refer to Nidumolu et al. 2011).

The dairy sector on La Réunion started in the early 1970s. It has seen a significant growth in milk production from about 1 million litres in the 1970s to more than 23 million litres per year currently. On this island, dairy farming is

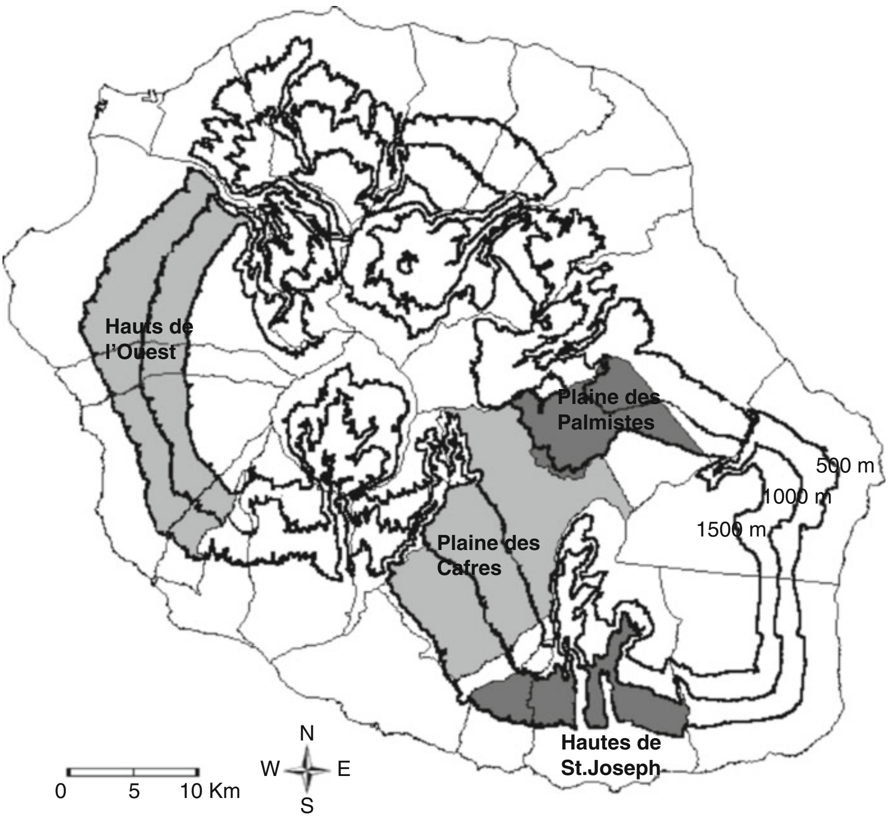


**Fig. 7.1** Agro-climatological zones of La Réunion (Source: Vayssières 2004)

promoted with financial and technical support from the European Union, with the French and local governments aiming at reducing dependency on imports of milk powder and dairy products (D'Haese et al. 2009). A number of subsidies are available to encourage farmers to take up dairying, for both social and economic reasons, such as reducing population pressure in the coastal areas by encouraging settlement at higher altitudes.

Currently, the dairy sector consists of 119 dairy farms with about 5,700 dairy animals (4,100 are milk cows). The Plaine des Cafres is the main dairy area of the island, with more than half the dairy farms located in this sub-region, while the sub-region Hauts de l'Ouest has the smallest number of farms. Average annual milk yield is about 5,500 kg per cow. The main dairy cattle breed on the island is Holstein, with a genetic potential of about 8,000 kg per lactation.

This work reported in this paper developed as a progression from the farm-scale modelling work carried out during 2000–2004 in Réunion island (Louhichi et al. 2004; Alary 2004).

# 3 Method

The model was developed in a linear programming with multi-purpose method. It was implemented under GAMS “General Algebraic modelling system” with objective function which will be optimized by taking account of the different constraints existing on the farm and in the region where the model performed. These constraints include: land, labour, etc. Ksheera is a mechanistic model, using a normative approach (Jansen and van Ittersum. 2007). It performs dynamically with a step of 6 months to calculate the objective function. The main objectives of the model are to optimize: (i) income, (ii) nitrogen excess, (iii) labour hours. The classical one-dimensional approaches are less effective because of multi-parameters which should be taken into account before making decisions. As the model uses a normative approach, the bench marking with the reality is difficult (Hazell and Norton 1986). The comparison is based on a reference year and the validation was attempted for the first 6 months of this year. The error rate is less than 10%. Environmental policy is included in terms of N management and the indicator chosen is N contributed from the dairy farms per ha.

## *3.1 Description of Model Components*

The dairy model (Fig. 7.3) centres on the nutritional requirements of the dairy cows while considering the genetic potential for milk production (VL) of the local cows (4,000–8,000 kg per cow per year – as an example, the term VL40 refers to cows with a potential of 4,000 kg per year, etc.). The calved and heifers are categorised in age groups from Gen1 (calves <6 months) to Gen5 (heifers 25–30 months old). Nutritional requirements vary as per the gae group of the animals, these are derived from monitoring of the weekly intake conducted in 1998–2000 (Hassoun et al. 2000). Nutrient contents, expressed in UFL (forage units for milk production), PDIN (digestible protein intake from nitrogenous components), PDIE (digestible protein intake from energy components), CA (calcium), PHO (phosphorus), and CB (crude cellulose) are defined for both roughage and concentrate rations. The nutritive value of the available fodder types and concentrates has been derived from monthly feed analyses (Grimaud and Thomas 2002). As discussed earlier, fodder types and yields (tonnes/ha) are different for the different sub-regions. This variation across sub-regions is included in the model. Fertilisation and mechanisation requirements in terms of both, labour and costs, for each forage type are quantified on the basis of (1) information collected in a household survey conducted in 2000 and (2) data collected by the Dairy Cooperative (UAFP) in the study area. Labour requirements for fodder and dairy cow management are considered from both, a labour utilisation and a cost point of view. Costs for fertilisers, concentrate

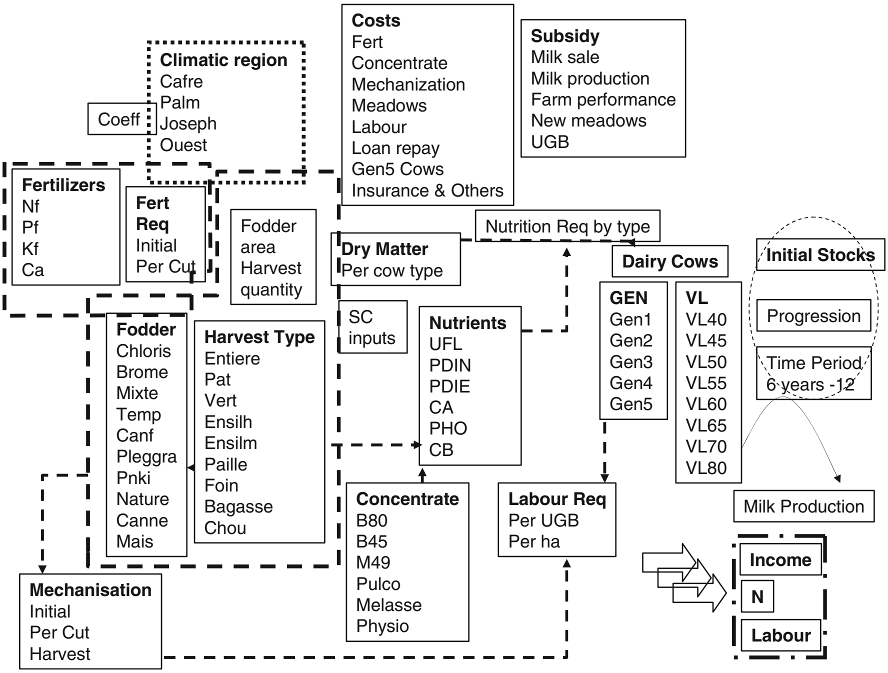


**Fig. 7.2** Model components (Nidumolu et al. 2011)

supplements, machinery, pasture establishment and maintenance, labour, interest and loan repayment, veterinary services, insurance and other inputs are taken into account. Revenue (income) in the model comprises proceeds from sales of milk plus subsidies for maintaining a herd (as a function of animal density), milk production, farm performance, establishment of new pastures and pasture maintenance.

For details of the social and economic components of the model please refer to Nidumolu et al. 2011 Fig. 7.2.

## *3.2 N in Fertiliser Application*

According to the structure of the model the fertiliser use per fodder type per ha both for initial application and after each cut is fixed according to the empirical studies conducted on the grass lands in Réunion. These are given in tables in Annexe 1. The fertiliser types are also fixed in the model such as A301010, A14736, A162912, A151224, A102020 (*fertilisers with N, P, K)*. For example the N content in the fertiliser A301010 is 30%. Depending on the type of fodder chosen by the model (as discussed the type of fodder selected is a function of nutrition value and the sub_region) the fertiliser use is calculated by the following equation.

*Fertiliser use per period per sub_region per fertiliser type (tonnes)*

$$\begin{aligned} Fert\_Use\_SR\_Ferttype = & \sum_{ft,ht,mr,fertiliser} \left(Fert\_\mathrm{Req}\_Initial_{ft,meadows}\right. \\ & \left. * Fert\_Coeff_{ft,sub\_region} * FX_{period,sub\_region,ft,ht,mr,fertiliser}\right) \\ & + \sum_{ft,ht,mr,fertiliser} \left(Fert\_\mathrm{Req}\_Cuts_{ft,meadows}\right. \\ & \left. * Fert\_Coeff_{ft,sub\_region} * FX_{period,sub\_region,ft,ht,mr,fertiliser}\right) \end{aligned} \tag{7.1}$$

Fertiliser requirements are different in the first 6 months and second 6 months of the year

And based on the type of fertiliser use the N content is calculated using the following equation

*Fertiliser nitrogen content per period per sub_region (tonnes)*

$$\begin{aligned} Fert\_Nitrogen\_SR_{period,sub\_region} = & \sum_{fertiliser} \left(Fert\_Use\_SR\_Ferttype_{period,sub\_region,fertiliser}\right. \\ & \left. * N\_Content_{fertiliser}\right) \end{aligned} \tag{7.2}$$

Applying these Eqs. 7.1 and 7.2, N application per ha per period can be calculated by dividing the total N used per period by the land area used per period. These values can then be used calculate N application by fertiliser per ha per year.

## 3.3 N in Manure

Advances in milk production and the expansion of dairy herds have increased the need for improved manure management (in this study focusing on N). This is especially relevant in relatively intensive dairy farming in Réunion Island. Most studies in literature focus on calculating the manure production on a farm scale. However, as the objective of the current study is modelling on a regional scale, the manure production was also attempted to be calculated at a regional scale. Intake and digestibility of feed DM and protein have a significant impact on excretion (Wilkerson et al. 1997). In the current model, nutrition requirements of the dairy animals are fixed for milking (VL40-VL80) and non-milking cows (Gen1-Gen5). Based on the values of Protein intake (PDIN), the N in protein is calculated as 22% of the PDIN intake. Using the following equation the N intake of the cows

**Table 7.1** N intake and excretion in manure of Holstein cattle (Beltsville, MD)

| | ASAE[a] standard for dairy cattle | | Cows averaging 29 kg/day of milk[b] | | Cows averaging 14 kg/day of milk[c] | | Non-lactating cows | | Growing and replacement cattle | |
|---|---|---|---|---|---|---|---|---|---|---|
| Measurement | Mean | SD | Mean | SD | Mean | SD | Mean | SD | Mean | SD |
| Total manure | 86 | 17 | 89 | 22.5 | 65.9 | 17.3 | 34.8 | 11.1 | 67.5 | 18.5 |
| Feces | 60 | | 60 | 18.1 | 41.2 | 13.8 | 15.1 | 7.4 | 32.6 | 10 |
| Intake N | | | 0.787 | 0.182 | 0.549 | 0.14 | 0.254 | 0.086 | 0.53 | 0.181 |
| Total excreta N | 0.45 | 0.096 | 0.542 | 0.146 | 0.399 | 0.116 | 0.237 | 0.077 | 0.447 | 0.153 |
| Fecal N | | | 0.27 | 0.077 | 0.192 | 0.049 | 0.077 | 0.029 | 0.193 | 0.062 |
| Urinary N | | | 0.272 | 0.093 | 0.208 | 0.09 | 0.16 | 0.056 | 0.254 | 0.11 |
| Milk N | | | 0.234 | 0.053 | 0.121 | 0.04 | | | | |

Reproduced from Wilkerson et al. (1997) [a]American Society of Agricultural Engineers [b]Milk production >20 kg/day [c]Milk production ≤ 20 kg/day

(Gen1-Gen5 and Vl40-Vl80) is calculated per period based on the number of cows (cow type) per period and their nutrition intake. Then annual intake of N is calculated per cow type.

$$
\begin{aligned}
&N\_IntakeCows\_SR\_CT_{period,sub\_region,cow\_type} = 0.183 * 0.16\\
&\quad * \sum_{cat\_animal(<=5)} (CowNutrition_{cow\_type,MAT} * AX_{period,sub\_region,cat\_animal}\\
&\quad + \sum_{milking\_cow(>5)} (CowNutrition_{milking\_cow,MAT} * VL\_\mathrm{Pr}ogression\_SR_{period,sub\_region,milking\_cow})\\
&\quad (CT\ Cow\ Type)
\end{aligned}
\tag{7.3}
$$

The above equation is used to calculate the N intake per cow type per period per sub_region.

The following equation summarises the N intake per sub_region per period

$$
N\_IntakeCows\_SR_{period,sub\_region} = \sum_{Cow\_Type} (N\_Intake\_Cows\_SR\_CT_{period,sub\_region,cow\_type})
$$
$$
(SR\ is\ Sub\_Region,\ CT\ is\ Cow\ Type) \tag{7.4}
$$

After deriving the annual intake of N, N excreted by way of manure, urine and milk is calculated. The N in manure is of interest as it is subject to management options. The N excreted is calculated based on the analytical study conducted by Wilkerson et al. (1997) in the USA on Holstein cattle. The summary is given in the following Table 7.1. The data available from the following Table 7.1 is given for high yielding cows yielding 29 kg/day (=8,700 kg/year – 300*/29 – in the present model

**Table 7.2** Coefficients of N excretion

| Cow type | Excretion | Manure | Urine | Milk |
|---|---|---|---|---|
| gen1 | 0.45 | 0.32 | 0.68 | 0 |
| gen2 | 0.59 | 0.32 | 0.68 | 0 |
| gen3 | 0.72 | 0.32 | 0.68 | 0 |
| gen4 | 0.79 | 0.32 | 0.68 | 0 |
| gen5 | 0.88 | 0.32 | 0.68 | 0 |
| VL40 | 0.73 | 0.48 | 0.52 | 0.121 |
| VL45 | 0.71 | 0.48 | 0.52 | 0.139 |
| VL50 | 0.70 | 0.48 | 0.52 | 0.157 |
| VL55 | 0.69 | 0.48 | 0.52 | 0.175 |
| VL60 | 0.69 | 0.48 | 0.52 | 0.193 |
| VL65 | 0.69 | 0.48 | 0.52 | 0.211 |
| VL70 | 0.69 | 0.49 | 0.51 | 0.225 |
| VL80 | 0.69 | 0.49 | 0.51 | 0.234 |
| | factor | % | % | Factor |

From the analysis of Wilkerson et al. (1997) Excretion as Manure + Urine is coeff times Intake Manure + Urine are in % ages Milk N is intake times the coeff (and is diff from excretion)

we can relate these to to VL80, VL70), cows yielding 14 kg/day (= 4,200 kg/year – 300*14 – in the present model this can be related to VL40, VL45), non-lactating cows and growing and replacement cattle (these refer to Gen-Gen5 in the present model).

Based on the study by Wilkerson et al. 1997, the following Table 7.2 is generated with values extrapolated to the cow types used in the current model. The N excretion factor (in Table 7.2 below) is used to calculate the amount of N excreted with reference to N intake. The N in Manure and Urine are calculated as % of the N excretion.

Using the values in Table 7.2, the N excreted is calculated by means of the following equations

* N in Excretion Total (Manure and Urine) calculation for Cow Type

$$N\_Excretion\_SR\_CT_{period,sub\_region,cow\_type} = N\_Out\_Coeff_{Cow\_Type,Excretion} * N\_IntakeCows\_SR_{period,sub\_region} \quad (7.5)$$

* N in Manure calculation

$$N\_Manure\_SR\_CT_{period,sub\_region,cow\_type} = N\_Out\_Coeff_{Cow\_Type,Manure} * N\_Excretion\_SR\_CT_{period,sub\_region,cow\_type} \quad (7.6)$$

* N in Urine calculation

$$N\_Urine\_SR\_CT_{period,sub\_region,cow\_type} = N\_Out\_Coeff_{Cow\_Type,Urine} * N\_Excretion\_SR\_CT_{period,sub\_region,cow\_type} \quad (7.7)$$

* N in Milk calculation

$$N\_Milk\_SR\_CT_{period,sub\_region,cow\_type} = N\_Out\_Coeff_{Cow\_Type,Milk} * N\_IntakeCows\_SR\_CT_{period,sub\_region,cow\_type} \quad (7.8)$$

The values derived from the above equations are then summed up over the sub_region per period-year which gives the value of N in manure, urine and milk.

## 4 N Calculation Outputs

Based on the calculation for N (by way of fertiliser and manure as described earlier) the following outputs are derived for the four sub_regions for Year 1-Year 6 (for base scenario as discussed in the following chapters) and are given in the Table 7.3 below.

### *4.1 Model Validation*

The model calculation for N by manure is compared to the farm scale data collected by Vayssières et al. (2006) (Table 7.4). Though the data could not be compared for the same year 2006 for want of a survey, it is still considered relevant to check the overall tendency with the model calculations.

A fairly good match is observed for the average N by manure for the region. N by manure differs by about 4.5% between the current model calculations (for year 1) and the data collected at a farm-scale. However, it is to be noted that only 36 farms are considered in the farm-scale data while the regional model considers all the farms in the island. Therefore, the difference could be higher or

**Table 7.3** Nitrogen (kg/per ha) calculated from N in fertiliser and N in manure

| Year | Cafre | Joseph | Ouest | Palm |
|---|---|---|---|---|
| 1 | 320 | 236 | 141 | 347 |
| 2 | 336 | 240 | 149 | 364 |
| 3 | 330 | 234 | 207 | 361 |
| 4 | 341 | 242 | 215 | 375 |
| 5 | 339 | 240 | 214 | 374 |
| 6 | 343 | 242 | 215 | 376 |

**Table 7.4** Comparing N by manure by model calculation and data collected at farm scale

| Farm scale data (kg/ha) | Model calculation (kg/ha) | Difference in kg/ha | % difference |
|---|---|---|---|
| 162.5 | 155.5 | −7 | −4.48 |

even lower when all the data is considered. It is also observed that in the Ouest the rate of mineral fertiliser application is far higher than the stipulated norms (see tables Initial Fertiliser requirement (in tons/ha), Fertiliser application after cuts (in tons/ha), Coefficient of fertiliser use per sub-region in Annexe 1), which are followed in model calculations. It is also noted that organic manure from piggeries is imported into the dairy farm lands of the Ouest which is not taken into account in the current dairy model. These validation figures therefore are to be seen as a broad reference check on the model calculation and not as absolute verification.

## 5 Discussion and Conclusions

The sustainability of dairy farms will depend increasingly not only on profitable milk production but also on farmers' ability to comply with nutrient management regulations (Powell 2003). The model calculates the N per ha by both fertiliser and manure. The innovation is the dynamism of the model (both land and animals) over a time-step of 6 months. By a logical extension of the dynamism of land and animals, the fertiliser use (and N use) and manure N excreted by animals is dynamic over the same time step. N in the context of Réunion is a management option than a modelling problem. Therefore, the idea is to use the results of the model in terms of N calculations to discuss N management options. The driving forces for discussing these options are EU Nitrate Directive, fertiliser costs, manure transportation issues, transformation options (compost for ease of transport, efficiency etc) and manure as a source of energy. The management options are discussed based on the empirical values calculated by the model. The options that are available to manage N generated from manure are as follows:

(i) Increase of spreadable land area for manure
(ii) Transform the manure to other forms such as compost
(iii) Utilise manure as a source of energy

Figure 7.3 below summarises an example where the N per ha in manure is calculated (for year 1) and the scenarios in which the land area is increased by 20% or 30%. The increase could be in dairy sector or as shown in the Fig. 7.3 below, the manure can be exported to sugarcane sector or vegetables sector. A 20% increase in spreadable area will reduce the N per ha by manure from 156 to 130 kg/ha/year in Plaines des Cafre area for example. Likewise the manure N per ha is reduced in other sub_regions as well.

The fact that manure can be used to reduce the intensity of fertiliser use per ha is one of the options that may be considered in the management options. This aspect needs further work on the fodder growth models with manure application and the social acceptability of using manure more extensively than at present. The fodder growth models, which establish a relationship between

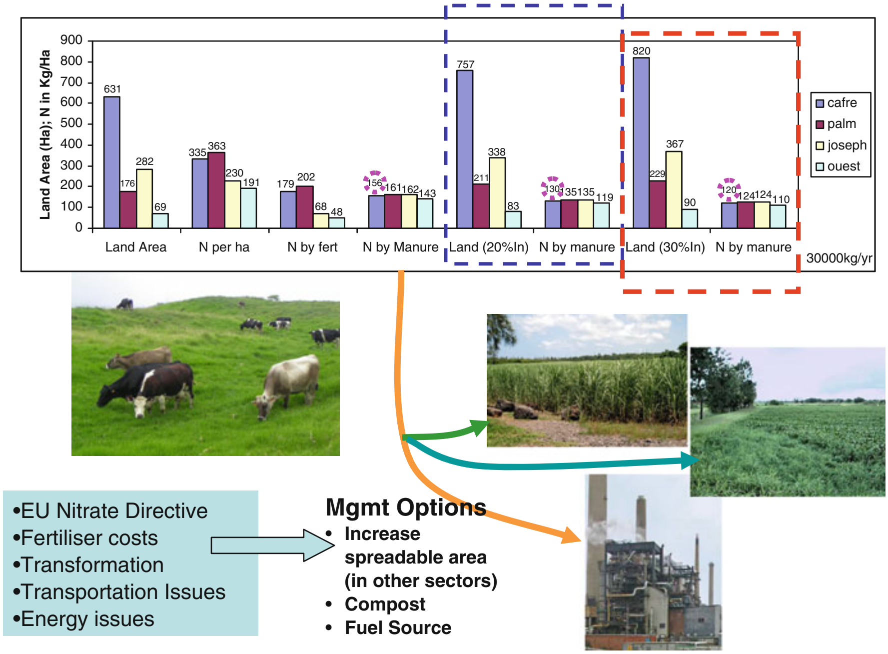


**Fig. 7.3** N in manure and management options

applications of manure-fertiliser combination to fodder yields, can then be integrated into the current model.

Other options such as transformation of the manure to compost are a management option in terms of cost-benefit analysis, creating storage facilities, transportation among others. There has also been an ongoing discussion into transforming the manure to pellets as a source of fuel for generating electricity. This is also a management option which has to be explored again in terms of cost-benefit and in a larger context of energy source.

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# Chapter 8
# On-Farm Weather Risk Management in Suckler Cow Farms: A Recursive Discrete Stochastic Programming Approach

**C. Mosnier, J. Agabriel, M. Lherm, and A. Reynaud**

## 1 Introduction

The 4.3 million French suckler cows represent more than one third of all European suckler cows and supply around 60% of the beef production in France. They also participate in rural development, as few economic alternatives to livestock farming exist in these production areas and they help in maintaining large areas under grassland which favors biodiversity and limits pollution and erosion (Le Goffe 2003), even if their complete environmental impact should be taken into account (FAO 2006). However, these farms rely on grassland production which is very sensitive to weather conditions (Gateau et al. 2006). Currently the EU and France are thinking at introducing a risk management framework into their agricultural policy. Since farmers individual risk-management strategies can supplement or replace public compensation policies and private insurance, they have to be well understood. Farm risk management aims at profitably securing and improving farms potential of profit over time. It encompasses two stages. The first one, prior to the realisation of a random event, deals with the mitigation of future risks of loss. The second stage, subsequent to the realisation of this uncertain event, corresponds to decisions adjustments in order to take advantage or to limit damages caused by the random event. These two stages are interlinked since first stage decisions can reduce for instance farm exposure or increase adjustments capacity.

In the case of French suckler cow farms, numerous production options exist to manage risks linked to weather conditions. Strategic decisions to mitigate risks encompass land allocation, average herd size and herd composition (Lemaire et al. 2006a; Mosnier et al. 2009). The definition of an appropriate level of animal

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stocking rate, of the source of feed supply (Lemaire et al. 2006b) and of calving date (Pottier et al. 2007) are crucial too. Adjustments are very diverse and concern for instance animal diets (Blanc et al. 2006; INRA 2007), animal sales, end use of crop production (Le Gall et al. 1998) or feed purchases and sales (Veysset et al. 2007).

The aim of this paper is to better understand how on-farm risk reducing strategies encompassing both risk anticipation strategy and risk modify the production system and profit distribution of French suckler cow enterprises.

Both econometric and mathematical programming methods can be used to model risk management. Although econometric models have the advantage of being based on statistic inference, they are hardly able to represent the sequential decision making process (REF) and to disentangle the complex relationships between the different components of the systems. In the vast literature devoted to farm modelling under uncertainty, two well known approaches can be distinguished: Discrete Stochastic Programming (DSP) and Stochastic Dynamic Programming (SDP). Previous bio-economic livestock farm models using DSP approach are though limited by the number of decisions stages introduced and by their short time span (Lambert 1989; Kingwell et al. 1993; Jacquet and Pluvinage 1997; Lien and Hardaker 2001). Livestock farm models using a SDP approach have to reduce the number of activities considered (Moxnes et al. 2001; Kobayashi et al. 2007) since model size explodes with the number of dynamic variable. To overcome limitations of the previous approaches, we propose to use a sequence of recursive DSP model in a way somewhat in the line to the proposal of Blanco and Flichman (2002) and to use this framework to simulate successive stochastic weather events over a long period.

The remainder of the paper is organized as follows. We describe first how the production system and the decision making process are modelled. We simulate then different weather risk management strategies according to farmers risk aversion and market hay price. A simulation of stochastic weather conditions observed over the period 1990–2007 is then simulated.

## 2 Model Description

Our model aims at simulating long-term strategies to manage weather risk in a suckler cow enterprise as well as the impacts of successive random weather conditions on annual technical and economic results.

The production system modelled consists of beef cattle production based on a suckler cow herd, combined with grassland crop production (Fig. 8.1). This modelling of the production system is based on the framework presented in Mosnier et al. (2009). Barn capacity, herd size and herd composition, herd live weight and animal feeding, haymaking and feed stock management are optimised for 100 ha of grassland. To represent farmers decision making, we assume that they optimise their decisions over a 3 year planning horizon. In this model, famers anticipate that grassland yield could be either favourable or unfavourable and that they will be able to adjust their decisions. In addition, each 2 months, the decision plan can be partly

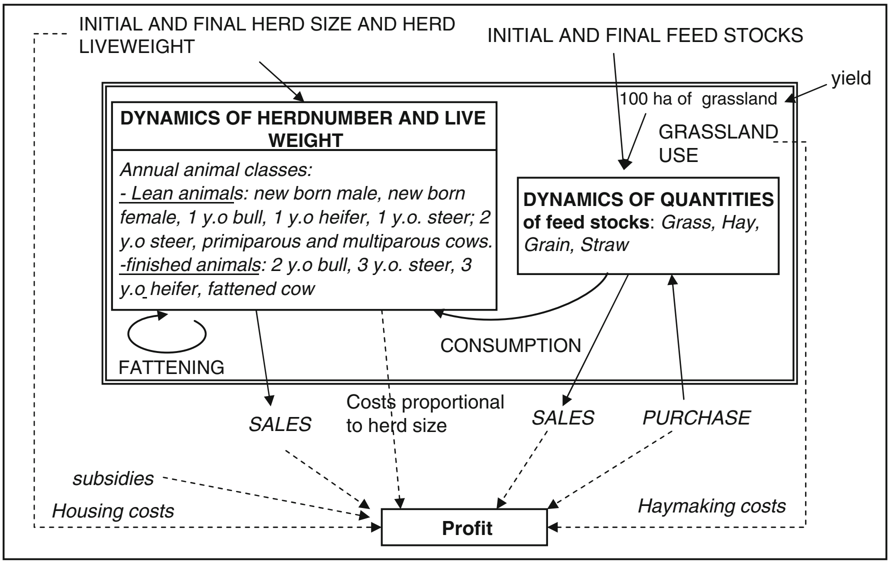


**Fig. 8.1** Representation of the modelled production system (optimised decisions are in *capital letter*)

revised based on the observed grassland yield. Technically, this is modelled thanks to a recursive sequence of discrete stochastic programming optimisations.

Our model is parameterized to represent suckler cow enterprise of farms located in the Northern part of the Massif Central which is the most important production area of the Charolais breed in France. It is resolved by the non linear programming solver *CONOPT* run in *Gams*.[1]

## 2.1 Farmer's Time and Risk Anticipations

Farmer decisions depend on their expectation regarding their future profit. The future encompass two dimensions: the possible weather conditions anticipated for each period and the length of the time horizon.

We assume that farmers only anticipate two states of nature for weather conditions: one corresponding to a favourable year and the other one to an unfavourable year. Two kinds of risks can be anticipated: embedded risk which occurs when farmers plan to adjust their decisions following the realization of an uncertain event, and, non-embedded risk if risks are expected to affect profit but

[1] GAMS development Corporation, 1217 Potomac Street W; Washington, DC 20007, USA. www.gams.com

without real possibility for the farmer to reduce their impacts *a* posterior (see also Hardaker et al. 2004). Previous works (Mosnier et al. 2009, 2010) emphasized that grassland yield shocks in the French Charolais area involve many adjustments of the production systems, namely adjustments of animal diet composition, of feed product trade and haymaking. Hence, weather shocks are introduced as embedded risks. Introduction of embedded risks involves that bimonthly decisions are differentiated after the realisation of the weather event. Such a risk representation is demanding in terms of computational capacity. In order to take account impacts of successive weather events while keeping the model tractable, we introduce weather risk for the two first years of the planning horizon. This means that the 'modelled farmers' foresee how they would react if two bad years or two good years occur in a row. Let's $\zeta$ be the weather risk with $\zeta 1$ and $\zeta 2$ being the random weather condition for respectively the year $t1$ and $t2$. They are characterized by two states of nature $\zeta 1$: $\{\zeta 1+;\ \zeta 1-\}$ and $\zeta 2$: $\{\zeta 2+;\ \zeta 2-\}$. Each decision depending on weather realisation is indexed by both risks $\{\zeta 1;\ \zeta 2\}$. However, the first year decisions depend only on the realisation of the first year weather event, while the second year decisions could differ according to the realisation of year 1 and year 2 weather events. Weather conditions directly influence grassland yields. As a consequence, we use grassland yield as an indicator of weather conditions. Grassland yield distributions correspond to annual estimation by Agreste (statistics from the French ministry of agriculture) in the Charolais area, over the period 1990–2007. Unfavourable event is set to average yield plus one standard deviation and an unfavourable one equals to average yield minus one standard deviation.

An infinite horizon or a very long planning horizon is often thought preferable since it can influence the long term equilibrium and how fast it is reached (Dawid 2005). However, in our case, the initial state of the system is optimised and corresponds already to the equilibrium state under current information. No long term adaptations are expected to be simulated but only variations around equilibrium. The issue is then to set a time horizon long enough to enable the system to recover from shocks while not giving too much weight to non risky years compare to risky ones (only the two first years are associated to embedded weather risks). We fix the planning horizon at 3 years which appear to us as a good compromise: it implies only 1 year without risk and enable for instance to sell more cows provided that less heifers would be sold the following years (the number of animals do not have to vary between the beginning and the end of the simulation). Let's $t$ being the years anticipated with $t$ $\{t1,\ t2,\ t3\}$.

## *2.2 Description of the Production System*

### 2.2.1 Animal Production

To cover the range of animal production in the Charolais production area, 13 annual animal classes characterized by sex (male, female or castrated male), age (from new born to mature) and production objective (fattening or storage) are introduced in the model. Classes, indexed by *a,* are described by two endogenous dynamic variables: the number of animals and their average live weight. The initial number of animals in each class is optimised under the constraint that the repartition of animals could be maintained over time i.e. that there is for instance at least as many new born heifers and 1 year old heifers than 2 year old heifers.

This initial repartition is chosen one for all. However, the (1) bimonthly control of animal sales, (2) bimonthly choice of animal diet composition and diet energy content and (3) annual fattening objectives could be adjusted to face weather events. The intra year animal number dynamics are defined by the motion function *f*. For each period *p* of year *t* and animal class *a*, this function draws the balance between past number of animals (*ANb*), sales decisions (*ASold*) and mortality (*mort*) (Eq. 8.1). Since animals are seldom purchased in our database, we do not introduce the possibility of buying animals.

$$ANb_{a,t,p,\zeta} = f_{p-1}(.) = ANb_{a,t,p-1,\zeta} \times (1 - mort_{a,p-1}) - ASold_{a,t,p-1,\zeta} \tag{8.1}$$

At the beginning of each following year (in April), an animal may change from one class to another because of natural ageing process (the number of 1 year old heifers at the end of a year becomes the initial number of 2 year old heifers the following year or calf numbers depend on the cow numbers) and because of fattening and reproduction objectives *(FAT*). The model can choose for instance to convert part of the number of 2 year old heifers into fat heifers (Eq. 8.2) and the remaining part into primiparous cows (Eq. 8.3). They are differentiated from multiparous cows because they are still growing and have different needs. In the studied area, females calve for the first time at 3 years old and then once a year in winter. Multiparous cows do not undergo an ageing process in our model (they correspond to only one animal class); consequently, a minimum cull rate is introduced and set at 0.2. We assume in our model that calvings occur on 1st February since in our panel dataset, 70% of calvings have occurred between January and March (however, the model formulation enable to modify this date if needed). Restricting calving spread and monitoring these calvings indoors give farmers more controlled management. Cows usually suckle their calves for 8 months. In our model, the number of cows must be high enough to suckle young animals until weaning and no sale of calves is allowed before their fifth month which corresponds to early weaning. Number of calves born per reproductive female (0.96), sex ratio (0.5) and mortality rates (9% for calves, 1% for the other) correspond to average annual records on the ‘Charolais’ database. Mortality is assumed to be spread evenly over the year except for calves for which we observe higher mortality rates after birth.

$$ANb_{a\_stor,t,'p1',\zeta} = \sum_{a} \left[ trans\_Anb_{a,a\_stor,\zeta} \times f_{a,t-1,p-1,\zeta}(.) \times \left(1 - FAT_{a,t-1,\zeta}\right)\right] \tag{8.2}$$

$$ANb_{a\_fat,t,'p1',\zeta} = \sum_{a} \left[trans\_Anb_{a,a\_fat,\zeta} \times f_{a,t-1,'march',\zeta}(.) \times FAT_{a,t-1,\zeta}\right] \quad (8.3)$$

*Where "a_stor" and "a_fat" are animal subclasses corresponding resp. to stored and fattened animals; "trans_Anb" the between year transition matrix for head number dynamics.*

Animal live weight dynamics (*LW*) are expressed in the same way: optimisation of the initial live weight, intra-annual dynamics described by a motion function which depends on the average daily weight gain (*ADG*), and inter year dynamics defined by a transition matrix. Similarly to animal number dynamics, initial live weight are set once for all but then the live weight can be adjusted according to observed weather event. Live weights are allowed to vary from $\pm 5\%$ of the "theoretical" live weight (*tlw*) and the weight gain from $\pm 20\%$ of the "theoretical" gain. For mature cows, we set gain interval at $[-0.6; +0.4]$ kg per day. These ranges of variation give flexibility to the model to adjust animal live weights according to market or weather conditions. Compensatory growth can be simulated since animals can have the same total weight gain (and consequently the same final live weight) with different growth paths. These "theoretical" values provide bounds to ensure that, according to expert knowledge, reproduction performance and animal health cannot be threatened by excessive weight variations. Theoretical live weight and weight gain are calculated with a sub model which draws standard growth curves according to animal sex, age and production objective. These growth curves are based on equations exposed in Garcia and Agabriel (2008) for females (cows and 3 year old heifers) at fattening and for other animal classes on Gompertz functions defined in INRA (2007).

The ADG variable (Eq. 8.4) is a function of the daily net energy balance (NEB). Parameters of this function have been obtained from INRA (2007). NEB is the difference between on the one hand net energy intake (*NEI*) which depends on quantity of feed and milk ingested by each animal and on their energy content, and, on the other hand, net energy requirement that comprises net energy for lactation and pregnancy (*nep*) and net energy to maintain (*nem*) live weight constant. Reproduction needs depends on stage of pregnancy and lactation (INRA 2007). Maintenance requirements are set according to animal theoretical live weight, to animal activity and to stage of lactation. These requirements are increased by 20% when animals are grazing (INRA 2007) to account for higher activity as did Jouven et al. (Jouven et al. 2008). Maintenance needs increase if animals are fatter and decrease if they are thinner than theoretically. To take into account these variations while keeping a linear formulation (the function provided by INRA 2007 is a power function), we introduce a corrective term *dnem*.

$$\begin{aligned} ADG_{a,t,p,\zeta} = {} & g1_{a,p} + g2_{a} \\ & * \left[NEI_{a,t,p,\zeta} - nem_{a,p} - dnem_{a,p} \times (LW_{a,t,p,\zeta} - tlw_{a,p}) - nep_{a,p}\right] \end{aligned} \quad (8.4)$$

Diets are not only characterized by their energy content but also by their fill value which cannot exceed the intake capacity of the animal. This capacity corresponds to the amount of Cattle Fill Units[2] (CFU) an animal can eat when fed *ad libitum* (Jarrige et al. 1989).

### 2.2.2 Grassland Production and Feed Stock

In the studied area, most animals graze on grassland from April to November and are fed inside at trough in winter. For this study, we simplify the cropping systems of the previous version of this model (Mosnier et al. 2009): we consider only 100 ha of the grassland area ($X$). The bi-monthly average yields ($y$) are calculated thanks to a sub model of herbage growth developed by Jouven et al. (2006). Grassland production is divided into production that can be grazed by animals (*PRODpast*) or harvested to make hay (*VHarv*). The average area of grassland cut every 2 month is optimized (*XH*). Adjustments of these areas are possible to help facing hazards (*Aj_XH*). However, in the model, decisions to adjust production grassland are taken knowing exactly what would be the production for the next 2 months which is only an approximation of the reality. We limit then the bimonthly adjustment at more or less 20% of the total grassland area. Moreover, modifying the initial harvest planning is assumed to have some drawbacks since to be efficient grazing or haymaking need to be anticipated. A grassland with already a rather high level elevation of grass above ground level means for instance more wasting by the animals. If the initial area planned to be harvested is decreased (*neg_XH*), then the area that could be grazed is increased but with a penalty (*ajloss_gr*), (Eq. 8.5). Conversely, if the initial area planned to be harvested is increased (*pos_XH*), the area harvested increases but with a penalty (*ajloss_harv*). Harvested hay quantity is also decreased by 20% (*loss*) to account for losses during haymaking, harvest, transport etc. (Eq. 8.6). *Neg_XH* is a negative variation of *Aj_XH* whereas *Pos_XH* is a positive one (Eq. 8.7).

$$PRODpast_{t,p,c1,c2} = y_{t,p,c1,c2} \times \left[X.\left(1 - pos\_XH_{t,p,c1,c2} - neg\_XH_{t,p,c1,c2} \times ajloss\_gr\right) - XH_p\right] \tag{8.5}$$

$$VHarv_{'hay',t,p,c1,c2} = y_{t,p,c1,c2} \times \left[X.\left(pos\_XH_{t,p,c1,c2} \times ajloss\_harv + neg\_XH_{t,p,c1,c2}\right) + XH_p\right] \times loss \tag{8.6}$$

[2] 1 CFU is the "standard" voluntary dry matter intake of a reference herbage by a 400 kg-heifer, set to 95 g/kg metabolic LW (INRA 2007)

$$Aj\_XH_{t,p,c1,c2} = pos\_XH_{t,p,c1,c2} + neg\_XH_{t,p,c1,c2} \tag{8.7}$$

Two other products are considered: grain and straw (only used as litter). Feed products are associated with parameters of qualities: (1) fill value[3] set in CFU; (2) energy content expressed in accordance with the INRA feed evaluation system in net energy for lactation when animals are lean and net energy for meat when animals are fattened. Regarding grain and hay feed values are set according to INRA (2007). A basal value of 0.3 CFU/kg of dry matter is fixed for concentrate. Qualities of green forage depend on the average Organic Matter Digestibility of the different structural compartments (green and dead matters). They are calculated thanks to equations given in Jouven et al. (2008).

Evolutions of the available quantity of feed products are described by dynamic variables. Stocks of conserved produce (all except grazed grass) are defined as the balance between inputs – production harvested and bought and withdrawals – herd consumption and sale – plus the stock remaining from the previous period. Secondly, the quantity of standing grass available in one period corresponds to the remaining balance between previous biomass stock (cut by losses due to senescence and abscission), the grass produced not harvested and herd consumption. Delaying the use of grass production leads indeed to standing biomass losses because of senescence (deterioration related to ageing process) and abscission (shedding of dead matter) processes (Jouven et al. 2006).

### 2.3 Receipts and Costs

Beef margin is calculated as the difference between yearly receipts (animal and hay sales plus Common Agricultural Policy payments) and costs associated to the beef enterprise. Animal product sales take into account the number of animals sold, their live weight and their price. These prices are defined per month, which enables us to introduce price modulation according to theoretical live weight (price per kg usually decreases with live weight for stored animals and increases for finished ones). It is important to introduce CAP payments, as production decisions in suckler cow farms are highly influenced by them (Veysset et al. 2005). The CAP premium specification is flexible enough to take into account the different kinds of direct payments belonging to the first pillar (production support) which have been effective between 1998 and 2010. For the year 2010, these payments encompass grassland area payments, suckler cows payments and Single Farm Payment (SFP). Suckler cow payments are upper bounded by historical reference and heifers

[3] Fill value of feed products is calculated as the ratio between the voluntary dry matter intake of the reference herbage by a 400 kg-heifer, set to 95 g/kg LW0.75 (Jarrige et al. 1989), and the voluntary dry matter intake of the forage considered.

are eligible to this premium too. Not all animals eligible can then receive this suckler cow premium. Here, the reference for suckler cow premium is set at 80. Under the SFP scheme farms are allotted payment entitlements, which can be activated by matching them with the corresponding number of eligible hectares. In our model, farm size is fixed and all hectares eligible. SFP is therefore considered as a constant. Moreover under this scheme, direct payments are reduced in proportion to the modulation rate which is 10% in 2010. The recent French premium attached to grassland is also introduced.

Variable costs can be divided into grassland crop production and animal production costs. Crop production costs include fixed input costs for grassland (50€/ha), haymaking costs (90€/ha). Animal production costs comprise value of purchased feeds and litter, diverse costs such as veterinary or feed complementation such as vitamins or minerals (78€/LU) and labour costs. The labour time required corresponds to the estimated daily time spend to feed animals and improve the litter. It varies greatly among farms according to farm equipment, barn configuration or farmer efficiency and attention to details (Pichereau et al. 2004). However, the amount of 16 h / LU/year appears to be the average time (Réseau d'Elevage Viande Bovine 2006). The cost per working hour is fixed at 12€.

Fixed costs (*FIXCOST*) linked to animal housing are added too. We assume that these fixed costs are proportional to the housing capacity (*BARN_LU*) of the barn and equals to 65€/LU (*fixcost_lu*). Since, to a certain extent, it is possible for farmers to let some animals outside during winter time, the barn capacity is not binding. However, we suppose that the cost for farmers to let one animal outside is similar to the one of providing it a place inside. This possibility is somehow already taken into account in the 65€/LU since the annual barn costs are divided by the total herd size, irrespectively of whether they stay inside or outside during winter. Moreover, if marginal housing costs decrease when the barn capacity is exceeded, it would lead to a permanent barn capacity excess. We allow only a 10% increase of LU compared to barn capacity (*pos_LU*) since young animals are for instance more sensitive to cold weather. Conversely, if farmers decide to have a herd size below barn capacity (*neg_LU*), we suppose that since the investment has already been done, fixed housing costs do not decrease (Eqs. 8.8 and 8.9).

$$FIXCOST_{t,\varsigma} = fix\cos t\_lu \times \left[BARN\_LU + 1.01 \times pos\_LU_{t,\varsigma}\right] \tag{8.8}$$

$$LU_{t,\delta} = BARN\_LU + pos\_LU_{t,\varsigma} + neg\_LU_{t,\zeta} \tag{8.9}$$

## 2.4 The Optimisation Program

In accordance with classic economic theories, optimal decisions are those that maximise the objective function *Z* which is equal to the expected (*E*) utility (*U*) of profit ($\Pi$) over a finite planning horizon. The utility function introduces farmer's preferences toward the distribution of profit. A risk averse farmer would for instance attribute a greater utility to a distribution characterised by a lower variability and could consequently choose a production plan that do not provide the highest expected profit. The utility function can be either modelled by a functional form such as the power function that assumes risk aversion decreases when expected wealth increases (Hardaker et al. 2004). It can also be summarized by its central moments. Although the "mean-variance" approach (Eq. 8.10) suppose that farmers has the same aversion for positive deviation from average profit as negative deviation, it appeared to us much more efficient to simulate the trade-off between expected profit and risks. The higher the Arrow-Pratt absolute risk aversion coefficient ($r_a$) is in the following utility function, the more risk reducing the production plan would be.

$$\mathrm{Max}\, Z = EU(\Pi_{t,\zeta}) = E(\Pi_{t,\zeta}) + 2r_a.E\left[\Pi_{t,\zeta} - E(\Pi_{t,\zeta})\right]^2 \qquad (8.10)$$

Usually, the value of stock variation is optimised too. However, since the objective of this model is not to simulate long term adaptations but short term variations, we constraint the stock variation to be null. This avoid dealing with problem of valuing stock which leads to stock depletion when the closing value is lower than the selling one and conversely to stock accumulation if the closing value is greater.

### 2.5 *Revisions of the Production Plan According to Observed Weather Events: The Recursive Framework*

To cover the entire period of the simulation (18 years) and to update information about current weather conditions, we follow Iglesias et al. (2003) and Barbier and Bergeron (1999) in using a recursive sequence of dynamic optimisations (Fig. 8.2). Model predictions for a given year are therefore optimal regarding the 3 year planning horizon but not necessary optimal regarding the entire period of simulation: if the 'modelled farmer' had anticipated that such a succession of shocks would have occurred, they would perhaps have opted for different production choices.

Not all the decisions can be revised. The initial herd size and animal live weight as well as the barn capacity are fixed once for all during the first optimization i.e. before the simulation of the 18 year sequence of weather events. Decisions depending on weather conditions, such as animal feeding and animal sales, feed trade or haymaking area, can be adjusted. Recursions are made at a bimonthly step in order to introduce real grassland production not more than 2 months in advance. In our model, the year starts in April. For the first optimisation of the simulated year, real grassland yields are known until May, for the second one yields are

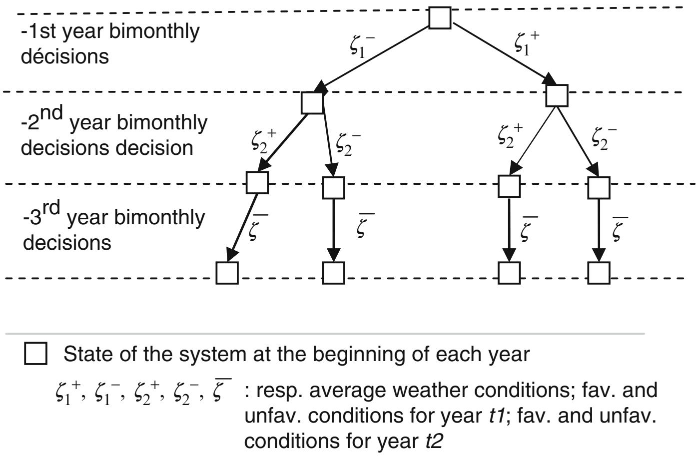


**Fig. 8.2** Anticipation of two embedded weather risk over the 3 year planning horizon

known until July etc. Once production is known, decisions are not differentiate anymore according to the weather conditions (c1+ and c1−) and they become definitive. Continuity within a year between the different optimizations is achieved by fixing the decisions that have been taken during the previous optimization. When 1 year has been covered, the whole planning horizon is shifted by 1 year. Starting values for dynamic variables are then set to their value at the beginning of the second year of the previous optimisation. This process is reiterated until the whole simulation period is covered. For each optimization, the constraint of null variation of stocks is resolved in reference to initial values set during the first optimization. The system can then benefit from hay stock accumulated previously.

# 3 Model Application

## 3.1 *Scenario Description*

The objective of this paper is to better understand how on-farm risk reducing strategies can modify the production system and profit distribution of French suckler cow enterprises. The willingness of farmers to reduce profit variability due to weather risks depends on their aversion for risk. We choose the value of risk aversion $r_a$ in order that the anticipated variability of profit would be reduced by half. We compare then technical and economic results of beef enterprise under an absolute risk aversion coefficient 1/null (i.e. $r_a = 0$) corresponding to the absence of risk aversion and 2/$r_{a=}$0.25. In addition, we test the impact of hay market price

**Table 8.1** Scenarios characteristics

| | P1 | P1A | P2 | P2A |
|---|---|---|---|---|
| Market price for hay in €/t | 90 | 90 | 120 | 120 |
| risk aversion ($r_a$) | 0 | 0.25 | 0 | 0.25 |

**Table 8.2** Characteristics of the initial production plan according to scenarios

| | P1 | P1A | P2 | P2A |
|---|---|---|---|---|
| Average profit (k€/year) | 32.3 | 31.7 | 31.6 | 31.0 |
| *s.d. of profit (k€/year)* | *3.0* | *1.2* | *3.5* | *0.9* |
| *Variation of profit* | *9%* | *4%* | *11%* | *3%* |
| Average herd size (in LU) | 128 | 116 | 118 | 104 |
| *s.d. of herd size (in LU/year)* | *0.4* | *0.3* | *0.6* | *0.1* |
| *s.d. of LW of animal (in %)* | *0.3* | *0.5* | *0.6* | *0.3* |
| Initial stock of hay (in t) | 0 | 12 | 69 | 59 |
| Initial stock of concentrated feed (in t) | 0 | 136 | 0 | 121 |
| Grassland area harvested end of may (ha) | 43 | 56 | 61 | 66 |
| *s.d. of concentrate feed (in kg/LU)* | 407 | 176 | 79 | 2 |
| *s.d. of purchased hay (in t/year)* | *46* | *28* | *18* | *0* |
| *s.d. of* area harvested end of may (ha) | 17 | 13 | 11 | 8 |

since the availability of off-farm feedstuff can supplement on-farm feed production and consequently impact on the risk management plan. The scenarios are summarized in Table 8.1.

Since farmers anticipation are only a partial representation of what could happen, we simulate a sequence of 18 years corresponding to the grassland yield observe over the period 1990–2007 in the Nièvre department, located in the Charolais area (the anticipation we have assumed for farmers are based on this sequence). Weather events are associated to a deviation of grassland yield from average yield.

## 3.2 *Results*

### 3.2.1 The Initial Production Plan

For a market price of hay reaching 90€/t and no risk aversion (scenario P1) expected profit reaches 32.3 k€/year with a variability of 9% (Table 8.2). The optimal herd size is 128 LU/year is characterised by 103 calvings, no fattening of young animals (they are sold at 10 months) and an objective of 9% of the cows sold fattened (i.e. ready to be slaughtered). Herd management is planned to be adjusted to face weather variations (average yield plus or minus one standard deviation for the first 2 years of the planning horizon). Although adjustments are in rather small proportions (0.3% of average value), additional simulations indicate that they are significant. The possibility to adjust of animal live weight limit for instance the purchase of hay and concentrate feed and increases average profit by around 200€. This adjustment concerns mainly summer cows live weight with a coefficient of variation of 3%. Main adjustments concern however the grassland and feed management. Adjustments of the quantity of feed bought represents around 100% of their average value, varying between 0 t of hay after a good first year to 108 t following insufficient yield. Concentrate feed consumption also varies a lot according to weather conditions with a standard deviation of 0.4 t/LU/year. The area of grassland is also planned to be adjusted with a coefficient of variation around 40%.

The risk reducing strategy (scenario P1A) simulated for a hay market price of 90€/t, induces 600€ of foregone expected profit but reduces profit variability by more than half. To decrease exposure to weather risk, the option simulated consists in lowering the long term stocking rate (and consequently the barn capacity) by 9%, the kinds of animal produced are however keep unchanged. The grassland area for haymaking is increased by 16% (including the second haymaking period in summer) and stocks of hay of concentrate feed are introduced. Adjustments of grassland and feed management subsequent to weather events are then much smaller: the standard deviation of concentrate feed per LU and purchased hay are divided by two. There is no significant changes of planned adjustments for animal production, except for animal live weights who is slightly more variables.

When market price for hay is 30% higher (scenario P2), the herd size shrinks by 8% compared to scenario P1 and is closed to the herd size simulated in scenario P1A. The grassland area for haymaking expands by 30% and an important initial hay stock is introduced to secure the system. Adjustments by the quantity of concentrate feed purchased decrease a lot and hay is only purchased for the case where two bad years occur in a row. The higher portion of area harvested, the higher initial stock of hay and, to a lesser extent, the more important adjustments of herd size, could explain the lower quantity of hay purchased compared to scenario P1A.

The risk reducing strategy (P2A), enables to decrease variability by 3 in reference to scenario *P2* for a foregone expected profit of 600€. Once more, lowering stocking rate and enlarging the area for haymaking help decreasing profit variability. Almost no hay nor concentrate feed is planned to be purchased and adjustment of herd management is very limited.

### 3.2.2 Impacts of a Sequence of Weather Events on Economic Results

The simulated sequence of weather event includes the year 2003 characterised by a very important decrease of grassland yield (almost by half) and the year 2004 when yield reaches 140% of their average value. Such extreme events have not been anticipated to optimize the production plan.

Over the sequence, regardless of the scenarios, we observe (Fig. 8.3a) a rather low variability of receipts from animal sales except for the year 2003 during which fewer cows have been fattened. Receipts for the scenario P2 have been impacted for several years following the 2003 drought. A higher number of cows have indeed

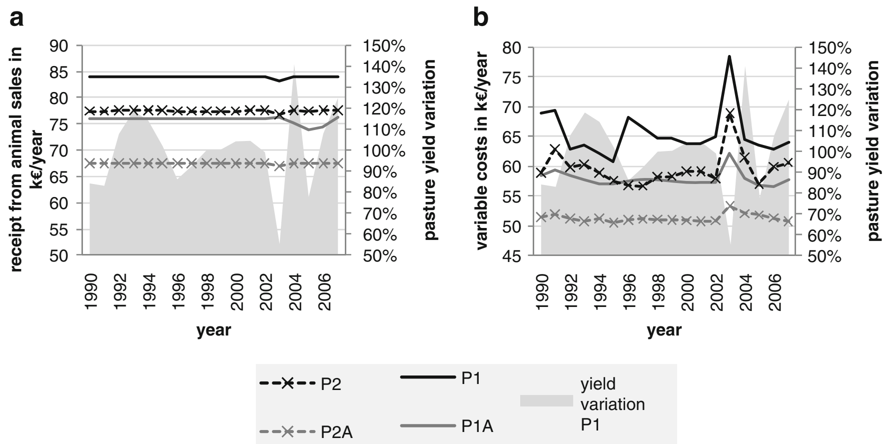


**Fig. 8.3** Evolution of (**a**) receipts from animal sales and (**b**) variable costs over the simulated grassland yield sequence from 1990 to 2007 according to hay price level (P1: 90€/t and P2: 120€/t) and to risk reducing strategies (no risk aversion or A: $r_a = 0.25$)

**Table 8.3** Profit distribution according to the scenario over the simulated period of grassland yield 1990–2007

| | Average profit (k€) | Standard deviation (k€) | Coefficient of variation (%) |
|---|---|---|---|
| P1 | 31.4 | 4.1 | 13 |
| P1A | 31.6 | 1.2 | 4 |
| P2 | 31.5 | 2.9 | 9 |
| P2A | 30.9 | 0.8 | 2 |

been sold, reducing the number of calves and the sales for the subsequent years. The variable costs fluctuate much more in general (Fig. 8.3b). However, the risk reducing scenarios help smoothing these costs. The year 2003 swells variable costs a lot because of induced feed purchase but the non risk reducing ones experience increases of greater amplitude.

Regarding profit distribution over the 18 year sequence (Table 8.3), the risk reducing strategy under hay market price set at 90€/t (*P1A*) performs better than the risk neutral one (P1) for both criteria: average profit and variability of profit. Although farmers expectations for grassland yield have been based on this 1990–2007 sequence, we have only considered average profit plus or minus one standard deviation. Profit loss caused by the extreme 2003 year has not been compensated by symmetric gain in very favourable years such as 2004. In this case the more cautious strategy has been more adapted to the uncertain weather. Under higher hay market price, the scenarios P2 conserves its advantage in terms of average profit while the profit distribution in P2A is lower. The initial production plan P2 already limits indeed feed purchases because of the higher price of hay.

## 4 Discussion

In this study, we have simulated on farm risk management according to risk reducing objective and economic conditions for feed market substitute. Both risk adjustments and production decisions intended at limiting risk exposure have been simulated.

In our simulations, temporary adjustments of animal live weights are found to bring significant outcomes to help farmers to face grassland yield variability. This supports Hoch et al. (2003) and Blanc et al. (2006) who underline that the plasticity of animals such as beef cows that have a low level of production and a small response of production to underfeeding could help facing feed shortage. However, although obtained live weights variations are substantial, they are limited and concern above all suckler cows. In the case of very serious crop production losses, some forced sales were simulated as well, but in limited proportion. Most of the time it consists in selling part of the cows lean instead of fattened. However, in one scenario a reduction of the cow numbers had consequences over several years. The most important sources of adjustments simulated target the purchase of hay and concentrates feed products and the area of grassland harvested. These results are corroborated by the empirical analysis of Veysset et al. (2007) and Mosnier et al. (2010) on a panel of farmers in the same Charolais area. They are also in accordance with Jouven and Baumont (2008) and Romera et al. (2005) who demonstrate the advantages of a flexible haymaking plan and Sullivan et al. (1981), Diaz-Solis et al. (2006), Gillard and Monnypenny (1990), as well Kobayashi et al. 2007 who provide evidence that supplementary feeding, when forage resource is scarce, improves farm profit.

Although choosing the appropriate combination of production adjustments to face weather variations improves expected farm profit for a given production system, limiting their amplitude helps decreasing profit variability. This is also what Mosnier et al. (2010) observed: farms characterised by the highest variability of feed purchases and haymaking were also those who have experienced the highest variability of production costs. This is in accordance too with conclusion of Veysset et al. (2007) who analyzed consequences of the 2003 drought on French suckler cow farms thanks to a panel data. They indeed noticed that the most self-sufficient farms were more profitable that the less self-sufficient ones. All simulated long term decisions associated to risk reducing strategies encompass a reduction of long term stocking rate. According to Lemaire et al. (2006b), the production systems characterized by the highest stocking rate would also be the most vulnerable to weather risk. Mosnier et al. (2010) also notice that low stocking rates of animals were associated to smaller adjustments and Rawlins and Bernardo (1991) simulate that risk aversion under weather and price risks induce a decrease of the stocking rate in Oklahoma.

Other important strategic decisions target feed stocks: initial stocks of concentrate feed and hay are expanded which could correspond to higher security stock, and, a larger area of grassland is allocated for haymaking in order to replenish security stocks. The importance of a strategy to constitute feed stocks in order to come through forage shortage has been underlined by Lemaire et al. (2006a). However, to our knowledge no empirical studies or simulation studies have dealt with this issue.

Conditions of the economic environment can also modify the long term strategy of farmers. We have analysed in this study the impact of price of purchased hay on the market but results could be extended to the question of availability (and price) of substitute to on-farm feed production. The more expensive and scarce are feed substitutes on the market, the more incentives farmers have to be self reliant for feed and to seek on-farm solutions to reduce their exposure to weather risk, included in the case of risk neutral farmer. Conversely, if commercial feed (or hay) become accessible at fair costs, only non insured risk averse farmers would have interest in keeping their stocking rate low to limit adjustments of feed purchases.

Risk reducing strategies induce indeed trade-off between expected profit and variability of profit. In the simulated scenarios farmers have to forego 600€ of profit to decrease the standard deviation of profit by 1.8 k€ or 2.6 k€ according to the scenarios. Gillard and Monnypenny (1990) also simulate lower average profit and lower variability (in absolute terms) for lower stocking rate in Australian rangelands. In Lien and Hardaker (2001) optimized risk reducing strategies do not lead to different production choice. However, we advocate that this could be explained by the high level of constraints of the model and the relative low risk aversion introduced by the power functional form of the utility. In the case of the whole sequence of yield variation observed between 1990 and 2006, the risk reducing scenario under hay price set at 90€/t performs better for both average and variation. In the empirical study of Mosnier et al. (2010), we observe similar fact: suckler cow farmers with higher stocking rate has experienced over this period a higher variability without significantly different average profit. This raises the problem of farmers' anticipation: how do and should farmers consider weather distribution when taking their decisions.

## 5 Conclusion

We presented in this paper an original bioeconomic model that takes into account both risk anticipation and risk adjustments and that details biotechnical relationships between the different component of the beef cattle production system and their dynamics. On-farm risk management strategies are endogeneized under weather uncertainty and to test them on real observed weather sequences. Our bioeconomic models help to better understand observed behavior of farmers and to simulate impacts of weather event on the evolution of technical and economic variables.

Results of our simulations emphasized that production adjustments, particularly the adjustments of area of grassland harvested and the possibility to purchase substitutes to on-farm forage production improve farmers profit under weather variability. However, the highest the price of feed substitute is, the most incentives farmers have to be self sufficient under weather uncertainty. Moreover, adjusting feed purchase and more generally the production system induce a greater variability of variable costs and profit. Simulated risk reducing strategies consist in limiting them. They are characterized by a lower average stocking, an extended area of grassland harvested and by the introduction of spare feed stock to secure the system.

Better private insurance and increased availability of feed substitute on the market could then result in higher pressure on grassland, which could conflict with environmental goals. This emphasize as well that some on farm-solution exist to reduce production risk. However, when some extreme event is simulated, even farmers with risk reducing strategies suffer from important profit loss. We have supposed indeed that farmers have not made their production plan in function of all extreme conditions that could occur. If this assumption is true, extreme events would require special treatments by policy makers or insurers to support farmers.

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# Chapter 9
# Using a Bio-Economic Model to Assess the Cost-Effectiveness of Measures Against Nitrogen Pollution

**I. Mouratiadou, D. Tarsitano, C. Topp, D. Moran, and G. Russell**

## 1 Introduction

Water resource management is an inherently complex, multi-scale and multi-disciplinary process involving many interdependent components. As each of these components is the focus of several socio-economic and scientific disciplines, an approach that crosses disciplinary boundaries is needed to provide constructive input to policy making. Scientific approaches have been developed that study the complex relationships between the economic and ecological systems and that aim to provide knowledge for sustainable management of water resources. Such approaches are interdisciplinary in nature, where interdisciplinarity refers to the cooperation of many scientific disciplines, in order to analyse the relationship between the economic and natural system (Baumgärtner et al. 2008). Bio-economic modelling is one of those.

The importance of interdisciplinary research, particularly when informing policy in the management of socio-ecological systems, is recognised by both social and environmental scientists (e.g. Mascia et al. 2003; Lawton 2007). However,

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successful interdisciplinary research requires that disciplines gain a common understanding of the problem at hand, identify the scales of relevant system subcomponents, the underlying phenomena or processes, and the important variables involved (Dollar et al. 2007). As a consequence, new research questions, new approaches to problems, new theories, and new generalisations are produced (Pickett et al. 1999). These can be seen as different forms of knowledge or constructions of reality that have resulted from the interplay between human intellect and empirical experience (Baumgärtner et al. 2008).

This paper adopts an interdisciplinary approach, drawing from the premise that economic systems form sub-components of the broader natural system so that an analysis of either of the two types of system cannot be achieved in isolation from the other, if the aim is to gain a better understanding of the economy-environment interactions and propose answers to problems with real life applicability. The complex economy-environment interactions that occur at the level of agricultural systems, using the example of nitrate pollution which is the departing point of this research, can be briefly described as follows: Farmers adjust their production decisions (e.g. tillage, fertilisation, sowing) in order to optimally combine inputs based on natural capital (e.g. soil, solar energy, rainfall) and inputs from human-made capital (e.g. fertilisers, pesticides, machinery). This process yields desired outputs, namely agricultural products, and undesired emissions to the environment (van der Werf and Petit 2002). It is these interrelated natural and economic processes that give rise to the need for an interdisciplinary approach informing policy design and that bio-economic modelling approaches aim to encompass.

A challenge that often appears in the analysis of integrated environmental-economic systems is how to combine heterogeneous information and systems boundaries in a consistent manner. Integration of scales is seen as a major research challenge by many authors (Bouman et al. 1999; Vatn et al. 2006), and is subject to a number of considerations. Firstly, the wide arrays of agronomic, environmental and economic processes, between which the causal relationships have to be established, operate at different spatial and temporal scales. Crop production and emission losses take place at the field level on a daily basis. Farmers make their main cropping decisions at the farm level on a seasonal or yearly basis, while some management decisions, such as fertilisation, are made on a daily or weekly basis. Pollutant transport into water bodies operates at the catchment level on a daily basis. Secondly, while the integration of bio-physical and economic models should ideally occur at a highly disaggregated level so as to capture bio-physical and economic behaviour heterogeneity, policy making is interested in larger units of analysis, as for example the river basin, the regional or the national level, and in the long-term effects of policies. These large scale and long term effects are effectively the result of the accumulation in time and aggregation in space of the effects that occur at smaller units of analysis. As Rossing et al. (2007) state, policy goals implicitly or explicitly express pertinent temporal and special scales and organisation levels, and thus affect the definition of the systems to be assessed. Thirdly, data and statistics are usually 'mono-disciplinary' in terms of both their content and boundaries. All of the above raise a number of questions such as what is the most appropriate level of integration of the ecological and economic relationships; how can these relationships be then upscaled or aggregated to greater levels so as to provide meaningful information to policy makers; how can limitations of existing data be overcome in order to achieve integration of scales?

These issues are explored in the present paper through an application of a bio-economic model to the Lunan Water catchment in Scotland to assess the relative cost-effectiveness of measures against agricultural nitrate pollution. The aims of the paper are to explore the challenges related to bio-economic modelling applications, present the methodology and results, and evaluate the overall appropriateness of the approach for integrated policy impact assessment.

# 2 Methodology

## *2.1 Integration Across Systems and Scales*

The bio-economic model used in this work was based on FSSIM-MP (Louhichi et al. 2010a, b), which was developed under the EU FP6 Project SEAMLESS (van Ittersum et al. 2008). The adapted model version includes the farm type dimension, so as to achieve a formal aggregation procedure. Integration in the model is achieved through the incorporation of information on yields and environmental indicators associated with the defined agricultural activities. This information was generated from bio-physical modelling simulations, with the Coupled Heat and Mass Transfer Model for Soil-Plant-Atmosphere Systems (COUP) (Jansson and Karlberg 2004).

Specification of the production activities is one of the most important steps in the conceptual and practical integration of the bio-physical and the bio-economic components. For the outputs of the bio-physical model to be successfully incorporated into the bio-economic model the agricultural activities need to be consistent between the two models. Therefore, the choice of the attributes to be used to characterise an agricultural activity needs to be based upon, firstly, the factors that explain most of the variation in outputs, and secondly the characteristics of the activity that can be effectively simulated by both models. The main characteristics of a cropping agricultural activity that influence yields and nitrate losses are (i) the crops grown and the sequence of these crops in crop rotations, (ii) the production techniques used, particularly N fertiliser levels, and (iii) the soil types on which each of the activities take place. All these factors can be incorporated into the definition of agricultural activities of both FSSIM-MP and COUP. A crop production activity has been defined in this work as a rotation, consisting of a sequence of crops, with a certain fertiliser level and cultivated on a specific soil type.

The methodology used to choose the appropriate spatial and temporal resolution and extent for each process is analogous to what Rastetter et al. (1992) called *partitioning* for the aggregation of fine scale ecological knowledge into coarser-scale attributes. Partitioning is a way of reducing aggregation errors by reducing the variability among the components to be aggregated through their classification into relatively homogenous sub-aggregates. As the number of partitions increases the level of aggregation errors will decrease (*ibid*).

The first level of resolution in the spatial hierarchy is the field level. Fields are partitioned into homogenous groups according to their soil properties. At this level, crop growth and nitrate leaching are simulated by COUP for a range of rotations and management practices on a daily basis for a series of years. The key outputs extracted from the simulations are average seasonal yields per crop in a rotation and average nitrate leaching per rotation, over the simulation period.

The second level in the hierarchy is the farm. Farms are classified into types according to their production orientation and size. Aggregation from fields to farms is done in two ways: (i) a constraint in the economic model specifies the number of fields of each soil class available to each of the farm types; (ii) the rotation and management on each of the available fields are selected through the optimisation procedure of the economic model. The information generated at the field level enters the economic model in the form of yield and leaching coefficients. Each field type characterised by soil, rotation and management is associated with a coefficient of average annual yield per crop in the rotation and a coefficient of average annual leaching per rotation.

The natural upper spatial level of the analysis is the catchment. However, because agricultural statistics are collected on a parish basis, the 12 agricultural parishes within which the catchment is situated have been used as the upper spatial level of the analysis. Aggregation of farms at this level was achieved through a formal aggregation procedure in the economic model that uses an objective function where the individual farm types are multiplied by the number of farms per farm type. At this level, farmers' decision making for each of the individual farm types is simulated for a number of scenarios in a comparative static framework. This generates information for each of the modelled farm types and scenarios on (i) socio-economic indicators, such as farmer utility, income, premiums, gross production, costs, labour use, (ii) technical information including land use and choice of rotations and management, and (iii) environmental indicators such as average per hectare input use and nitrate leaching at the farm level. Information such as utility, income and land use is also provided at the aggregate level.

## 2.2 *Overview and Specification of the Bio-Economic Model*

FSSIM-MP assumes that farmers make their decisions in order to maximise expected income minus some measure of its variability, caused by yield variations due to weather, and price variations due to market conditions. This risk specification is taken into account through the Mean-Standard Deviation method (Hazell and

Norton 1986). The model non-linear objective function represents the expected income and risk aversion towards price and yield variations for a number of farms:

$$maxU = \sum_{f} n_f (Z_f - \varphi_f \sigma_f) \tag{9.1}$$

where $f$ indexes farm types, $U$ is expected utility, $n$ is the number of farms per farm type, $Z$ is expected income, $\varphi$ is a scalar for the risk aversion coefficient, and $\sigma$ is the standard deviation of income defined under price variability and yield variability.

Expected income is defined as total revenue, consisting of sales from agricultural products and subsidy compensation payments minus total variable costs from crop production. Total variable costs include accounted linear costs for fertilisers, crop protection, seeds, etc., and for hired labour, and unaccounted costs due to management and machinery capacity reflected by the quadratic term of the cost function. The non-linear income function is:

$$Z_f = \sum_{j} p_j q_{f,j} + \sum_{i,t} \left(s_{i,t} - c_{i,t}\right) \frac{X_{f,i}}{\eta_i} + \sum_{i,t} \frac{\left(d_{f,i,t} + \frac{\Psi_{f,i,t} X_{f,i}}{2}\right) X_{f,i}}{\eta_i} - \varpi L_f \tag{9.2}$$

where $i$ indexes agricultural activities, $j$ indexes crop products, $t$ indexes the year in a rotation, $p$ is a vector of average product prices, $q$ is a vector of sold products, $s$ is a vector of subsidies, $c$ is a vector of variable costs, $X$ is a vector of the levels of agricultural activities, $\eta$ is a vector of the number of years of a rotation within each agricultural activity, $d$ is a vector of linear terms used to calibrate the model, $\psi$ is a symmetric, positive (semi-) definite matrix of quadratic terms used to calibrate the model, $\varpi$ is a scalar for labour cost, and $L$ is the number of hours of hired labour.

The standard deviation of income is given by:

$$\sigma_f = \sqrt{\frac{\sum_k \left(Z_f - Z_{f,k}\right)^2}{N}} \tag{9.3}$$

where $k$ indexes the states of nature, and $N$ is the number of states of nature.

The expected income over states of nature is calculated using the same equation used to calculate expected income but with the average prices and yields replaced by the prices and yields for each state of nature. These are independent, normally distributed random numbers, estimated using a normal distribution function based on the average and the standard deviation of price and yield. Price and yield variations are assumed to be independent.

The risk aversion coefficient can be estimated manually or automatically. In the first case, the user assigns a value ranging from 0 to 1.65 to the coefficient (see Hazell and Norton 1986). The value of the coefficient rises with the farmer's risk aversion. Alternatively, the model automatically assigns a value between 0 and 1.65 to the coefficient, as described later.

FSSIM-MP is an activity-based model with primal representation of the technologies employed. Specifically, the production processes are represented by discrete production activities defined as vectors of technical/environmental coefficients which describe the production inputs, the agricultural outputs (desirable products), and their environmental effects (undesirable products). The definition of the agricultural activities is multi-dimensional, allowing their specification as discrete and independent options, whether they refer to different crops, to different technologies for the same activity, or to variations of the same technology. Crop agricultural activities are defined as a combination of a rotation, soil, management technique, and production orientation.

The principal socio-economic and technical model constraints are arable land per soil type, irrigable land per soil type, labour and water constraints. Rotational constraints are implicitly included in the model through the definition of agricultural activities as rotations rather than crops. Water and irrigable land constraints were inappropriate for this model application. The model set of constraints is shown below:

$$A_i X_{f,i} \leq B_f \qquad \forall f \tag{9.4}$$

$$C_i X_{f,i} \leq D_f + L_f \qquad \forall f \tag{9.5}$$

$$X_{f,i} \geq 0 \qquad \forall f \tag{9.6}$$

where $A$ is a matrix of technical coefficients for arable land per soil type, irrigable land per soil type or water, $X$ is a vector of agricultural activity levels, and $B$ is a vector of available resource endowments for arable land per soil type, irrigable land per soil type and water, $C$ is a matrix of technical coefficients for labour, and $D$ is a vector of available resource endowments for labour.

FSSIM-MP is calibrated in two stages. First the model automatically assigns a value to the risk aversion coefficient, choosing the value which gives the best fit between the model's predicted crop allocation and the observed values in the base year period, after a number of parametric simulations. This fit is assessed by the Percent Absolute Deviation (PAD). The closer the PAD value is to zero, the better the results of the calibration are.

$$PAD_f = \frac{\sum_{i=1}^{n} \left| \hat{X}_{f,i} - X_{f,i} \right|}{\sum_{i=1}^{n} \hat{X}_{f,i}} 100 \tag{9.7}$$

where $\hat{X}_i$ is the observed activity level, and $X_i$ is the simulated activity level.

Once the risk aversion coefficient has been assigned, the model is partly calibrated. For exact model calibration, the approach described in Kanellopoulos et al. (2010) has been used. This approach is a Positive Mathematical Programming (Howitt 1995) variant that calibrates the model by (i) raising the value of land to the weighted average gross margin of the observed activity levels,

(ii) using upper and lower bound calibration constraints for activities with higher and lower gross margins compared to the average gross margin respectively, and (iii) using information related to the supply elasticity of different activities along with the dual values of the calibration constraints to determine the weights of the linear and non-linear parts of the quadratic cost functions. The information on the supply elasticity of agricultural activities can be either drawn from econometric studies or estimated by using an ex-post analysis and choosing the value that gives the best forecast (Kanellopoulos et al. 2010). A default value is currently used.

### *2.3 Overview and Specification of the Bio-Physical Model*

Four models were considered for this application: CropSyst (Stöckle et al. 2003), NDICEA (Van der Burgt et al. 2006), APES (Donatelli et al. 2009) and COUP (Jansson and Karlberg 2004). Of these, only COUP was found to be able to adequately simulate yields and N dynamics in the conditions of high soil organic matter, low temperatures, and relatively high rainfall of the catchment.

The COUP is a dynamic and deterministic model of plant and soil processes. It simulates soil water and heat processes, and plant growth processes on a daily time step. The SOIL (Jansson 1996) and SOILN models (Eckersten et al. 1996), which are integral parts of the COUP model (Jansson and Karlberg 2004) have been previously used and validated for Scottish conditions (McGechan et al. 1997; Wu et al. 1998). McGechan et al. (1997) explored the suitability of SOIL for studying the processes of water transport in soil. Their simulations showed sufficient agreement with measured data to permit the use of the model for the study of soil water dynamics and the transport of water-borne pollutants through the soil. Wu et al. (1998) showed that simulated yields agreed with measured values for both cereal and grass crops, and that there were similar trends in nitrate leaching between simulations and site experiments. They concluded that SOILN can make realistic predictions about the effects of varying crop, soil and fertiliser management practices.

COUP has been used to simulate forestry as well as agricultural systems (e.g. Norman et al. 2008; Conrad and Fohrer 2009). In COUP, the plant is described by four C pools: leaves, stem, roots and grains. The C required for plant development is calculated as function of the global radiation absorbed by the canopy, with temperature, water conditions and N availability being considered as limiting factors. The plant demand for N is a function of the plant C:N ratio. N enters into the soil in the form of manure application, fertiliser and atmospheric deposition, which are external inputs. In addition, a smaller fraction of the N input is provided by the vegetation litter, which contributes to the main C input into the system. Organic C and N are added to the soil organic pools, faeces and litter, while mineral N goes into the ammonium and N mineral pools. The organic pools are characterised by a fast decomposition rate, which determines the flux of C and N

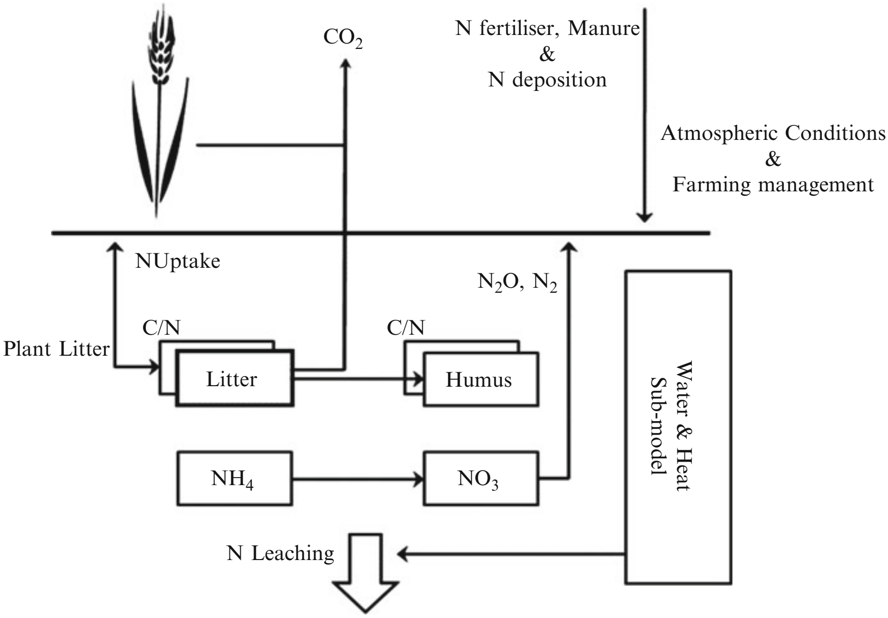


**Fig. 9.1** Block diagram of the COUP model

into a third organic pool (humus), characterised by a slow decomposition rate. Part of the C present in this pool will be lost due to soil respiration. The N cycle is described in terms of immobilisation/mineralisation between the organic and mineral pools, nitrification, which determines the flux between the ammonium and N pool, denitrification where N is lost into the atmosphere, and finally N leaching. These key model processes are depicted in Fig. 9.1.

Soil water dynamics is a crucial part of the overall system as several of the N processes are strongly dependent on water content and fluxes. Denitrification is particularly dependent on the oxygen present in the soil layer. Therefore, the higher the water content in the soil layer, the faster the process of denitrification taking place. The soil profile is divided into layers, where water and heat fluxes are estimated from soil characteristics, such as the water retention curve, and the hydraulic and thermal conductivities.

The crop model was manually tuned using as guidelines values reported in the literature (e.g. Eckersten and Jansson 1991; Kätterer et al. 1997; Nylinder, pers. comm., 20/11/2010). In addition, expected crops yields for Scottish conditions reported in the Farm Management Handbook (FMH) (Chadwick various years) have been used as target values for the parameterisation process. COUP has been previously parameterised for a representative Scottish soil (M. McGechan, pers. comm., 20/06/2010). This sub-model parameterisation has been used in this study, as it is similar to the soil scenarios under investigation, described in the following section.

## *2.4 System and Data Specification*

### 2.4.1 Case Study Area

The case study area used in this study is the Lunan Water catchment located in the Angus region on the East Coast of Scotland. The main reasons for the selection of this catchment are that it is representative of intensive arable production in Scotland (Vinten et al. 2008) and is one of two priority catchments monitored under the *Monitored Priority Catchment project*, established in 2005 as a partnership approach between SEPA, the Macaulay Institute and the Scottish Agricultural College (MPCPa undated). The catchment is at risk of not meeting the environmental objectives of the Water Framework Directive (SEPA 2007), and because of its vulnerable status data availability is better than for other catchments.

The catchment is predominantly rural with no major settlements (MPCPb undated). Land use consists mainly of intensively arable agriculture with cereal, potato and root crops cultivated over wide areas of the catchment and only a small proportion of the land given over to pasture and forage.

As mentioned earlier, the analysis of crop areas and farm types has been carried out for the area of the 12 parishes rather than the catchment. This is because the *UK June Agricultural and Horticultural Census Data* (JCD),[1] that were one of the main sources of information used, are collected and published on a parish basis. Although it is technically possible to identify the farms that fall within the catchment using the *Integrated Administration and Control System* (IACS) data, which provide spatially referenced information on the land use of agricultural parcels each year, confidentiality issues prevented their use in this case.

### 2.4.2 Production Activities

The crops to be modelled were determined primarily by using the JCD. The local agricultural extension officer was also consulted (E. Hart, pers. comm., 04/07/07). Most crops occupying more than 1% of the total area, in any single year, were selected for the analysis. A comparison with 2003 IACS data (E. Guillem, pers. comm., 11/03/2010) confirmed these crops as the most common in the area. Carrots were also considered, because they are a high value crop. Land uses associated with grass have been ignored as these are related to livestock activities that have not been considered in this study. *Set-aside* assumes the *sown cover* option under the set-aside management rules, and *peas* were assumed to be *peas for human consumption* or *vining peas*. The final list of crops/land uses consists of *winter wheat*, *winter*

[1] The JCD are collected and published annually, and provide holding level information on land use, crop areas, livestock numbers, labour use, and horticulture and glasshouse production. http://www.scotland.gov.uk/Topics/Statistics/Browse/Agriculture-Fisheries/PubFinalResultsJuneCensus

*barley*, *spring barley*, *spring oats*, *winter oilseed rape*, *seed potatoes*, *main crop potatoes*, *peas*, *carrots*, and *set-aside*. The selected land uses cover almost three quarters of the area, the remainder includes woodland, rough grazing and pasture grass. The crop products considered include *grain* and *straw* for cereals, *grain* for winter oilseed rape and peas, *seed* for seed potatoes, and *ware/root* for maincrop potatoes and carrots. For the combination of these crops into rotations, expert consultations took place with two experienced agronomists (J. Elcock, pers. comm., 30/11/07; 28/02/08; S. Hoad, pers. comm., 04/03/08).

The soil typology needed to adequately express soil heterogeneity within the area in terms of yield and leaching effects, without being too complicated, as this would lead to unrealistic data requirements, excessive bio-physical simulations, and difficulty in transposing the results into meaningful policy actions. Several classifications of Scottish soils and soil attributes were available for characterising the soils of the Lunan Water Catchment. The classification of homogenous soil classes was achieved by using the *Scottish Soil Type Classification System*,[2] the *Scottish Soils Knowledge and Information Base* data[3] (SSKIB) (Macaulay Institute), the *Soil Leaching Potential Classification* (Lewis et al. 2000), the *HOST Classification* (Boorman et al. 1995), and the typology described in the *SAC Technical Note T516 Nitrogen Recommendations for Cereals, Oilseed rape and Potatoes*[4] (Sinclair 2002). From the list of candidate soil types based on the soil series occurring in the catchment, the soil typology was simplified to two soil types (Soil A and Soil C) differing in their vulnerability to N leaching. The specific attributes that were used for the classification were *drainage class*, *leaching potential*, *HOST class*, and *T516 class* of the involved soils.

To assist in identifying the possible combinations of crops and soil types, the percentages of areas of each soil series for each of the crops was calculated by combining the *1:25,000 Scale Soils Data*[5] and the IACS data, without direct access to the IACS data to preserve farmer confidentiality (E. Guillem, pers. comm., 11/03/2010). The information for each soil series was then aggregated for the two defined soil types. Even though the proportions of crops differed between soil types, no combination could actually be excluded without introducing errors. Thus all combinations of land use and soil types were considered.

The agronomic techniques considered in the analysis were:

1. Fertilisation: Each activity can be characterised by two possible levels of N fertilisation: the first level represents the fertiliser recommendations for farmers in Nitrate Vulnerable Zones (NVZs) provided by Scottish Executive (2008)

[2] http://www.scotland.gov.uk/Publications/2006/09/21115639/17

[3] http://www.macaulay.ac.uk/mscl/gis2_dataset.php

[4] These recommendations form the basis for the Guidelines for Farmers in NVZs (Scottish Executive 2008).

[5] http://www.macaulay.ac.uk/mscl/gis2_dataset.php

(*ScA*) and the second corresponds to a 20% reduction in the recommended values (*ScB*).

2. Tillage: Traditional tillage has been assumed for all the activities, as it was advised that this is the most typical practice (E. Hart, pers. comm., 04/07/07).

Information on N fertiliser is used by both FSSIM-MP and COUP. FSSIM-MP requires input on total N use per crop within each agricultural activity and COUP requires the timing and N levels for all fertiliser applications within an agricultural activity. As there is no accurate source of information on N fertiliser applications in the catchment, we mainly used information from the Guidelines for Farmers in Nitrate Vulnerable Zones (Scottish Executive 2008), as (i) they take into account crop and soil requirements; (ii) they are tailored to Scottish agricultural systems, and (iii) a great part of the catchment falls within an NVZ. The two N fertiliser scenarios are based on the assumption that farmers take into account crop and soil requirements by respecting the rules in NVZs. The N fertiliser levels vary by crop, soil, and technique. The effect of the preceding crop in a rotation has not been taken into account because the difference between the recommended N levels for each of the two crop residual groups involved in this study was only 10 kg/annum. Such a small difference would not have had a significant effect on yield and nitrate leaching levels within rotations of different crops simulated with long-term weather data. As the guidelines do not provide information for seed potatoes and carrots, the SAC Technical Note T516 (Sinclair 2002) and the FMH (Chadwick, 2000–2002) values have been used for these crops, respectively. Timing and percentage of N application per N dose were provided by S. Hoad (pers. comm., 03/09/2008). The data used for the parameterization of the two models are shown in Table 9.1.

### 2.4.3 Farm Types

The construction of the farm typology has been achieved by using the JCD for the years 2001–2003. The U.K. Farm Classification System[6] was the starting point for the establishment of the typology, as it is tailored to British agricultural production systems. Additionally, it is used by the JCD, allowing consistency between data sources and farm level modelling. This typology uses the main source of income of the farm as the primary classification criterion. The U. K farm classification allows significant farm diversity in terms of cropping and livestock activities. As purely cropping farms have different production possibilities, equipment, farmers' abilities, and fertilisation potential from manure, compared to cropping farms also engaging in livestock activities, cropping farms have been further segregated into farms with and without livestock. Size has been represented by the economic farm output expressed in European Size Units (ESUs), as this criterion can be used

[6] http://www.scotland.gov.uk/Publications/2005/01/20580

**Table 9.1** N fertilisation data

| Crop | Soil | Fertiliser scenario | 1st N dose timing | 2nd N dose timing | 1st N dose quantity (kg/ha) | 2nd N dose quantity (kg/ha) | Total N (kg/ha) |
|---|---|---|---|---|---|---|---|
| Winter wheat | A | A | 05/04 | 05/05 | 78 | 117 | 195 |
| Winter wheat | A | B | 05/04 | 05/05 | 62.4 | 93.6 | 156 |
| Winter wheat | C | A | 05/04 | 05/05 | 86 | 129 | 215 |
| Winter wheat | C | B | 05/04 | 05/05 | 68.8 | 103.2 | 172 |
| Winter barley | A | A | 15/03 | 15/04 | 70 | 105 | 175 |
| Winter barley | A | B | 15/03 | 15/04 | 56 | 84 | 140 |
| Winter barley | C | A | 15/03 | 15/04 | 78 | 117 | 195 |
| Winter barley | C | B | 15/03 | 15/04 | 62.4 | 93.6 | 156 |
| Spring barley | A | A | 05/03 | 25/03 | 62.5 | 62.5 | 125 |
| Spring barley | A | B | 05/03 | 25/03 | 5 | 5 | 100 |
| Spring barley | C | A | 05/03 | 25/03 | 72.5 | 72.5 | 145 |
| Spring barley | C | B | 05/03 | 25/03 | 58 | 58 | 116 |
| Spring oats | A | A | 05/03 | 05/04 | 47.5 | 47.5 | 95 |
| Spring oats | A | B | 05/03 | 05/04 | 38 | 38 | 76 |
| Spring oats | C | A | 05/03 | 05/04 | 57.5 | 57.5 | 115 |
| Spring oats | C | B | 05/03 | 05/04 | 46 | 46 | 92 |
| W. oils. rape | All | A | 15/03 | 15/04 | 88 | 132 | 220 |
| W. oils. rape | All | B | 15/03 | 15/04 | 70.4 | 105.6 | 176 |
| Seed pot. | A | A | 10/05 | 30/05 | 42.5 | 42.5 | 85 |
| Seed pot. | A | B | 10/05 | 30/05 | 34 | 34 | 68 |
| Seed pot. | C | A | 10/05 | 30/05 | 52.5 | 52.5 | 105 |
| Seed pot. | C | B | 10/05 | 30/05 | 42 | 42 | 84 |
| Maincr. pot. | A | A | 06/05 | 26/05 | 110 | 110 | 220 |
| Maincr. pot. | A | B | 06/05 | 26/05 | 88 | 88 | 176 |
| Maincr. pot. | C | A | 06/05 | 26/05 | 120 | 120 | 240 |
| Maincr. pot. | C | B | 06/05 | 26/05 | 96 | 96 | 192 |
| Carrots | A | A | 06/05 | 26/05 | 25 | 25 | 50 |
| Carrots | A | B | 06/05 | 26/05 | 20 | 20 | 40 |
| Carrots | C | A | 06/05 | 26/05 | 30 | 30 | 60 |
| Carrots | C | B | 06/05 | 26/05 | 24 | 24 | 48 |
| Peas | All | All | n/a | n/a | 0 | 0 | 0 |
| Set-aside | All | All | n/a | n/a | 0 | 0 | 0 |
| Fallow | All | All | n/a | n/a | 0 | 0 | 0 |

*Source:* Own elaboration from Scottish Executive (2008); Sinclair (2002); Chadwick (2000–2002); S. Hoad (pers. comm., 03/09/2008)

for assessments between farms of different production orientation, and it is closely correlated to farm size. The ESU thresholds have been drawn from the Economic Report on Scottish Agriculture (ERSA) (Scottish Government, 2001), which identifies five ESU classes. Due to the small size of our farm sample, the above classes have been merged into two classes: (i) 40 and less; and (ii) above 40. The resulting typology of modeled farms (Table 9.2) uses the criterion of the U.K. Farm Classification farm type, the criterion of farm engagement with livestock activities, and the economic size of the farm expressed in ESUs.

**Table 9.2** Averages and standard deviations of farm type characteristics

| Farm type label | U.K. classification | Livestock activities | ESUs | Area (ha) | Crops (ha) | ESU | Labour/ha | ESU/ha |
|---|---|---|---|---|---|---|---|---|
| CC1 | Cereals | No | <40 | 21 (25) | 16 (18) | 8 (10) | 96 (241) | 0.43 (0.14) |
| CC2 | Cereals | No | >40 | 129 (66) | 113 (52) | 78 (40) | 17 (18) | 0.63 (0.15) |
| GC1 | General Cropping | No | <40 | 22 (15) | 17 (12) | 21 (12) | 40 (73) | 1.36 (1.04) |
| GC2 | General Cropping | No | >40 | 126 (88) | 110 (79) | 126 (98) | 15 (15) | 1.01 (0.26) |

*Source:* Own elaboration from JCD ( ): standard deviations

### 2.4.4 FSSIM-MP Data

The input coefficients for the characterisation of the agricultural activities in our application are labour requirements, and fertiliser inputs. In FSSIM-MP, these coefficients vary per rotation, crop, soil, and technique. For the estimation of labour requirements, the Standard Labour Requirements (SLR) in hours per hectare and per annum for different types of agricultural activities published by DEFRA (2010) and the FMH (Chadwick, 2000–2002) have been used. The SLRs of the FMH are more representative of practices in Scotland and have thus been used for most crops. For crops for which the FMH does not provide SLR figures, the SLR coefficients published by DEFRA have been used or appropriate assumptions have been made. No changes per rotation, soil type, or technique are assumed. N fertiliser inputs have already been discussed. P and K inputs have been extracted from the FMH (Chadwick, 2000–2002).

The output coefficients for each agricultural activity correspond to (i) yield per crop product for each rotation, crop, product, soil, and technique; (ii) yield variability per crop; and (iii) nitrate leaching per rotation, soil and technique. The yields for the main crop products and nitrate leaching coefficients are the key outputs of the bio-physical simulations, and are thus presented and discussed in the following section. Yields of straw for cereal products have been estimated from grain yields using the data on straw and grain yields in the FMH (Chadwick, 2000–2002). The coefficients were estimated to be equal to 0.65 for winter wheat, 0.75 for winter and spring barley, and 0.86 for spring oats. The yield variability per crop has been estimated using the annual national yield estimates published in ERSA (Scottish Government, 1994–2003).

The used economic data are (i) variable costs (except fertiliser costs) per rotation, crop, soil, and technique; (ii) fertiliser costs per rotation, crop, soil, and technique; (iii) prices per agricultural product; (iv) price variability per crop; and (v) wages for hired labour. The variable costs per crop have been estimated using the FMH (Chadwick, 2000–2002). The estimation of fertiliser costs has been based on prices quoted in the FMH (Chadwick, 2000–2002). The average price for N is equal to 0.35 £/kg, for P equal to 0.32 £/kg and for K equal to 0.20 £/kg.

**Table 9.3** Various input–output coefficients

| Crop | SLRs (hours/ha/annum) | P input (kg/ha) | K input (kg/ha) | Yield variab. | Variable costs (£/ha) | P and K costs (£/ha) | Price variab. |
|---|---|---|---|---|---|---|---|
| W. Wheat | 20 | 70 | 70 | 0.46 | 161 | 36.4 | 19.2 |
| W. Barley | 20 | 70 | 70 | 0.46 | 120 | 36.4 | 24.4 |
| S. Barley | 20 | 50 | 50 | 0.44 | 113 | 26 | 24.4 |
| S. Oats | 20 | 40 | 40 | 0.46 | 117 | 20.8 | 20.7 |
| W. Rape | 20 | 58 | 58 | 0.29 | 187 | 30.2 | 23.5 |
| Seed Pot. | 170 | 200 | 135 | 4.94 | 1,804 | 91 | 39.8 |
| M. Pot. | 170 | 150 | 240 | 6.35 | 1,594 | 96 | 40.6 |
| Peas | 32 | 25 | 25 | 0.41 | 208 | 13 | 40.6 |
| Carrots | 170 | 125 | 125 | 6.35 | 4,486 | 65 | 40.6 |
| Set-aside | 1 | 0 | 0 | 0 | 51 | 0 | 0 |

*Source:* Own elaboration from Chadwick (2000–2002); DEFRA (2010); ERSA (Scottish Government 1994–2003; 2006)

The figures published in the FMH were found appropriate to represent average prices as they are more likely to represent farmers' expectations on prices. On the other hand, ERSA reports past prices, as these have been formed in the market, and is thus more suitable for expressing the variability of past prices. Thus the FMH (2000–2002) has been used for the calculation of average prices, and the ERSA (Scottish Government, 1994–1996; 2000; 2003; 2006) has been used for the estimation of price variability for the period 1991–2003. For peas and carrots, the same variability as for maincrop potatoes has been assumed due to lack of information in ERSA. Wages have been assumed to be equal to the average minimum rate for full time workers in Scotland for 2001–2003, i.e. £4.52/h (Table 9.3).

The data related to the characterisation of farm types are (i) farm numbers; (ii) land availability per soil type; (iii) family labour availability; and (iv) crop pattern. The farm-related data have been estimated with the use of the JCD (2001–2003) using average values between the 3 years. Land availability per soil type and farm type has been estimated by combining the SSKIB data, the IACS data and the JCD. The exercise was repeated for years 2001–2003. The percentages of soil type per farm type varied significantly between the years, which demonstrates the limitations of available data for deriving such information. For the estimation of labour availability, the number of occupiers and spouses working *full-time*, *half-time or more*, or *less than half time* per farm type have been multiplied by their hours per year equivalent. These have been assumed to be 1,900 h for *full-time*,[7] 1,425 for *half-time or more*, and 475 for *less than half time*. Farm numbers and observed activity levels per crop and farm type have been directly estimated by the JCD (Table 9.4).

[7] http://www.scotland.gov.uk/Topics/Statistics/Browse/Agriculture-Fisheries/agritopics/farmstruc

**Table 9.4** Farm type information

| Farm type | Farms (No) | Land soil type A | Land soil type C | Family labour (hours/annum) |
|---|---|---|---|---|
| CC1 | 30 | 8.83 | 5.97 | 585.37 |
| CC2 | 7 | 92.15 | 22.14 | 1,886.81 |
| GC1 | 12 | 17.38 | 0.64 | 788.37 |
| GC2 | 72 | 88.40 | 15.11 | 1,481.69 |

*Source*: Own elaboration from JCD (2001–2003)

### 2.4.5 COUP Data

COUP requires information on daily precipitation, mean air temperature, net and global radiation relative humidity, and wind speed (Jansson and Karlberg 2004). Two weather data sets for the years 1974–1998 and 1999–2007 were obtained from the meteorological station at Mylnefield Dundee, which was considered representative of the Lunan Water Catchment. Missing daily values have been filled in by assuming equality to mean values of the previous and following day or to values corresponding to the same day of the year from other years. Mean air temperature, net radiation, relative humidity, and wind speed have been estimated from the raw data.

COUP requires a considerable amount of data for the parameterisation of the water/heat sub-model, which were not available for the soil series of the area considered. Values for hydraulic and thermal conductivity are not regularly measured. The values used have been obtained from a soil characteristics database present in COUP. The two soil candidates have been selected considering the similarities in organic matter, sand and silt content, through the soil profile, with the Scottish soils scenarios. The use of these soils has not been considered a limiting factor, as the COUP level of detail makes the similarities between the two areas adequate for this study.

Sowing dates have been provided by G. Russell (pers. comm., 08/08/2008). Harvest dates have been automatically calculated by the model as a function of crop development. The fertilisation data have been previously presented.

### 2.4.6 Bio-Physical Modelling Scenarios

The agricultural activities defined earlier resulted in 118 simulation scenarios being run with COUP. These consist of 29 rotations, under two alternative fertilisation scenarios, each on two soils, together with a continuous set-aside rotation on the two soils.

A 35 year simulation period for the years 1974–2008 has been used for all scenarios. The first 10 years (1974–1983) were needed to allow the initial model conditions to stabilise. Hence, the estimation of the coefficients was based on 24 years of model output, which was considered adequate to represent climatic variability. Since the rotations do not all consist of the same number of years, the number of occurrences of a rotation within the simulation period is not the same for all rotations.

When the average crop yields under fertiliser ScA were compared to the FMH (Chadwick, 2000–2002) yield estimates, it was found that the yields for maincrop and seed potatoes were considerably underestimated by COUP. To correct for this, the yield, but not the leaching, for these crops for each scenario was multiplied by a conversion factor (see Sect. 1.3.1).

### 2.4.7 Bio-Economic Modelling Scenarios

A number of agricultural and water policy scenarios were simulated. The baseyear scenario corresponds to Agenda 2000, the baseline scenario corresponds to the CAP Health Check, and the nitrate scenarios correspond to per unit taxes on N inputs and nitrate leaching, quotas on average N inputs at the farm level, and standards for average nitrate outputs.

The *baseyear* reflects a specific base period that relates to both model calibration and policy representation. The model is calibrated using data inputs representing the reference years of the base period. Thus the selected period should be representative of the typical socio-economic and climatic environments of the case study under examination and the years included in the base period should be homogenous in terms of policy regime so that this can be modelled uniformly. The selected base-year reference period for this study consists of the years 2001–2003. These years reflect the Agenda 2000 policy regime, that was introduced in Scotland in July 2000 and remained in force until the implementation of the 2003 CAP Reform in 2005. A description of the implementation of Agenda 2000 within our modelling framework can be found in Mouratiadou et al. (2008).

The *baseline* scenario represents the policy environment against which additional scenarios are compared and it represents the 2008 CAP Health Check. The key differences of this scenario from Agenda 2000 are changes that have been enforced with the introduction of the 2003 CAP Reform, i.e. the decoupling of payments and the introduction of compulsory modulation, and changes that were introduced with the CAP Health check, i.e. the abolition of set-aside obligations and the attribution of set-aside entitlements to other land uses.

The following policy scenarios explored the effects of taxes on N inputs or nitrate emissions. The tax level has been set as a function of the price of N fertiliser in year 2010, assumed to be equal to 0.52 £/kg (McBain and Curry 2009). The tax scenarios have been simulated by the incorporation of additional cost factors in the model objective function. The scenarios on N input quotas and nitrate emission standards simulate the effects of these measures on an average per hectare basis.

The income equation and additional model constraints used for the simulation of the scenarios are shown below:

$$
\begin{aligned}
Z_f =& \sum_j p_j q_{f,j} - \sum_{i,t} c_{i,t} \frac{X_{f,i}}{\eta_i} + \sum_{i,t} \frac{\left(d_{f,i,t} + \frac{\Psi_{f,i,t} X_{f,i}}{2}\right) X_{f,i}}{\eta_i} \\
& - \varpi L_f + \left( \left( \sum_{i,t} s_{i,t} \frac{X_{f,i}}{\eta_i} \right) (1 - v) - P_f m \right) \left(1 - rV_f\right) - kT_f - hQ_f
\end{aligned}
\tag{9.8}
$$

$$
\frac{T_f}{G_f} \leq St \qquad \forall f \tag{9.9}
$$

$$
\frac{Q_f}{G_f} \leq Qu \qquad \forall f \tag{9.10}
$$

where $v$ is a scalar representing the rates of voluntary modulation, overshooting of base areas and the national reserve, $P$ is a vector of the amount of premiums that exceeds the amount that is exempt from compulsory modulation, $m$ is a scalar for compulsory modulation rate, $r$ is a scalar for the rate of premium reductions if cross-compliance is not respected, $V$ is a vector of the binary variable associated with cross-compliance measures, $k$ is a scalar for the level of tax per kg of nitrate leaching, $T$ is a vector of nitrate leaching at farm level, $h$ is a scalar for the level of tax per kg of N input, $Q$ is a vector of N inputs at farm level, $G$ is a vector of available land per farm type, $St$ is a scalar for the nitrate leaching standard, and $Qu$ is a scalar for the N input quota.

First, the above scenarios were run for all farm types and all rotations. The taxes have been varied between 0% and 200% of the price of N fertiliser, in ten parametric simulations at increments of 20% of the price per simulation. The starting value for quotas and standards has been the highest level of average N use or nitrate leaching at the farm level in any of the four farm types for the baseline scenario. Thus, the quota on N input ranged between 170 and 70 kg/ha in ten parametric simulations of increments of 10 kg/ha, and the standard on nitrate leaching ranged between 60 and 20 kg/ha in eight parametric simulations at increments of 5 kg/ha. Subsequently, similar scenarios were run for farm type CC2 including only rotations without rotational set-aside. Input and leaching taxes ranged between 0–500% and 0–1,000% of the fertiliser price in ten scenario simulations at increments of 50% and 100% per scenario, respectively. Standards on leaching were varied between 40 and 20 kg/ha at increments of 2 kg/ha, and quotas on N inputs between 160 and 10 kg/ha at increments of 10 kg/ha.

# 3 Results

## 3.1 Bio-Physical Modelling Results

In order to facilitate comparison between soil and fertiliser scenarios the fertiliser and soil combinations have been defined as follows: (i) *Sc1*: Soil A + ScB; (ii) *Sc2*: Soil C + ScB; (iii) *Sc3*: Soil A + ScA; (iv) *Sc4*: Soil C + ScA.

Yield estimates provided in the FMH were compared to the fertiliser ScA average values per crop (Table 9.5). Model predictions are satisfactory for most crops with the exception of maincrop and seed potatoes, where yields are significantly under-predicted. Yields from simulations for these crops have been multiplied by the estimated conversion factor shown below. Yields are slightly under-predicted for wheat and over-predicted for spring barley.

Table 9.6 shows average yields and relative differences of the averages between scenarios. Cereal crops show a realistic pattern of variability attributed to climate, previous crop in the rotation, soil and fertilisation level. The highest yields are achieved for the highest fertiliser input (Sc4) and the lowest yield to lowest fertiliser input (Sc1). Sc2 and Sc3 provide very similar outputs, due to the similarity in the

**Table 9.5** Comparison of yield estimates from literature and model predictions

| Crop | FMH | ScA | Relative difference FMH-ScA | Applied conversion factor |
|---|---|---|---|---|
| W. Wheat | 8 | 7.08 | −12.20 | 1 |
| W. Barley | 7.5 | 7.67 | 2.24 | 1 |
| S. Barley | 5.5 | 6.86 | 22.01 | 1 |
| S. Oats | 5 | 5.21 | 4.11 | 1 |
| W. Rape | 3.5 | 3.71 | 5.83 | 1 |
| Seed Pot. | 23 | 10.38 | −75.61 | 2.22 |
| M. Pot. | 50 | 33.27 | −40.18 | 1.50 |
| Peas | 4.6 | 4.5 | −2.20 | 1 |
| Carrots | 43.7 | 44.73 | 2.33 | 1 |

*Source*: Own elaboration from Chadwick (2000–2002); COUP outputs

**Table 9.6** Absolute and relative average yields per crop and scenario

| Crop | Sc1 | Sc2 | Sc3 | Sc4 | RD Sc1–Sc2 | RD Sc3–Sc4 | RD Sc1–Sc3 | RD Sc2–Sc4 |
|---|---|---|---|---|---|---|---|---|
| WDWH | 5.39 | 6.44 | 6.52 | 7.64 | 17.89 | 15.83 | 18.99 | 16.93 |
| WBAR | 6.92 | 7.47 | 7.42 | 7.91 | 7.71 | 6.39 | 7.04 | 5.71 |
| SBAR | 5.88 | 6.48 | 6.60 | 7.13 | 9.68 | 7.69 | 11.50 | 9.52 |
| OATS | 4.16 | 5.08 | 4.90 | 5.52 | 20.07 | 11.90 | 16.44 | 8.26 |
| RAPE | 3.77 | 3.52 | 3.88 | 3.55 | −6.95 | −9.02 | 2.88 | 0.81 |
| SDPO | 23.67 | 22.05 | 24.00 | 22.07 | −7.07 | −8.36 | 1.38 | 0.09 |
| POTA | 50.17 | 48.77 | 50.59 | 49.22 | −2.82 | −2.76 | 0.85 | 0.91 |
| PEAS | 4.74 | 4.21 | 4.76 | 4.25 | −11.72 | −11.30 | 0.50 | 0.92 |
| CARR | 44.29 | 45.16 | 44.30 | 45.16 | 1.95 | 1.94 | 0.01 | 0.00 |

*RD* relative difference

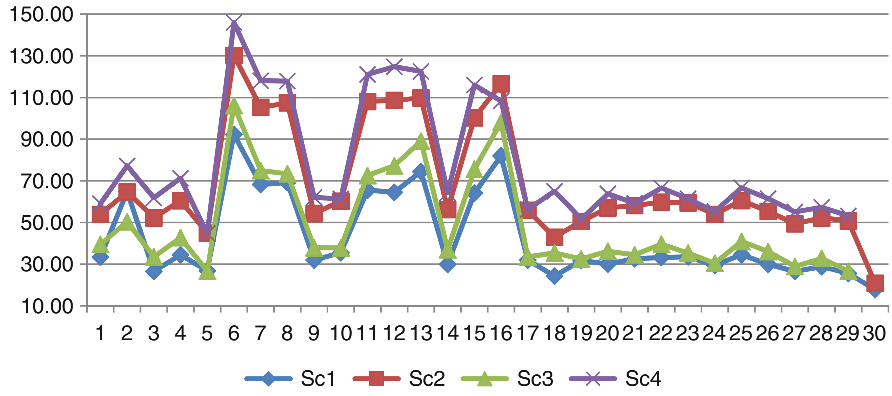


**Fig. 9.2** Average annual rotational leaching

fertiliser inputs. Since the fertiliser input for most crops is slightly higher for Sc3, the yields are also higher. This indicates either that that the model is more sensitive to N inputs than to soil attributes, or that the fertiliser levels proposed for soil A result in lower yields compared to those proposed for soil C.

For the other crops, yield estimates are insensitive to soil, and even less sensitive to fertiliser levels. For potato crops, carrots and peas, a very small difference is observed between the two soils. The difference between soils is negligible for oilseed rape. Differences between fertiliser scenarios within each of the two soil types are insignificant for all crops. This result was unexpected and may be due to the parameterisation of the crop model component in relation to these crops, or to a poor capacity of the model to simulate non cereal crops.

Leaching values for the simulated scenarios can be seen in Fig. 9.2 and basic statistical figures for leaching per soil and fertilisation scenario in Table 9.7. Leaching values are higher for Sc4 and Sc2, as soil C is more vulnerable to leaching. The average relative difference between soils for each of the two fertilisation scenarios is about 50%, and between fertilisation scenarios for each of the two soils about 10%.

The highest leaching corresponds to rotations with rotational set-aside. Despite there being no N fertiliser inputs for set-aside, incorporation of grass cover at ploughing prior to the sowing of the crop following set-aside leads to a massive release of nitrates. However, although expert opinion suggests that leaching from set-aside after ploughing might range from 30 to 200 kg/ha (B. Rees, pers. comm., 17/02/11), the model seems to be over-predicting mineralisation of background N in organic soils. This effect may also have been aggravated by crop model parameterisation in relation to vegetation cover, which results in low N uptake by this land use. This is not the case for the non-rotational set-aside rotation, which corresponds to the lowest leaching level per scenario, as it has been assumed that nutrient demanding weeds grow in the field. This rotation corresponds to the lowest nitrate leaching on both soils (Table 9.7).

**Table 9.7** Average annual leaching

| **All rotations except non-rotational set-aside** | **Sc1** | **Sc2** | **Sc3** | **Sc4** |
|---|---|---|---|---|
| Average (kg/ha) | 43.31 | 70.32 | 48.78 | 77.59 |
| Standard Deviation | 20.23 | 26.22 | 23.12 | 28.97 |
| Minimum (kg/ha) | 24.26 | 42.96 | 26.59 | 44.7 |
| Maximum (kg/ha) | 92.28 | 130.21 | 106.17 | 146 |
| **Rotations without set-aside** | **Sc1** | **Sc2** | **Sc3** | **Sc4** |
| Average (kg/ha) | 32.18 | 54.90 | 35.61 | 60.69 |
| Standard Deviation | 8.15 | 5.36 | 5.48 | 7.04 |
| Minimum (kg/ha) | 24.26 | 42.96 | 26.59 | 44.70 |
| Maximum (kg/ha) | 64.89 | 64.67 | 50.37 | 77.20 |
| **Rotations with rotational set-aside** | **Sc1** | **Sc2** | **Sc3** | **Sc4** |
| Average (kg/ha) | 72.52 | 110.79 | 83.37 | 121.84 |
| Standard deviation | 9.98 | 9.08 | 12.86 | 10.95 |
| Minimum (kg/ha) | 64.11 | 100.29 | 72.50 | 108.37 |
| Maximum (kg/ha) | 92.28 | 130.21 | 106.17 | 146 |

## *3.2 Bio-Economic Modelling Results*

Taxes on nitrate leaching caused a very inelastic response for both nitrate leaching and N use. For cereal farms the marginal change between scenarios and the average change in both N use and N leaching coincide, due to a linear response within the sequence of the ten tax scenarios. Nitrate losses per scenario for farm types CC1, CC2, and GC2 respectively reduced by (i) 0.02 kg/ha, (ii) 0.10 kg/ha, and (iii) 0.12 kg/ha until the tax reaches 120% of the price of commercial fertiliser, and by 0.15 kg/ha thereafter. N use reduced by (i) 0.05 kg/ha, (ii) 0.30 kg/ha, and (iii) 0.32 kg/ha. GC1 shows the most responsive change in nitrate losses, which is, however, associated with an increase in N use. Marginal per scenario decrease in nitrate losses ranges between 0.56–1.61 kg/ha, and marginal N use gradually increases by 0.42 kg/ha up to 0.49 kg/ha. Utility and income losses are almost linear along the sequence of scenarios and correspond to 4 £/ha per increment of 20% tax increase, for farm types CC1, CC2, and GC2, and 5 £/ha for GC1.

For CC1, changes are associated with a linear increase in the prevalence of a rotation with spring barley and oats and a corresponding decrease of a rotation with spring barley and rape, both under Sc3. These changes target the rotation with the highest leaching which is substituted by the rotation with the lowest leaching. Even though rotations under Sc4 correspond to higher leaching, these are not altered as this would induce a higher loss in yields and a lower relative decrease in nitrate leaching. A similar pattern, but for different rotations, is observed for CC2 and GC2. In the case of GC1, the decrease in leaching is achieved by a reduction of rotations with set-aside, thus the corresponding increase in N use. It should be noted that GC1 is the only farm type where set-aside appears in the baseline land use mix. As a consequence average nitrate leaching in the baseline is higher compared to the other farm types, and this explains greater responsiveness of this farm type to the applied tax.

Taxes on inputs caused a more elastic response compared to scenarios of the same level of taxes on nitrate leaching. This is a realistic result due to the higher level of N use compared to nitrate leaching at a field and farm level. Marginal per scenario N use reductions for CC1, CC2, GC1, and GC2 respectively range between (i) 0.49–7.04 kg/ha, (ii) 0.98–7.41 kg/ha, (iii) 0.84–1.79 kg/ha, and (iv) 1.22–2.78 kg/ha. In most cases, marginal N use decreases tend to be higher than the average changes when the tax is higher than 100% of the price of fertiliser. Nitrate leaching initially decreases slightly for CC1 and GC2, but starts to increase after the tax goes beyond 100% and 60% of the fertiliser price, respectively. For GC1, nitrate leaching increases, with the marginal per scenario increase ranging from 0.28 to 0.49 kg/ha. Nitrate leaching decreases only in the case of CC2, where the marginal decrease ranges between 0.20 and 0.49 kg/ha. Marginal utility losses per 20% increase of the tax for CC1, CC2, GC1, and GC2 respectively range between (i) 16–13 £/ha, (ii) 17–16 £/ha, (iii) 12–11 £/ha, and (iv) 15–13 £/ha. Marginal income losses are not in complete accordance with marginal utility losses, which is expected as the objective function of the model takes into account both income and income variability.

In the case of CC1, changes correspond to decreases in all rotations that contain winter crops, and in particular oilseed rape, and increases in the rotation of spring barley and oats, all on soil A. This is because oilseed rape has considerably higher N inputs compared to spring crops. On soil C, until the 100% tax there is a substitution of a rotation of winter barley and spring barley by a rotation of spring barley and oats. Beyond this point there is an increase in rotations containing set-aside. This explains why nitrate leaching appears to be increasing after this tax scenario. In GC2 the pattern is similar to CC1, only that the set-aside rotations, and consequently leaching, start increasing at the 60% tax. For GC1, on soil C changes are sough through changes in fertiliser levels. On soil A, there is an increase in rotations with peas and seed potatoes as these crops have no and low inputs, respectively. A very slight increase in the set-aside rotation is observed. This is partly, but not fully, the cause of the increase in nitrate leaching. This case provides some evidence that it is likely to have a decrease in N inputs accompanied by an increase in nitrate leaching. For CC2, no set-aside rotations appear along the ten tax scenarios. N input reductions are achieved by reductions in the fertilisation intensity on both soils, and by decreases in rotations containing many winter crops and in particular oilseed rape.

Setting standards for nitrate leaching is generally accompanied by a reduction of N use for all farm types (Fig. 9.3). For cereal farms the standard starts to have an effect after the level of 40 kg/ha, as average per hectare leaching at the farm level is already below this threshold in the baseline scenario. A similar pattern is observed for GC2. GC1 has the highest average leaching values amongst all farm types and thus land use changes start to be induced at a standard level of 50 kg/ha (S50). In general, N use is reducing as nitrate leaching is reducing, with the exception of GC1 between scenarios S50–S35. Marginal utility and income losses are increasing at a high rate for all farm types. Marginal utility losses per scenario for CC1, CC2, GC1,

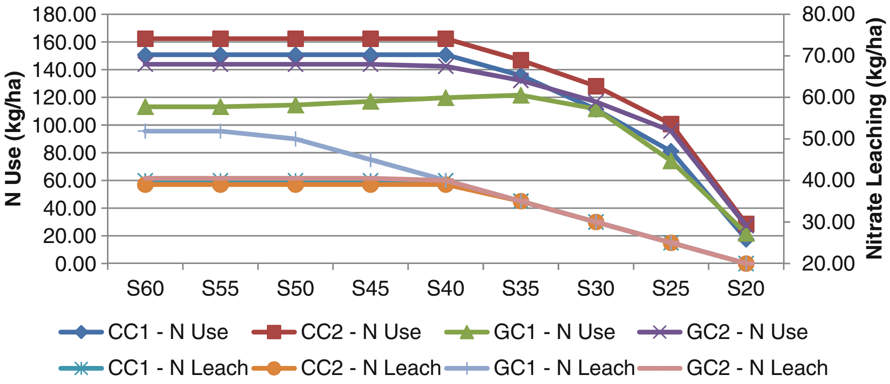


**Fig. 9.3** Farm-level average N use and leaching for scenarios on leaching standards

and GC2 respectively are (i) 14–213 £/ha, (ii) 11–262 £/ha, (iii) 3–392 £/ha, and (iv) 10–452 £/ha. Marginal income losses are higher than marginal utility losses.

The main land use change induced by the standards is an increase in non rotational set-aside. This is the case after scenario S30 for the small farms and after scenario S25 for the big farms. This result is expected as the non-rotational set-aside rotation corresponds to the lowest nitrate leaching. Prior to this, the main land use patterns are switches into low nitrate leaching rotations, as for example in CC2 a switch from a winter wheat and spring barley rotation under Sc3 (leaching 32.43 kg/ha) into a spring barley and rape rotation under Sc1 (leaching 24.26 kg/ha).The increase of N use with decreasing nitrate leaching observed for GC1 between S50 and S35 is caused by replacement of rotations with rotational set-aside.

Quotas on inputs do not result in a uniform relationship between N use and leaching (Fig. 9.4). Similar land use patterns are observed in all farm types. These correspond to increases in set-aside and lowering of fertilisation intensity. Farmers initially switch towards rotations with rotational set-aside as these are more profitable compared to non-rotational set-aside. However, when the quota goes below a certain threshold, a switch towards non rotational set-aside is unavoidable for reaching the input level required. That is why leaching is increasing at the first quota levels and then reducing at the very last scenarios. Marginal utility and income losses per scenario are gradually increasing for all farm types. Marginal utility losses are 6–37 £/ha for CC1, 7–35 £/ha for CC2, 1–86 £/ha for GC1, and 1–70 £/ha for GC2.

Similar scenarios were simulated for farm type CC2 including only rotations without rotational set-aside. This was performed in order to isolate the effect of increasing nitrate leaching with decreasing N use at the farm level that in most cases was caused by the high leaching observed for rotations with rotational set-aside. Table 9.8 shows the absolute values of per ha utility, income, N use and nitrate leaching for the baseline scenario, and the relative differences compared to the baseline for selected scenarios. Analysing the relationship between N use and N

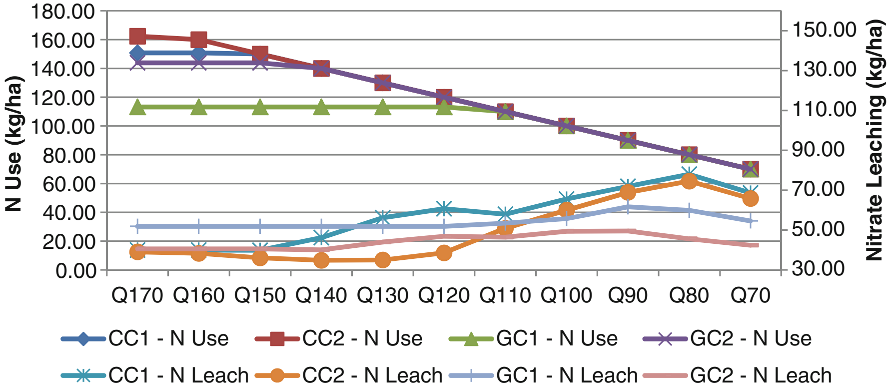


**Fig. 9.4** Farm-level average N use and leaching for scenarios on input quotas

**Table 9.8** Economic and environmental results for selected scenarios

| | Utility (£/ha) | Income (£/ha) | N use (kg/ha) | Leaching (kg/ha) |
|---|---|---|---|---|
| Baseline | 629.8 | 710.8 | 158.1 | 38.5 |
| Tax leaching – 500% | −15.6 | −13.5 | 0.3 | −3.0 |
| Tax leaching – 1,000% | −29.9 | −26.3 | −13.5 | −16.4 |
| Tax input – 150% | −19.5 | −17.1 | −1.1 | 0.3 |
| Tax input – 500% | −57.0 | −50.4 | −22.0 | −19.5 |
| Stand. leaching – 38 kg/ha | 0.0 | 0.1 | 0.1 | −1.2 |
| Stand. leaching – 20 kg/ha | −59.0 | −61.1 | −81.4 | −48.0 |
| Quota input – 90 kg/ha | −22.1 | −22.1 | −43.1 | −35.7 |
| Quota input – 10 kg/ha | −70.5 | −72.2 | −93.7 | −49.7 |

leaching, it can be seen that there are cases where measures aiming at a reduction of leaching are not accompanied by a reduction in N use (Tax Leaching – 500%, Stand. Leaching – 38 kg/ha), and conversely measures aiming at N use reductions are not accompanied by reductions in leaching (Tax Input – 150%). Additionally, one of the scenarios shows the possibility of achieving minor leaching reductions without utility losses (Stand. Leaching – 38 kg/ha).

Figure 9.5 shows the trade-off curves between utility and leaching reductions and thus the relative cost-effectiveness of measures, for all the simulated scenarios. Not surprisingly, the highest cost-effectiveness and responsiveness of the outcome to the measure is achieved by standards on leaching. Quotas on inputs do not result in significantly higher utility losses, due to the high prevalence of spring barley and subsequently set-aside for both leaching and quota scenarios. Taxes are less cost-effective, if considering only the costs imposed on farmers. This is because they reflect both the costs incurred by farmers from changing land use and intensity patterns and the payments for the associated taxes. If tax payments are considered as a gain to society, the net utility losses are similar to the ones observed in the trade-off curves for quotas and standards.

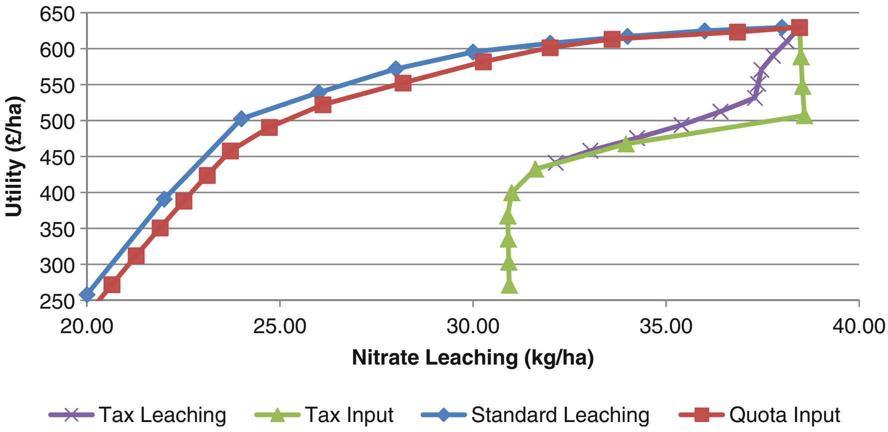


**Fig. 9.5** Trade-off curves between utility and leaching

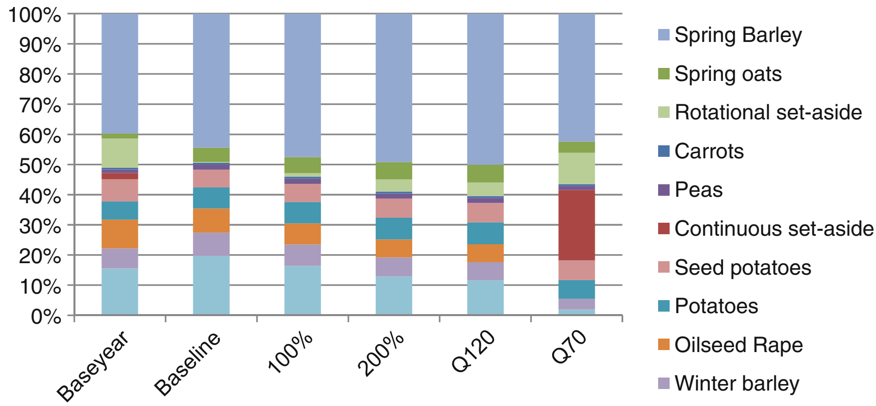


**Fig. 9.6** Catchment-level land use

Figure 9.6 shows the land use implications of the CAP Health Check, the 100% and 200% taxes on inputs, and the 120 and 70 kg/ha quotas, at the catchment level. In the baseline scenario, the abolition of set-aside leads to the replacement of both rotational and continuous set-aside by winter cereals. The imposition of a 100% tax on N inputs leads to an increase in the level of spring crops and in particular spring barley, a slight increase in rotational set-aside, and a reduction in winter cereals and particularly winter wheat. These effects are augmented when the tax increases to 200%. Similar results are obtained from the imposition of a quota of 120 kg/ha. When the N fertiliser quota level is very low, land abandonment occurs, at the expense of high input crops such as oilseed rape and winter wheat. Potato crops are not particularly affected by the scenarios but this may be due to model calibration effects.

## 4 Discussion

Considering the available input values, FSSIM-MP shows a reasonable response to the simulated agricultural and water policy measures. Additionally, previous work (Mouratiadou et al. 2008) has demonstrated the model's capacity to represent farmers' decision making in a realistic manner, through comparison of model predictions with observed land use patterns. The key implications of taxes and quotas on N inputs are land use swapping from winter into spring crops as they are associated with lower N inputs, and lowering of fertilisation intensity in rotations where the relative differences in yields are lower. High levels of taxes induce an increase of rotations with set-aside as these are still more profitable than land abandonment represented by the non rotational set-aside. As it would be expected, similar results are obtained for quotas on inputs. However, after a certain level the required reduction can only be achieved by land abandonment accompanied by significant economic losses. Nitrate leaching taxes result in replacement of rotations with high leaching appearing in the baseline rotation mix, by rotations with lower leaching. The most important reductions are found for those rotations where the difference between utility losses is lower and nitrate leaching gains higher. Similar patterns are observed in the case of leaching standards. However compared to the effects of nitrate leaching taxes, changes in intensity levels and land abandonment are also necessary for reaching certain standard levels.

In some of the simulated scenarios, even where rotational set-aside is not involved, there is some evidence that N use reductions do not result in nitrate leaching reductions. The reasons for this are exposed and analysed Chap. 1 of this book. Previous work (Mouratiadou et al. 2010; Belhouchette et al. 2011) has also showed that the relationship between N inputs and nitrate leaching at the farm level is not straightforward. In the current research, this effect is more pronounced for simulations containing rotational set-aside. This highlights the importance of considering multiple factors, such as ploughing after set-aside periods, for assessing the polluting effects of agricultural activities. However, before these results are applied, further work is needed to ensure that the observed results are consistent with reality and are not caused by over-estimation of nitrate leaching for rotational set-aside. Nevertheless, a strongest correlation between N inputs and nitrate leaching is observed in simulations without rotational set-aside, providing some justification for measures targeting N use as a means to abate nitrate leaching. The cost-effectiveness of measures targeting either N use or nitrate leaching was not found to differ much, as in the simulated farm type the predominant activities associated to low leaching correspond also to lower input levels. Nevertheless, the cost-sharing between farmers and society differ between taxing and command-and-control policies, as taxing instruments result in significantly higher costs for farmers which represent however revenues to society.

A simultaneous exploration of the effects of agricultural and environmental policies on other pollutants, such as phosphorus emissions and greenhouse gases, would also be possible within the employed framework. Indeed it would be desirable for, as Belhouchette et al. (2011) showed a policy aimed at the resolution of one environmental problem may result in counter intuitive effects on others. Additionally, a comparison of the cost-effectiveness of measures through marginal abatement cost curves (e.g. MacLeod et al. 2010; Mouratiadou et al. 2010) would be an interesting addition to this work.

COUP provided satisfactory results regarding absolute and relative values of yields of cereal crops under different soil and fertiliser scenarios, and relative differences in leaching. This was not the case for non-cereal crops. The flexibility of the plant growth sub-model in terms of the type of plant that can be simulated is a significant model advantage. However, the drawback is that several assumptions and compromises need to be made prior to full model implementation, and this might simplify or exclude some of the more plant specific processes or physiological characteristics, or lead to misspecification of the associated parameters. This experience reinforces the argument for extensive testing of models prior to their utilisation, particularly when a model is applied in a new environment. However, a serious limiting factor for this task is the limited amount of experimental information on both yield and nitrate leaching associated with different levels of inputs and soils against which model predictive capacity can be assessed. Such data sets are truly scarce, and even when they do exist they are hardly traceable beyond the teams that conducted the experiments. For progress to be made regarding model testing and validation, the gaps between experimental and mechanistic approaches should be narrowed. Certainly the creation of databases where experimental data could be publicised would be a valuable asset.

Additionally, this work emphasises the necessity of striking the appropriate balance between model complexity and practicality in implementation in a bio-economic modelling framework, and at the same time the need for generic and user-friendly tools. Bio-economic modelling requires (i) a considerable number of simulations per rotation, soil and technique, for a long-term sequence, and (ii) easily obtainable outputs from bio-physical simulations. Clearly, there are very few models that are generic enough so as to accurately reproduce most plant and soil systems. Additionally, the majority of bio-physical models, including COUP, are not intended for bio-economic modelling use and are thus subject to severe operational limitations regarding procedures to import the required parameters, use of rotations rather than crops as model objects, output extraction time, and format and time scale of the generated output. This comes at a significant cost regarding time requirements per simulation scenario, that may lead to simplifying assumptions regarding the number of agricultural activities considered. In this study, such limitations led to the simulation of a limited number of rotations and fertiliser levels that might have consequences for model results.

Another challenge in bio-economic modelling applications is the successful integration across systems and scales in the absence of a complete set of appropriate data. In this study, this is demonstrated by lack of sufficient information for separating the farms of the catchment from the farms of the broader area of the parishes, difficulties associated with estimating the soil distribution per soil type within farms and eventually farm types, and insufficiency of information for characterising agricultural activities in terms of their intensity levels in physical inputs and outputs. Further, established farm classification schemes disregard some aspects of importance, such as variables that successfully represent the essential inputs and outputs of agricultural production (Kostrowicki 1977), differences in the proportions of resource endowments and size, yields, and technologies (Hazell and Norton 1986) and criteria for assessing the environmental performance of farms (Andersen et al. 2007). The source of the above limitations is threefold. Firstly, data on farms have been typically collected for administrative regions. Integrated analysis requires overlaying administrative and natural boundaries, which implies that perhaps an approach that respects natural boundaries and collects information on the natural characteristics of farms is more appropriate when considering that the farm should no longer be considered as a business unit that operates regardless of its surrounding environment. Secondly, strict confidentiality agreements exist for existing but not publicly available data. Thirdly, the economic efficiency of agricultural production and farms has for long been the centre of attention, and thus farm data are typically collected in economic as opposed to physical units. Current policies, and as a consequence research priorities, focus on multi-functionality and environmental efficiency of agriculture. For relevant policy questions and research endeavours to be meaningfully explored, the nature and focus of public statistics also need to move towards this direction.

## 5 Conclusions

The bio-economic modelling approach used in this paper provides an appropriate and consistent framework for agricultural and water policy assessment in the agricultural sector, as it integrates economic, agronomic and environmental information. Thus, it allows the integrated assessment of how agricultural and water management policies along with the natural environment are likely to influence farmers' choices, and how in turn these choices impact on the natural environment, by taking into account both the socio-economic and environmental products of agriculture. This is achieved at three spatial scales: the field scale capturing agronomic and environmental diversity, the farm scale that offers a better representation of farmers' potential behaviour, and the catchment scale that allows consideration of the aggregate policy impacts. Additionally, the approach allows the distributional effects of policies on different farms to be explored.

The paper demonstrates the complexity of the issues involved, and highlights the challenges to be overcome. The latter are related to the lack of truly generic ready-to-use bio-physical simulation models, operational limitations imposed by insufficient procedures for model communication, and limitations of publicly available data. The former implies that even though models are efficient tools for ex-ante impact assessment of policies, model outcomes should be considered as hypotheses that become the input to further discussions with experts, farmers and policy makers rather than definite answers to policy questions. Results were found to differ considerably depending on initial assumptions on nitrate leaching, rotations, and associated management. This reveals the need for intensive sensitivity analysis of end results in relation to model inputs. Additionally, subsidies, cross-compliance measures, and measures targeting soils more vulnerable to leaching are to be considered. Further research aims in these two directions.

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# Chapter 10
# Integrated Bio-Economic Farm Modeling for Biodiversity Assessment at Landscape Level

**M. Schönhart, T. Schauppenlehner, and E. Schmid**

## 1 Introduction

The United Nations declared 2010 as the *Year of Biodiversity* to raise public awareness on the role of biodiversity in supplying ecosystem services to humans. It shall also make aware of the objectives of the Rio *Convention on Biological Diversity*. The convention calls for a significant reduction of biodiversity losses from national to global scales by the year 2010, (Convention on Biological Diversity 2010). Among the most important drivers, i.e. land use, atmospheric $CO_2$ concentration, nitrogen deposition, acid rain, climate, and biotic exchanges, land use have had and will have in the twenty-first century the most important although bi-directional effects on biodiversity globally (cf. Sala et al. 2000). One direction is that agricultural land use is responsible for severe losses through the conversion of natural habitats to farmland as well as for the on-farm losses induced by production intensification (Pimm and Raven 2000; Secretariat of the Convention on Biological Diversity 2006). The other direction is that crop and animal breeding have enriched genetic diversity and extensive agricultural land use has created cultural landscapes of high ecological values and unique semi-natural habitats (Wrbka et al. 2004; Fischer et al. 2008; EEA 2009). However, ongoing processes in agriculture such as intensification and abandonment of farmland can threaten these high nature value (HNV) landscapes (Benton et al. 2003;

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Tscharntke et al. 2005) and may reduce the ecosystem services provided to the society (Björklund et al. 1999). Intensification of farmland is frequently accompanied by high agro-chemical inputs. Semi-natural landscape elements such as field margins or hedges have been removed as a consequence of field consolidations to facilitate mechanization. Fragmented farmland has been negatively perceived by stakeholders such as 'the blackest of evils, to be prevented by legislative action as one would attempt to prevent prostitution or blackmail' (Farmer 1960, p. 225; cited in Bentley 1987, p. 31). In contrast, ecologists and agronomists have often alerted to the loss of valuable landscape elements as consequence of field consolidations (Krebs et al. 1999; Benton et al. 2003) with biodiversity as 'the big loser of technological changes in agriculture' (Giampietro 1997, p. 161). Many species have been able to adapt to changing environments during the previous millennia of agricultural development. However, adaptation is limited with fast and large scale changes such as during agricultural industrialization (Tucker 1997). Its scale and dynamics of pressures may even increase under global change phenomena such as climate and demographic changes (Tilman et al. 2001). Furthermore, abandonment of marginal agricultural lands as observed in several parts of Europe (Höchtl et al. 2005; Strijker 2005) often lead to substantial losses of HNV farmland (Tasser and Tappeiner 2002; EEA 2009). Consequently, the European Policy for instance has adopted biodiversity policies in recent reforms such as the birds and habitats directives, the NATURA 2000 networks, or agri-environmental measures as part of the Common Agricultural Policy (CAP) (European Commission 2006). Monitoring and evaluation are already integral elements of many policies. They require scientific analysis tools to investigate complex systems such as agricultural land use and ecosystem effects ex-post as well as ex-ante (Pain and Pienkowski 1997; Mattison and Norris 2005). Integrated land use models are able to analyze such complexities by linking thematic data and disciplinary models.

In this article, an integrated farmland use modeling framework (IMF) is applied to analyze impacts of agri-environmental measures on biodiversity at landscape level. Opportunity costs of biodiversity provision at farm and landscape levels are assessed for an Austrian case study landscape. We do not attempt to model the development of single species but rather apply surrogate indicators, correlations, and sensitivity analysis for species and habitat diversity. We also provide a literature review on landscape ecological foundations for biodiversity in agricultural landscapes and show how biodiversity issues have been applied in land use models (Sect. 2). In Sect. 3, we present the IMF including the data requirements and indicator set applied for biodiversity assessment. Sect. 4 describes the case study region and scenarios. It is followed by a presentation (Sect. 5) and discussion (Sect. 6) of model results, their policy implications, and remaining methodological challenges.

## 2 Biodiversity from a Landscape Ecological and Agricultural Economic Perspective

### *2.1 Biodiversity and Agricultural Land Use*

Reviews on landscape ecological studies identify a vast amount of concepts, definitions, and indicators with respect to biodiversity and highlight the need for well defined value systems, corresponding research objectives, and indicators (Duelli and Obrist 2003; Clergue et al. 2005). A basic categorization applicable to different spatial levels separates structural, functional, and compositional attributes of biodiversity (Noss 1990). The latter represents the frequently applied concept of biodiversity i.e. species or habitat diversity in agricultural landscapes (Duelli and Obrist 2003). In our analysis, we refer to this concept of biodiversity due to its central role in conservation policies.

Species and habitat diversity in agricultural landscapes may be influenced by a number of natural site conditions such as slope gradients, soil quality, and climate (Kleijn et al. 2009), but agricultural land use seems most relevant with respect to the magnitude of effects and to controllability. Particularly two aspects are seen as important, which are land use intensity at the field level (e.g. application rates of agro-chemicals, mowing frequencies and livestock densities of meadows and pastures) and the composition and configuration of landscape elements at the landscape level (e.g. extent and distribution of semi-natural farmland, diversity of agricultural crops) (Benton et al. 2003; Tscharntke et al. 2005; Billeter et al. 2008; Concepción et al. 2008; Kleijn et al. 2009). Landscape complexity or landscape structure refers to the spatial distribution of ecotopes such as fields, hedges, or trees in a landscape (cf. Wrbka et al. 2004).

Empirical studies indicate that land use intensity and landscape complexity are interacting and both determine biodiversity and the effectiveness of agri-environmental measures (Tscharntke et al. 2005; Concepción et al. 2008; Smith et al. 2010). Such relationships are demonstrated in Fig. 10.1. It shows a hypothetical linear relationship between land use intensity and biodiversity (solid line in the left chart). Increases in landscape complexity, such as attained by agri-environmental programs (dashed and dotted lines), can shift the curve and/or alter its slope. A parallel shift would reflect a proportional higher but in relative terms a constant impact of landscape complexity on biodiversity. The relative impact of landscape complexity on biodiversity is increasing with land use intensity as shown in (i) or decreasing as shown in (d). Consequently, the rate of species diversity through extensification under a given landscape structure decreases (increases) with increasing (decreasing) landscape complexity, which has been shown among others for arable weed species (Roschewitz et al. 2005). In addition, landscape complexity also determines the relative effectiveness of agri-environmental measures that regulate land use intensity (Fig. 10.1, right chart).

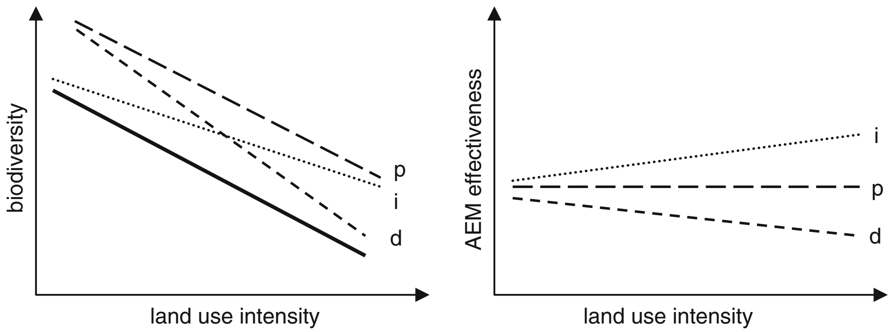


**Fig. 10.1** Hypothetical relationships between biodiversity and land use intensity under lower (*solid line*) and higher (*dashed and dotted lines*) landscape complexities (*left*) and the corresponding effectiveness of agri-environmental measures (AEM) (*right*). Notes: p (land use intensity and landscape complexity are independent), d (impacts of landscape complexity on biodiversity are relatively decreasing with land use intensity), i (impacts of landscape complexity on biodiversity are relatively increasing with land use intensity). (*Source:* Own figure based on Concepción et al. (2008) and Tscharntke et al. (2005))

### *2.2 Biodiversity Assessment in Economic Land Use Optimization Models*

There are several strategies to include biodiversity aspects in economic land use models and we review some contrasting examples. Thereby, we only focus on optimization models due to their importance for ex-ante policy analysis and the methodology applied hereafter.

One way is to directly include biodiversity objectives together with others in a multi-objective function. The challenge here is to find representative preference systems to rank and weight multiple societal objectives either prior to the model application or to the selection among multiple model results (e.g. Groot et al. 2007; Holzkämper and Seppelt 2007). Alternatively, biodiversity maintenance can be directly included with constraints to guarantee minimum provision levels (e.g. van Wenum et al. 2004). The challenge here is to represent minimum provision levels in spatial contexts and to appropriately account for synergies and trade-offs between species and habitats to avoid model solution infeasibilities. Other authors have applied economic land use optimization models for alternative scenarios and have sequentially evaluated scenario results with respect to biodiversity effects (e.g. Brady et al. 2009). Consequently, the corresponding land use and biodiversity effects of predefined policy objectives may only be assessed with multiple model runs.

Any of these methodological options rely on functions between land use and biodiversity either directly or indirectly. Direct functions can portray rather simplistic relationships between biodiversity and single management criteria such as nitrogen application rates and biodiversity (Groot et al. 2007) or dose–response functions of nitrogen deposition (Fraser and Stevens 2008). Münier et al. (2004) have applied a database on ecotopes to assess species diversity, where ecotopes represent homogenous biodiversity response units consisting of bio-physical and land use management characteristics. A frequently applied concept is some kind of species-area relationship that relates the expected number of species to its habitat area (Brady et al. 2009; Nelson et al. 2009). More elaborated approaches combine economic land use models and stand-alone biodiversity models. These models have been developed as simulation models for single species to estimate population developments under changing habitat quality (e.g. Johst et al. 2002; Wätzold et al. 2008), or as regression models based on empirical field data for several species or taxonomic groups (e.g. Gottschalk et al. 2007, 2010; Holzkämper and Seppelt 2007). Indirect or surrogate indicators can replace direct biodiversity functions. They are frequently applied in cases where detailed data on species-habitat relationships are lacking and build on the experiences of empirical case studies from landscape ecology. Both land use intensity and landscape structure may be covered by such indicators (e.g. Pacini et al. 2003; Reidsma et al. 2006).

## 3 Materials and Methods

### *3.1 Overview on the Research Methodology*

Despite the different approaches to integrate biodiversity issues in economic land use optimization models, there are still rather few applications published in the literature. Bio-economic farm models are superior to other methods for the assessment of agricultural systems and the ex-ante evaluation of agri-environmental measures (Janssen and van Ittersum 2007). However, recent reviews reveal a lack in the representation of biotic indicators and identify biodiversity assessments as an important research topic (Janssen and van Ittersum 2007; Rossing et al. 2007; Zander et al. 2008). In addition, biodiversity issues are gaining importance in land use policies such as agri-environmental programs and may stimulate further research demand from stakeholders and administration.

We apply an IMF to assess the impacts of selected agri-environmental measures on biodiversity at field and landscape level. The IMF consists of the farm optimization model FAMOS[space], the crop rotation model CropRota, and the bio-physical process model EPIC (Environmental Policy and Integrated Climate; Williams 1995; Izaurralde et al. 2006). CropRota provides farm specific crop rotations, which are integrated in EPIC together with crop management data and geo-referenced field and climate data to simulate field specific bio-physical impacts. Further details on these two model components, data, and validation are presented in Schönhart et al. (2011a, b). Crop rotations and crop yields are inputs to FAMOS[space], which explicitly considers alternative land use

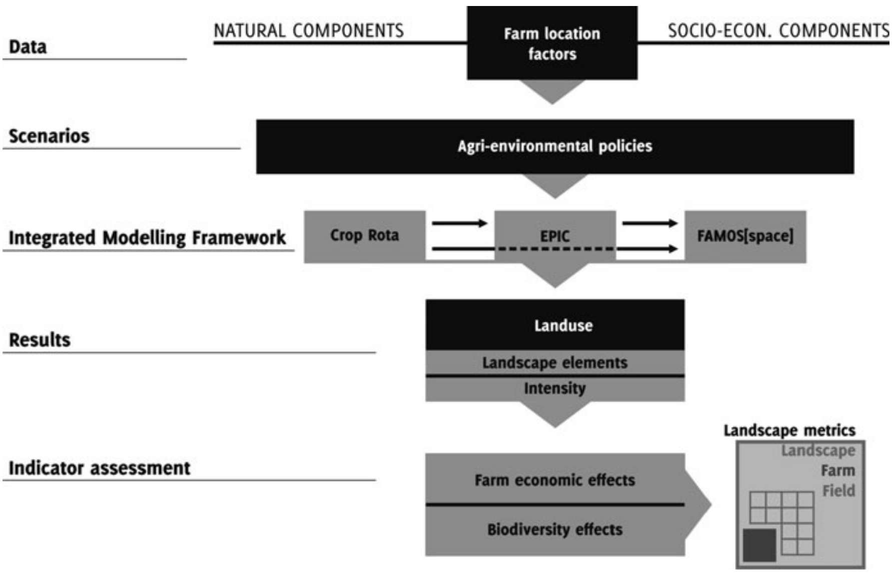


**Fig. 10.2** Overview on the research approach

intensities as well as landscape elements. Biodiversity effects of land use choices are evaluated with a set of field and landscape indicators. Because the composition and configuration of a landscape is influencing ecosystem processes and habitat quality, we apply landscape metrics at the end of the model chain to quantify the spatial biodiversity impacts of landscape development scenarios. Neighborhood metrics are used to analyze the settings of specific landscape elements and their roles as ecological networks. All GIS modeling is done with ESRI® ArcGIS 9.x software package and the vLate Extension (Lang and Tiede 2003) for landscape metrics calculations. Figure 10.2 gives an overview on the research approach.

## *3.2 Land Use Intensity and Landscape Elements in FAMOS[Space]*

FAMOS[space] is developed in GAMS (General Algebraic Modeling System, www.gams.com) and is based on the FAMOS model (Farm Optimization System, Schmid 2004). It has been expanded towards environmental and landscape structure analysis by integrating spatial field contexts. A loop procedure allows for sequential and independent simulations of farms in a landscape. The model is described in detail in Schönhart et al. (2011). Here, we only discuss the representation of land use intensity and landscape elements.

FAMOS[space] is a mixed integer linear farm programming model. It maximizes total farm gross margin (*GROS*) subject to farm specific resource endowments and field properties (farm location factors) by finding optimal production and management activities. Equation 10.1 portrays the objective function in FAMOS[space]. *OPUT* represents farm output variables and *PROD* alternative farm production activities for livestock and land use. Prices, costs, and subsidies are represented by $\rho$, $\chi$, and $v$.

$$\max GROS = \sum (OPUT \cdot \rho_{OPUT}) + \sum (PROD \cdot v_{PROD}) - \sum (PROD \cdot \chi_{PROD}) \tag{10.1}$$

Fields are the spatial decision units in FAMOS[space] and provide the basic structure for all further indicator assessments. The distances of fields to the farmstead, soil quality, size, weather, and slope conditions determine crop production costs and yields. The alternative land use activities (*PROD*) on a field consist of crops and forages as well as landscape elements (hedges and orchard meadows).

Orchard meadows are an ecologically valuable agro-forestry system widespread over central Europe that consists of tall fruit trees dispersed over managed meadows or pastures (Herzog 1998). Fruits and cider from orchard meadows are marketable products. We assume the long-term average fruit price for orchard fruits of 60.7 €/t and average harvest productivities and yields. For further details on the implementation of orchard meadows in FAMOS[space] see Schönhart et al. (2011c). Other landscape elements in FAMOS[space] are hedges. Hedges do not usually provide marketable outputs but a number of societal benefits such as reductions in wind erosion and nutrients leaching as well as provision of nesting and feeding grounds to farmland birds (Hinsley and Bellamy 2000). Furthermore, they are widely acknowledged for their role in connecting habitat patches in fragmented agricultural landscapes (Baudry et al. 2000). Establishment costs for hedges depend on their design, which is related to a purpose, e.g. wind protection or habitat improvement. In Lower Austria, the costs vary between 10,000 €/ha and 20,000 €/ha including maintenance costs during the first years according to the Agrarbezirksbehörde Niederösterreich, a public authority responsible for hedge establishment (personal communication, 8 November 2005). Farmers may be granted subsidies covering up to 90% of these establishment costs. Roth und Berger (1999) estimated establishment costs of about 9,000 €/ha for smaller hedges to increase habitat quality. In our analysis, we assume costs of 12,000 €/ha including maintenance and do not consider any establishment subsidies. The hedges as well as orchards are assumed to remain for a 30-years period. Annuities have been calculated using a discount rate of 5%. Transitions from cropland to grassland and vice versa seem unlikely and are not considered in the model, because forage production activities are available for cropland anyway and permanent grassland conversions to cropland are prohibited by cross compliance legislation. Transitions between landscape elements and other land uses are possible on pre-defined sites, which have been identified from historical surveys in the case of orchard meadows and assumed to improve networks in the case of hedges.

Land use intensity in FAMOS[space] is considered by crop rotation choices, nutrient application rates (N, P, K) as well as mowing frequencies. The model can

**Table 10.1** Management intensities on permanent grassland and cropland in terms of nitrogen (N) application rates (kg/ha), mowing frequencies (cuts/a), and hemerobic state

| | Land use intensity | | | | | | Hemerobic state | | |
|---|---|---|---|---|---|---|---|---|---|
| | High intensity (HI) | | Medium intensity (MI) | | Low intensity (LI) | | | | |
| Land use activity | *N rates* | *cuts/a* | *N rates* | *cuts/a* | *N rates* | *cuts/a* | HI | MI | LI |
| Permanent mead. | 190 | 4 | 120 | 3 | 70 | 3 | 2 | 2.5 | 3 |
| Orchard meadows | 190 | 3 | 120 | 3 | 70 | 3 | 2 | 2.5 | 3 |
| Extensive (orchard) meadows | 50 | 1 | 30 | 1 | 20 | 1 | 4 | 4 | 4 |
| Temporary grassland | 210 | 4 | 150 | 3 | 120 | 3 | 1 | 1.5 | 2 |
| Clover-grassland | 40 | 4 | 20 | 3 | 0 | 3 | 1 | 1.5 | 2 |
| Red clover | 0 | 4 | 0 | 3 | 0 | 3 | 1 | 1.5 | 2 |
| Alfalfa | 0 | 4 | 0 | 3 | 0 | 3 | 1 | 1.5 | 2 |
| Winter wheat | 165 | – | 130 | – | 90 | – | 1 | 1.5 | 2 |
| Winter barley | 150 | – | 120 | – | 80 | – | 1 | 1.5 | 2 |
| Triticale | 130 | – | 110 | – | 80 | – | 1 | 1.5 | 2 |
| Rye | 120 | – | 100 | – | 70 | – | 1 | 1.5 | 2 |
| Oats | 110 | – | 90 | – | 70 | – | 1 | 1.5 | 2 |
| Summer barley | 100 | – | 85 | – | 60 | – | 1 | 1.5 | 2 |
| Grain corn | 175 | – | 140 | – | 110 | – | 1 | 1.5 | 2 |
| Silage corn | 175 | – | 150 | – | 120 | – | 1 | 1.5 | 2 |
| Sugar beet | 140 | – | 110 | – | 80 | – | 1 | 1.5 | 2 |
| Rapeseeds | 175 | – | 140 | – | 110 | – | 1 | 1.5 | 2 |
| Sunflower | 80 | – | 60 | – | 50 | – | 1 | 1.5 | 2 |
| Soybeans | 0 | – | 0 | – | 0 | – | 1 | 1.5 | 2 |
| Field bean | 0 | – | 0 | – | 0 | – | 1 | 1.5 | 2 |
| Field peas | 0 | – | 0 | – | 0 | – | 1 | 1.5 | 2 |
| Fallow | 0 | – | 0 | – | 0 | – | 3 | 3 | 3 |
| Hedges | – | – | – | – | – | – | 4 | 4 | 4 |

*Source:* Own table, hemerobic states based on Zechmeister et al. (2002) and Zechmeister and Moser (2001) [p (poly-; 1), a (a-eu-; 2), b (b-eu-; 3), m (meso-; 4)]

choose among four intensity levels – high intensity (HI), medium intensity (MI), low intensity (LI), and organic farming (Table 10.1). Organic farming is not considered in this analysis, because no organic farms have been reported for the case study landscape in the farm survey data.

Agri-environmental measures to maintain and establish landscape elements and to reduce land use intensity impact farm production choices. For example, maintenance and establishment of landscape elements usually increase direct and opportunity land use costs but may provide market and non-market benefits to farmers and the society. Table 10.2 lists the two agri-environmental measures applied in this study and describes how they influence the variables and parameters in FAMOS[space]. Some of these effects such as natural pest control may be important (cf. Gardiner et al. 2009) though difficult to quantify and are therefore not included in the analysis.

**Table 10.2** Assumed impacts of agri-environmental measures and their consideration in FAMOS [space]

| Agri-environmental measure | Impacts and consideration in FAMOS[space] | |
|---|---|---|
| Landscape elements – maintenance and establishment | Reduced crop and forage yields and available land for crop and forage production [field edge effect from hedges n.c.] | ↓ $OPUT_{n\ LE}$ |
| | Loss of direct payments (e.g. single farm payment, less favored area payments) | ↓ $\nu_{n\ LE}$ |
| | Increased production costs through reduced field sizes and adverse mechanization | ↑ $\chi_{\mathrm{n\ LE}}$ |
| | Increased labor requirements through LE maintenance | ↑ $\chi_{LE}$ |
| | Direct costs through LE establishment and maintenance | ↑ $\chi_{LE}$ |
| | More natural pest control, additional pest infestation | ↓↑ $\chi_{n\ LE}$, [n.c.] |
| | Market benefits from fruit harvest | ↑ $OPUT_{LE}$ |
| Land use intensity – reduction | Reduced quantity and quality of crop and forage yields | ↓ $OPUT_{n\ LE}$ |
| | Higher product prices, e.g. organic farming | ↑ $\rho_{LE}$ , ↑ $\rho_{n\ LE}$ [n.c.] |
| | Reduced fertilization costs | ↓ $\chi_{n\ LE}$ |
| | Reduced mechanization costs through decreased mowing frequencies | ↓ $\chi_{n\ LE}$ |

Legend: *n.c.* not considered in this study, *LE* landscape elements, *n LE* non landscape elements, arrows indicate an increasing (↑) or decreasing (↓) effect in the model, *OPUT* farm products, $\chi$ cost coefficients, $\rho$ price coefficients, ν subsidy coefficients

## *3.3 Landscape Data and Indicator Selection*

The IMF operates on a high level of detail with respect to field, farm, and landscape location factors. Consequently, it requires farm resource and landscape element data from field to landscape levels (for a description of the data sources see Schönhart et al. 2011b). Besides the common set of economic and farm resource data, high resolution field data are of crucial importance as well. They are extracted from the geo-referenced IACS (Integrated Administration and Control System) database and merged with other thematic IACS and statistical data sources. Instead of applying the concept of artificial landscapes (cf. Brady et al. 2009), actual fields have been integrated as polygons to portray the landscape as detailed as possible with respect to their production and ecological functions. Field data are complemented by landscape element data to derive current and potential sites for landscape element establishments. Maps on landscape elements have been generated by a semi-automated segregation process based on ortho- and aerial

**Table 10.3** Overview on the type and measurement of biodiversity indicators

| | | | Normalization range | |
|---|---|---|---|---|
| Spatial level | Indicator | Description | Min | Max |
| *Intra-patch* | Habitat value | Mean hemerobic state | 1 | 5 |
| | Nitrogen use intensity | Mean nitrogen application rate (kg/ha) | 190 | 0 |
| | Mowing frequency | Mean mowing frequency of permanent grassland (cuts/a) | 4 | 1 |
| *Matrix* | Landscape diversity | Shannon diversity index (SDI) $SDI = -\sum_{i}^{s}[(PROD_i/PROD_I) \cdot \ln(PROD_i/PROD_I)]$ | 0 | $SDI_{\max} = \ln S$ |
| | Patch number | Total number (TP) of different land use patches | 273 | 1,092 |
| | Patch size | Mean size of different land use patches (MPS) (ha) | 2 | 0.5 |
| | Edge length | Total length of edges (TE) between landscape elements and grassland or cropland (km) | 0 | 98.5 |
| | Habitat connectivity | Total area with a distance $> 50$ m from landscape elements (ha) | $PROD_I$ | 0 |

Notes: All indicators are analyzed at the landscape level. The normalization range refers to an assumed effect on biodiversity, where min is the lowest level with negative or zero effects and max the highest possible value with positive effects on biodiversity. $PROD_i$ refers to the area of a land use activity $i$ and $PROD_I$ to the area sum over all land use activities. $S$ is the number of different $i$

photos (cf. Schauppenlehner et al. 2010; Schönhart et al. 2011), from which potential sites have been drawn considering landscape planning criteria.

We apply a broad set of surrogate indicators to indicate the biodiversity effects from alternative land uses. Their choice has been guided by empirical studies on the relationship between habitat quality and biodiversity. Indicators include an intra-patch dimension at the field level and a matrix dimension at the landscape level (Dauber et al. 2003). Only agricultural land use is considered, while all other land uses such as infrastructure or forests are kept constant and are not included. For reasons of simplicity, we do not consider natural site conditions and the land cover/use history as biodiversity drivers, although such aspects can be included in the IMF.

Field level intra-patch indicators, such as habitat type and land use intensity describe field management effects (Table 10.3). Habitat type is based on the concept of hemeroby, which is an indicator for the naturalness of habitats and frequently applied in empirical and model-based biodiversity assessments (Zechmeister and Moser 2001; Zechmeister et al. 2002, 2003b; Zebisch et al. 2004; Schreiber 2010). Agricultural land use activities from FAMOS[space] are numerically classified according to the hemerobic states p (poly-; 1), a (a-eu-; 2), b (b-eu-; 3), and m (mesohemerob; 4) as presented in Zechmeister et al. (2002)

and Zechmeister and Moser (2001) (Table 10.1) and aggregated to the landscape level. Nitrogen application rates can serve as important biodiversity indicator (Zechmeister et al. 2003a; Schmitzberger et al. 2005; Kleijn et al. 2009). It is complemented by mowing frequencies of permanent grasslands (Zechmeister et al. 2003a) to describe land use intensity at the field level.

At the landscape level, matrix indicators based on landscape metrics describe the extent, composition, and spatial configuration of different habitats (Bennett et al. 2006). 'Extent' relates to the total area of habitat types in a landscape and is approximated already by the intra-patch indicator for habitat quality. The 'mosaic concept' in landscape ecology (cf. Duelli 1997) pronounces landscape composition and configuration. Composition or habitat variability refers to the number (richness) and relative areas (evenness) of habitats in a landscape (Duelli 1997; Bennett et al. 2006), which both can be expressed by the Shannon diversity index (SDI) (cf. Gottschalk et al. 2007, 2010; Brady et al. 2009). SDI-categories are the cropland and grassland activities as well as landscape elements according to Table 10.1. Two other indicators for landscape composition are the total number of patches (NP) and the mean patch size (MPS). However, composition does not sufficiently describe the spatial configuration of habitats in the landscape, which is important among others to describe network characteristics of a landscape. In this analysis, habitat configuration is indicated by the total length of patch edges (TE) between the three major land use categories cropland, grassland, and landscape elements (orchard meadows, hedges). For instance, edge length is an indicator for plant species diversity on grasslands (Marini et al. 2008). Furthermore, we assess the network of landscape elements as it can be important for example to habitat specialists and larger mammals (cf. Steffan-Dewenter 2003; Pereira and Rodríguez 2010). Therefore, we sum the area with a distance of more than 50 m from the next landscape element as an indicator for the distribution of landscape elements in a landscape.

Intra-patch and matrix indicators differ by the spatial level of indicator application – either at single fields, subfields, or the landscape. However, model results on biodiversity effects are only presented at the landscape level. To increase comparability among scenarios, we normalize the indicator values as presented by the assumed ranges in Table 10.3, where the minimum (min) represents the lowest possible value and the maximum (max) the ecologically most favorable value. The hemerobic states range between 1 and 5 (Zechmeister and Moser 2001; Zechmeister et al. 2002). Ecologically sound mowing frequency is assumed to 1 cut/a and nitrogen application rates to 0 kg/ha for all lands, although there may be differences between cropland and grassland (cf. Zechmeister et al. 2003a; Schmitzberger et al. 2005). The upper limits are set to the highest possible nitrogen application rates and mowing frequencies in the model. For the SDI, the lowest value is 0, while it is limited by the total number of alternative land use activities $i$. The best level for MPS is assumed to 0.5 ha, although a judgment on appropriate field sizes from an ecological perspective seems difficult and little empirical literature on this issue is available. The upper bound of MPS is set to 2 ha, which is assumed as threshold for efficient mechanization. Beyond it, rationalization gains may not be substantial enough to justify further increases (Rodríguez and Wiegand 2009).

Minimum and maximum levels of NP are derived from the total modeled farmland in the case study area and the assumptions on MPS. The minimum for TE is 0 in a landscape without landscape elements. TE maximum is equal to the maximum value of TE attained for all three land use categories (cropland, grassland, landscape elements), as this is assumed to be a proxy for the total possible edge length in the region given the current distribution of potential sites for landscape elements. The minimum habitat connectivity equals the total area, while the maximum is reached if no area is farther than 50 m from a landscape element.

### 3.4 Biodiversity Data and Sensitivity Analysis

The surrogate biodiversity indicators are supplemented by functions between selected indicators and plant species diversity, as the latter is seen as useful indicator for overall species richness (Sauberer et al. 2004). Data on plant species diversity are extracted from published field study data from Austria. Schmitzberger et al. (2005) investigated cropland at different locations in Austria and relate nitrogen application rates to arable weed diversity. Zechmeister et al. (2003a) correlate total plant species richness (vascular and bryophyte plants) in grasslands based on data from Austrian-wide samples. Furthermore, the scenario values for habitat quality (hemeroby) of the landscape are correlated to the species number of bryophyte plants based on an Austrian-wide assessment (cf. Zechmeister and Moser 2001). Due to similar climatic and land use conditions, we assume that all three studies are an appropriate approximation for relative changes in biodiversity depending on different management intensities. We have translated absolute values to relative changes to reduce biases from varying site conditions. A site is assumed to reach its maximum in species diversity with a hemerobic state of five and a rate of 15 kg/ha nitrogen fertilizer application on grassland and 0 kg/ha on cropland.

Landscape complexity and land use intensity can interact at the landscape level, which may also determine the effectiveness of agri-environmental programs (compare to Sect. 2 and Fig. 10.1). The possibilities for functional relationships are numerous and are a potential source of uncertainty. Hence, we apply a sensitivity analysis to show the impact of different functional relationships discussed in Sect. 2.1. We assume three hypothetical linear functional relationships based on the results of Schmitzberger et al. (2005) and Zechmeister et al. (2003a) and analyze the effects of nitrogen application rates (kg/ha) and landscape complexity (SDI) on relative plant species diversity. In all three functional forms, landscape complexity is assumed to be effective between the lowest SDI value and the largest possible in the landscape. The SDI value either increases the upper (at 0 and 15 N kg/ha) or lower level (at 150 N kg/ha) of the relative plant species diversity. Table 10.4 lists the different functional forms of the sensitivity analysis. For example, gl_i_0.5 is a functional relationship of type (i) for grassland (gl), i.e. landscape complexity is assumed to be more effective on biodiversity at higher land use intensities. In the scenario, the relative plant species diversity is increased by

**Table 10.4** Sensitivity analysis on functional forms between land use intensity, landscape complexity and biodiversity

| Functional relationships | Change of relative biodiversity value (percentage points) | Nitrogen application rate (kg/ha) |
|---|---|---|
| gl_i_0.5 | 50 | 150 |
| gl_i_1.0 | 100 | 150 |
| gl_d_0.5 | 50 | 15 |
| gl_d_1.0 | 100 | 15 |
| gl_p_0.5 | 50 | 15 and 150 |
| gl_p_1.0 | 100 | 15 and 150 |
| cl_i_0.5 | 50 | 150 |
| cl_i_1.0 | 100 | 150 |
| cl_d_0.5 | 50 | 0 |
| cl_d_1.0 | 100 | 0 |
| cl_p_0.5 | 50 | 0 and 150 |
| cl_p_1.0 | 100 | 0 and 150 |

Legend: *gl* grassland, *cl* cropland; functional relationships: *p* land use intensity and landscape complexity are independent, *d* impacts of landscape complexity on biodiversity are relatively decreasing with land use intensity, *i* impacts of landscape complexity on biodiversity are relatively increasing with land use intensity

50 percentage points at high land use intensities (150 N kg/ha) and a normalized SDI value of 1, while it remains unchanged at low intensities (15 N kg/ha) with the lowest normalized SDI of 0.53, which occurred in the reference scenario (cf. Sect. 4).

## 4 Case Study Landscape and Model Scenario Descriptions

The IMF is applied to a landscape in the Lower Austrian 'Mostviertel' region, which is characterized by a rather homogenous northern part with respect to landscape structure and relief and a southern part that features the traditional landscape element of the 'Mostviertel' region, namely orchard meadows on gentle hills. We model 20 conventionally producing farms specialized in cash crop or livestock production or a mixture of both. The farms manage about 430 agricultural fields with 546 ha in total, of which are 399 ha cropland and about 147 ha permanent grassland. We have chosen a smaller portion of adjacent fields out of the total modeled farmland for the designation of potential landscape element sites due to data limitations. Fields outside are assumed to have neither existing nor potential landscape element sites.

In our case study analysis, we assess the joint effects of landscape structure and land use intensity as a consequence of agri-environmental measures. We have developed a reference scenario (REF) and an agri-environmental policy scenario with different measures (S1-S6). The latter introduces agri-environmental measures with alternative levels of land use intensities and landscape elements (Table 10.5),

**Table 10.5** Overview on the case study scenarios

| | Description | |
|---|---|---|
| Scenario | *Landscape elements* | *Land use intensity* |
| *REF* | No intervention | No intervention (nitrate directive binding) |
| *S1* | No removal of existing sites | No intervention (nitrate directive binding) |
| *S2* | No removal of existing sites, at least 50% of potentially available sites on each farm | Low or medium intensity |
| *S3* | 100% of potentially available sites on each farm | Low or medium intensity |
| *S4* | 100% of potentially available sites on each farm | Low intensity |
| *S5* | 100% of potentially available sites on each farm | Low intensity, at least 25% extensive grassland |
| *S6* | 100% of potentially available sites on each farm | Low intensity, at least 75% extensive grassland |

which are seen as important to maintain farmland biodiversity such as farmland birds (Tucker 1997). Landscape elements such as hedges and orchard meadows can be grown on existing sites or may be established on new sites, which both sum up to the potentially available sites.

Both, hedges and orchard meadows are considered as valuable semi-natural elements for habitat and biodiversity provisioning in rather intensively managed grassland landscapes of Austria to which the case study landscape belongs to (Wrbka et al. 2005). In the case study landscape, 1.8 ha orchard meadows but no hedges are currently cultivated. New orchard meadows can be established in the model on historical orchard meadows land, which amounts to 4.1 ha (cf. Schönhart et al. 2011). The establishment of hedges is often regarded to increase the ecological value of a landscape while simultaneously allowing profitable agricultural land use (Briemle et al. 2000). In landscapes with a high share of orchard meadows, hedges increase the network among frequently fragmented orchard meadows patches (Weller 2006), while species in hedges such as birds may benefit from the vicinity of extensively used grasslands as feeding grounds (Herzog et al. 2005). We have identified potentially available sites for hedges along field edges and in the case of large fields throughout fields according to their proximity to other semi-natural areas such as forests and orchard meadows. The hedge width is set to 3 m and can double where farmers establish hedges at the same field boundary. The ecologically effective distance criterion between landscape elements is assumed to be 50 m (cf. Herzog et al. 2005). This leads to a total hedge area of 3.3 ha, which sums up to a total landscape elements area of 9.2 ha or 1.7%.

## 5 Results

The main results of our case study analysis with respect to the biodiversity indicators are presented in Table 10.6. Without policy interventions, the average nitrogen application rate among all farms is 145 kg/ha, which is below the

**Table 10.6** Average landscape indicator results and percentage changes in total farm gross margin ($GROS_{landscape}$) from the reference scenario (REF)

| Scenario | Indicator (average value for the landscape) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | *Habitat value (hemerobic state)* | *Nitrogen use intensity (kg/ha)* | *Mowing frequency (cuts/a)* | *Landscape diversity [SDI]* | *Patch number [NP]* | *Patch size [MPS] (ha)* | *Edge length [TE] (km)* | *Habitat connectivity (ha)* | *$GROS_{landscape}$ reduction (% from REF)* |
| *REF* | 1.5 | 145 | 3.0 | 1.51 | 464 | 1.18 | 0 | 546 | – |
| *S1* | 1.5 | 144 | 3.0 | 1.53 | 497 | 1.10 | 4.6 | 513 | 0.1 |
| *S2* | 1.8 | 124 | 3.0 | 1.72 | 534 | 1.02 | 23.0 | 411 | 1.8 |
| *S3* | 1.8 | 123 | 3.0 | 1.75 | 560 | 0.98 | 38.6 | 343 | 2.3 |
| *S4* | 2.3 | 84 | 3.0 | 1.92 | 556 | 0.98 | 38.6 | 343 | 15.1 |
| *S5* | 2.3 | 82 | 2.5 | 2.05 | 590 | 0.92 | 38.5 | 344 | 17.5 |
| *S6* | 2.4 | 77 | 1.5 | 2.06 | 586 | 0.93 | 38.5 | 345 | 23.5 |

Note: For a description of the indicators see Table 10.3

maximum levels permitted by the nitrate directive. In the reference scenario (REF) all orchard meadows are removed due to their high costs, while neither new orchard meadows nor hedges are established. This results in the lowest values for SDI and NP, and the largest for MPS. The introduction of an agri-environmental measure to promote landscape element maintenance in scenario S1 has only minor effects on most indicators due to the small share of existing orchard meadows in relation to the total farmland (0.3%). However, effects on individual farms can be economically important as farm gross margins decrease by 280 €/ha of orchard meadows on average. The establishment of additional landscape elements and medium to low land use intensities (MI, LI) in S2 and S3 lead to decreasing average nitrogen application rates mainly on cropland (Fig. 10.5a). The average hemerobic state as well as SDI and NP increase and MPS decreases, which indicates a more heterogeneous landscape with moderately lower total farm gross margins over all farms ($GROS_{landscape}$) of 2.3% in S3 compared to REF. Direct and opportunity costs increase with further reductions in land use intensity (LI) in S4 and reduce $GROS_{landscape}$ by 12.8 percentage points compared to S3. From S4 onwards, a small share of agricultural land becomes abandoned. The introduction of minimum extensive grassland areas mown only once a year (25% of all permanent grassland in S5 and 75% in S6) further reduce land use intensity to average nitrogen application rates of 77 kg/ha in S5. In the model, farms partially compensate the forage yield losses in quantity and quality by cultivating temporary grassland on their croplands and by forage purchases. $GROS_{landscape}$ in S6 is 23.5% below REF, which can even be up to 42% for single farms. In S6, all potentially available sites are covered by landscape elements, the land use intensity is reduced to the low level (LI) and 75% of the permanent grassland is extensified. Consequently, landscape heterogeneity further increases with an SDI in S6 of 73% of the maximum possible value compared to 53% in REF. Figure 10.3 presents normalized indicator values and Fig. 10.4 maps for the scenarios REF and S6. All indicators in Fig. 10.3 show increasing trends from REF to S6. Some values such as the mowing frequency are closer to the assumed maximum values, others such as the hemerobic state remain rather low. Higher values for this indicator may be achieved by a conversion from cropland to extensive grasslands or semi-natural landscape elements, which likely would have further consequences for total farm gross margins.

Figure 10.5b and c correlate nitrogen application rates and the hemerobic value with the reductions in $GROS_{landscape}$ and the relative biodiversity changes of plants (cf. Sect. 3.4). According to Fig. 10.5b, plant species on cropland increase from about 10% in REF to 50% in S6 and grassland species from 60% to 90% as a consequence of grassland extensification. From a hemeroby perspective, changes in species number show similar magnitudes resulting from extensification and landscape element creation if the higher share of cropland and therefore its higher weight compared to grassland is acknowledged. Clearly, Fig. 10.5b and c cannot be simply aggregated as hemeroby among others is a function of nitrogen fertilizer intensity.

The results of the sensitivity analysis are presented in Fig. 10.6 (i), (d), and (p) (cf. Fig. 10.1, Sect. 3.4 and Table 10.4). The figures show the influence of both, the

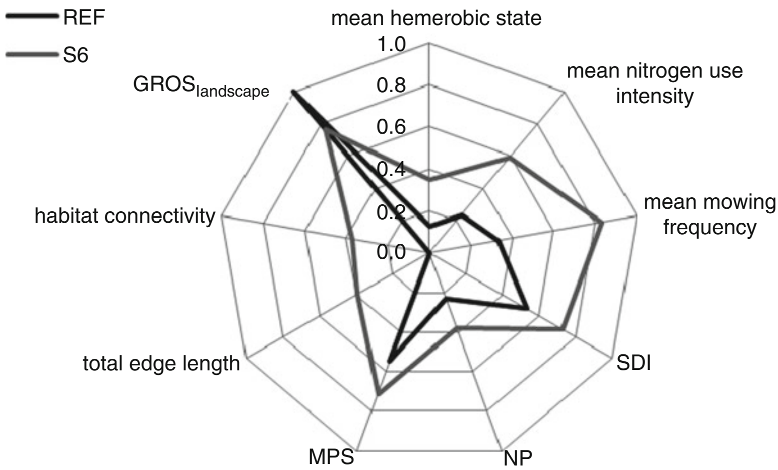


**Fig. 10.3** Normalized intra-patch and matrix indicator values for the reference scenario (REF) and the agri-environmental policy scenario S6. Legend: *MPS* mean patch size, *NP* number of patches, *SDI* Shannon diversity index, $GROS_{landscape}$ sum of total farm gross margins

shape of the hypothetical functional relationships as well as the assumed quantitative influence of the SDI on the plant species developments on grassland and cropland. Despite their different shapes (cf. Fig. 10.1), all three functional relationships cause similar effects on relative plant species diversity due to the simultaneous changes of land use intensity and landscape complexity in the scenarios. In general, the sensitivity of landscape complexity is lower at higher land use intensities and therefore becomes more important during extensification. The highest changes of relative plant species diversity are observed for the parallel shift (p) of 100 percentage points at low and high land use intensities (cl_p_1.0, gl_p_1.0) and nearly double relative plant species diversity on cropland and increase grassland values by 50%.

# 6 Discussion

## 6.1 *Agri-Environmental Policy Implications of the Case Study Results*

Farm economic and biodiversity effects of agri-environmental measures have been assessed in an integrated farmland use modeling framework (IMF). The implemented measures represent rather strong limitations on land use compared to the current situation. For example, scenario S6 forces farms to a production intensity comparable to organic farming, to the maintenance and establishment of landscape elements on all potentially available sites in the case study landscape


**Fig. 10.4** The landscapes for the reference scenario (REF, left) and the agri-environmental policy scenario S6 (right) (© BEV 2002, ZI. 6843/2002. © Landesregierungen und Land-, forst, und wasserwirtschaftliches Rechenzentrum GmbH. (*Sources*: Own drawing with data from BMLFUW (2008))

(Fig. 10.4), and to extensification of 75% of the permanent grassland to one-cut meadows. Birdlife (2009) proposes a 10%-standard of farmland that should be mainly managed for biodiversity conservation. Such value is approximated in the scenarios S5 and S6 dedicating 7.3% and 20.4% of total farmland for nature conservation (landscape elements, extensive meadows), respectively. However, these shares seem rather high considering the already available forest patches and other natural vegetation in the case study landscape. Model results show declining total farm gross margins of up to 25% on average with single farms facing even

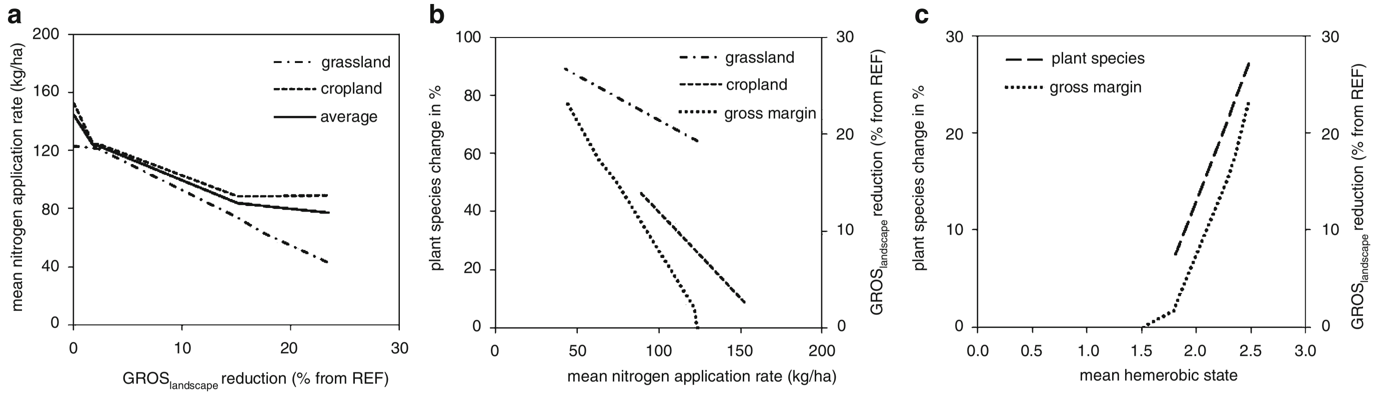


**Fig. 10.5** (**a**) Trade-off curves between total farm gross margin ($GROS_{landscape}$) %-changes from the reference scenario (REF) and mean nitrogen application rates (kg/ha); (**b**) correlation between mean nitrogen application rates (kg/ha) and total farm gross margin ($GROS_{landscape}$) %-changes from the reference scenario (REF) as well as %-changes of species richness of vascular plants (cropland) and vascular and bryophyte plants (grassland); (**c**) correlation between mean hemerobic state and total farm gross margin ($GROS_{landscape}$) %-changes from the reference scenario (REF) as well as %-changes of species richness of relative bryophyte plants (Sources: Own figures, (**b**) based on Schmitzberger et al. (2005) and Zechmeister et al. (2003a), (**c**) based on Zechmeister and Moser (2001))

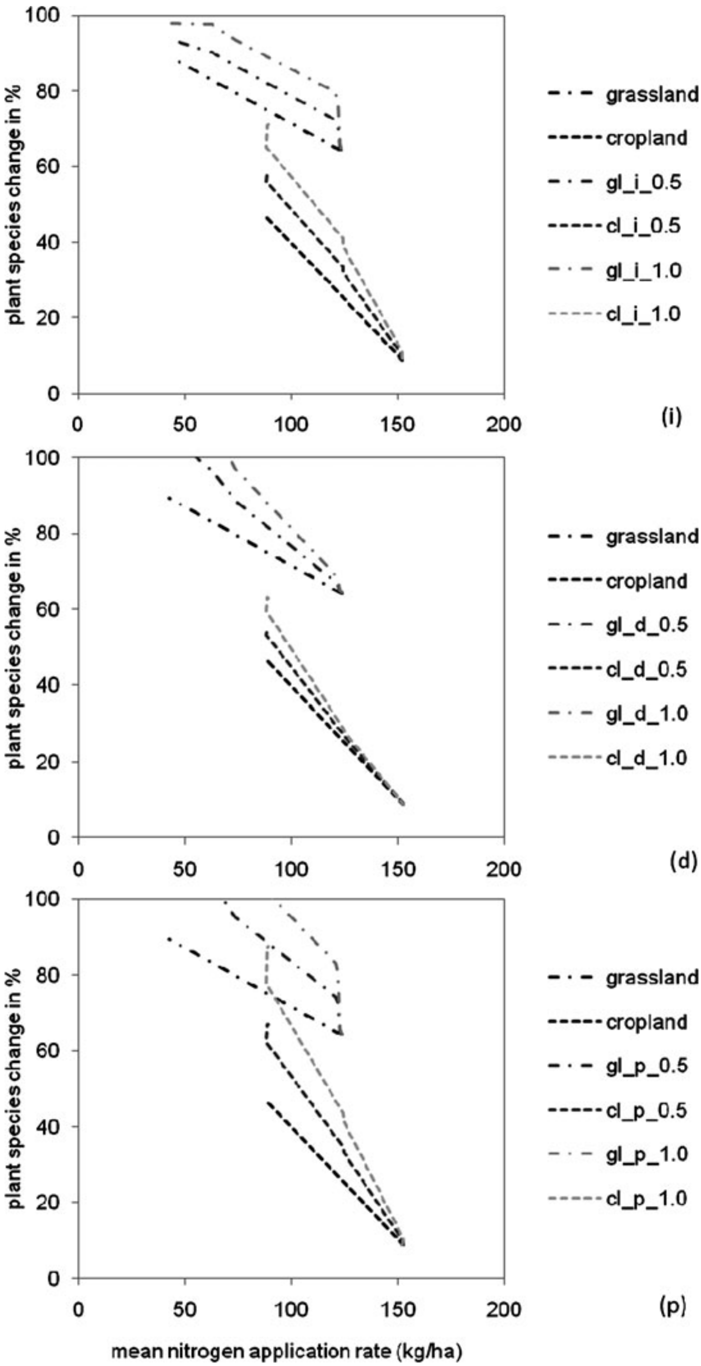


**Fig. 10.6** Sensitivity analysis results for hypothetical correlations between nitrogen application rates (kg/ha), landscape complexity (SDI) and relative plant species richness (%). Notes: *gl* grassland, *cl* cropland; i, d, p, functional forms according to Fig. 10.1 and Table 10.5

higher reductions. These results take into account historical land use and livestock choices. Therefore, if farmers have already applied agri-environmental measures in the past, FAMOS[space] may underestimate the full intensification potential and opportunity costs. In the model, farms can compensate forage yield losses by purchases or forage production on cropland. We may also underestimate opportunity costs, because agri-environmental measures implemented on a larger scale likely reduce the regional supply of marketed forage and increase its price, which is currently assumed constant in the model. On the other side, products from extensive land use systems may gain higher market prices, which are also not considered in FAMOS[space].

A negative relationship between biodiversity and gross margins per ha has also been shown by other empirical studies (Zechmeister et al. 2003a; Schmitzberger et al. 2005). The absence of agri-environmental measures likely leads to a loss of semi-natural landscape elements such as orchard meadows and hedges as well as to farmland intensification. Pascual and Perrings (2007) highlight the need to correct for market failures in order to reduce the disinvestments in farmland biodiversity. Empirical findings indicate that well structured agricultural landscapes of high ecological value are appreciated by the society (cf. Lindemann-Matthies et al. 2010) and agri-environmental measures have been implemented to reward farmers for maintaining heterogeneous landscapes and to reduce land abandonment and intensification. However, premiums seem insufficient to maintain HNV farmland at a European scale and even in Austria, where the support for HNV farmland is higher than in other European countries (EEA 2009). For example, decreasing areas of hedges in grassland landscapes as well as extensive orchard meadows have been observed (Pötsch et al. 2009; Schönhart et al. 2011c) and empirical studies could not confirm a major influence of the Austrian agri-environmental program ÖPUL on the development of landscape elements in selected agricultural landscapes (Bartel 2006). The current ÖPUL premiums granted for orchard meadows of 120 €/ha are below the modeled average reductions in total farm gross margins of about 280 €/ha and would be insufficient to maintain current levels of landscape elements. Hedges are currently protected from removal in ÖPUL but do receive only limited agri-environmental premiums and neither single farm payments, nor less favored area payments. Low financial rewards to hedges have been reported for Switzerland as well (Herzog et al. 2005). Situations like this can create further disincentives for landscape element maintenance (cf. Birdlife 2009). Technological progress in agriculture towards larger machinery can be beneficial for the protection of abiotic resources, but it may even increase the opportunity costs of landscape elements in the long run (Heißenhuber 1999; Kapfer et al. 2003). Examples on how to bridge this gap are the subsidies provided for hedge establishments in Lower Austria or the 'Ecopoints Program', which is a sub-program of ÖPUL. It promotes heterogeneous farmland by subsidizing landscape elements maintenance.

## 6.2 A Critical Note on the Interpretation of Biodiversity Results

Besides basic farm model assumptions such as constrained farm profit maximization, other assumptions have been made on the relationship between land use management and biodiversity. We followed a rather European perspective and see agriculture as potential supplier of biodiversity and pleasant landscapes subject to appropriate land management (cf. Tscharntke et al. 2005). However, there is a second perspective in landscape ecology that underlines the role of undisturbed land for nature protection. Its proponents argue that intensification in some regions may spare land in others for conservation purposes (Green et al. 2005; Polasky and Vossler 2006). There seems to be no final answer on the superiority of one of these two strategies over the other so far (Pain and Pienkowski 1997; Tucker 1997), because it may depend on the detailed objectives of biodiversity and habitat protection as well as on local contexts and framework conditions such as the demand for agricultural products under population growth. Furthermore, it may also depend on the question whether or not it is possible to develop intensive agricultural systems with lower environmental impacts (e.g. precision farming).

Context sensitivity also relates to the assumed relationships between land use management and biodiversity. In the case of species diversity and nitrogen application rates (Fig. 10.5b), we assume a linear relationship although there are empirical evidences for non-linear relationships as well (Kleijn et al. 2009). Furthermore, one has to stress contradicting empirical studies about the effectiveness of agri-environmental measures on biodiversity maintenance (Kleijn et al. 2001) and the importance of local or site-specific conditions as well as the species to be protected. Heterogeneous landscapes are not favorable to all species (Filippi-Codaccioni et al. 2010), which highlights the need for clear objectives prior to any policy implementation and evaluation.

The complex nature of biodiversity in agricultural landscapes calls for a rich indicator set instead of single indicators (Duelli and Obrist 2003). We are aware of this complexity and therefore evaluate land use results from FAMOS[space] with a rich surrogate indicator set. The correlations are based on Austrian case studies and expressed in relative rather than absolute terms. Furthermore, we apply sensitivity analysis to show potential impacts of a changing landscape complexity (SDI) on the correlation of land use intensity and relative plant species richness. The sensitivity is drawn on hypothetical relationships from landscape ecology literature (Tscharntke et al. 2005; Concepción et al. 2008). Surrogate indicators are criticized for their limited explanatory power (Clergue et al. 2005) and further research is necessary to improve both, the validity of intra-patch as well as matrix indicators as proxies for biodiversity. This includes knowledge on the interactions between both levels (cf. Concepción et al. 2008), which may determine the effectiveness of agri-environmental measures especially in already heterogeneous landscapes such as the case study landscape. Such interactions have been assessed by the sensitivity analysis. It shows that the interference of landscape complexity on biodiversity is relevant for results interpretation and reveals the substantial uncertainties related to the effects of agri-environmental measures concerning biodiversity. Although hypothetical in its nature, the sensitivity analysis gives an impression on the magnitude of interaction and emphasizes the importance of further research. Functional relationships like the ones presented can be used to better target agri-environmental measures.

## 6.3 Methodological Considerations on Integrated Farm Land Use Modeling and Biodiversity

There are several methodological challenges related to integrated land use optimization models at landscape levels such as model evaluation, data availability, the trade-offs between model complexity, size and dynamics, and the linkages to disciplinary knowledge (Schönhart et al. 2011b). The IMF requires high resolution landscape data, which becomes increasingly available as a consequence of agricultural policy administration (e.g. IACS), improved remote sensing technologies and the implementation of GIS to collect, process, and store data. Nevertheless, more efforts are necessary to collect landscape data on habitat types and qualities as well as natural site conditions (e.g. soils) in sufficient quality and resolution. The increasing availability of powerful computers supports the application, processing, and integration of large datasets. It also alleviates the trade-offs between spatial scales and model complexity (Seppelt and Voinov 2002). For example, the IMF has employed grid-computing, which has reduced the model solving time by a factor of 20 compared to single core computing on a standard PC.

There is increasing demand for collaborative research between different disciplines to better assess the relationships between farm decision making, agri-environmental measures, land use, and farmland biodiversity at the landscape level (Opdam and Wascher 2004; Pascual and Perrings 2007; Smith et al. 2010). Bio-economic farmland use models can act as interdisciplinary tools for knowledge integration on biodiversity because they are able to provide the necessary interfaces to landscape ecology and estimate field and farm specific opportunity costs of alternative land use management choices. The latter is achieved in our IMF by the integration of field specific crop yields, which have been simulated with the bio-physical process model EPIC. Crop rotations are integral to sustainable agricultural systems, which have been generated by CropRota for each farm. The IMF allows to jointly consider important land use effects such as on biodiversity on a field and landscape level and to assess the cost-effectiveness of agri-environmental measures or landscape planning strategies such as the design of environmental networks for biodiversity enhancement (Dutton et al. 2008; Nassauer and Opdam 2008). In contrast to some approaches presented in Sect. 2.2, we evaluated biodiversity effects subsequent to the modeling process, which allowed us to apply a rich indicator set and empirical functions on biodiversity and land use including sensitivity analyses. We did not integrate the indicator sets directly into FAMOS[space], because this would create non-linearity and would require simultaneous optimizations at farm and landscape levels. Furthermore, any kind of biodiversity targets or objective function weight would be needed, which are usually difficult to obtain.

To conclude, the integration of biodiversity in economic land use optimization models remains rather superficial concerning the assumptions on functional relationships between land use intensity, landscape complexity and biodiversity. However, joint optimization of land use and biotic effects seems desirable such as presented by Groot et al. (2007) and Parra-López et al. (2009). Consequently, further methodologies need to be developed that can jointly and endogenously consider the complexities of the socio-economic land use system and the surrounding natural processes at sufficient detail for biodiversity assessments.

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# Conclusions

**Guillermo Flichman**

The bio-economic modelling approach presented in this book is a result of two distinct developments: by one side, the improvement of bio-physical simulation models applied to agricultural systems and by the other, the evolution of agricultural policies demanding a kind of assessment that conventional economic models are not able to provide.

Some economists began to realise that biophysical models could be considered as detailed engineering production functions, allowing to represent in a consistent manner the joint products of agricultural activities. The perspectives that this vision provides allow dealing with environmental and natural resources issues with an economic perspective in an efficient manner. Representing environmental impacts of agricultural activities measured in physical units allows performing cost-efficiency calculations of alternative policies, potentially able to attain specified policy targets. This capability permitted in recent years the development of applied research related with institutional demands from national and international public institutions.

But this approach requires a multidisciplinary approach, with a positive and negative effects. The positive one is, both for economists and biophysical scientists, to enlarge their vision of the world. The negative effects are the greater difficulty to get recognized in their specific discipline, the obstacles to obtain the necessary information for properly use these models, and the longer time to perform the research activity. The "productivity" for producing papers is lower for economists applying this approach compared with economists applying econometric methods, using available published data.

In spite of these negative effects, as the demands from the real world for the assessment these models are able to provide is increasing, it is quite possible that there is a future for bio-economic models applied to agricultural systems. The challenges of Climate Change, the increase care for the preservation of natural resources and the environment will require further developments of this kind of approach.

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