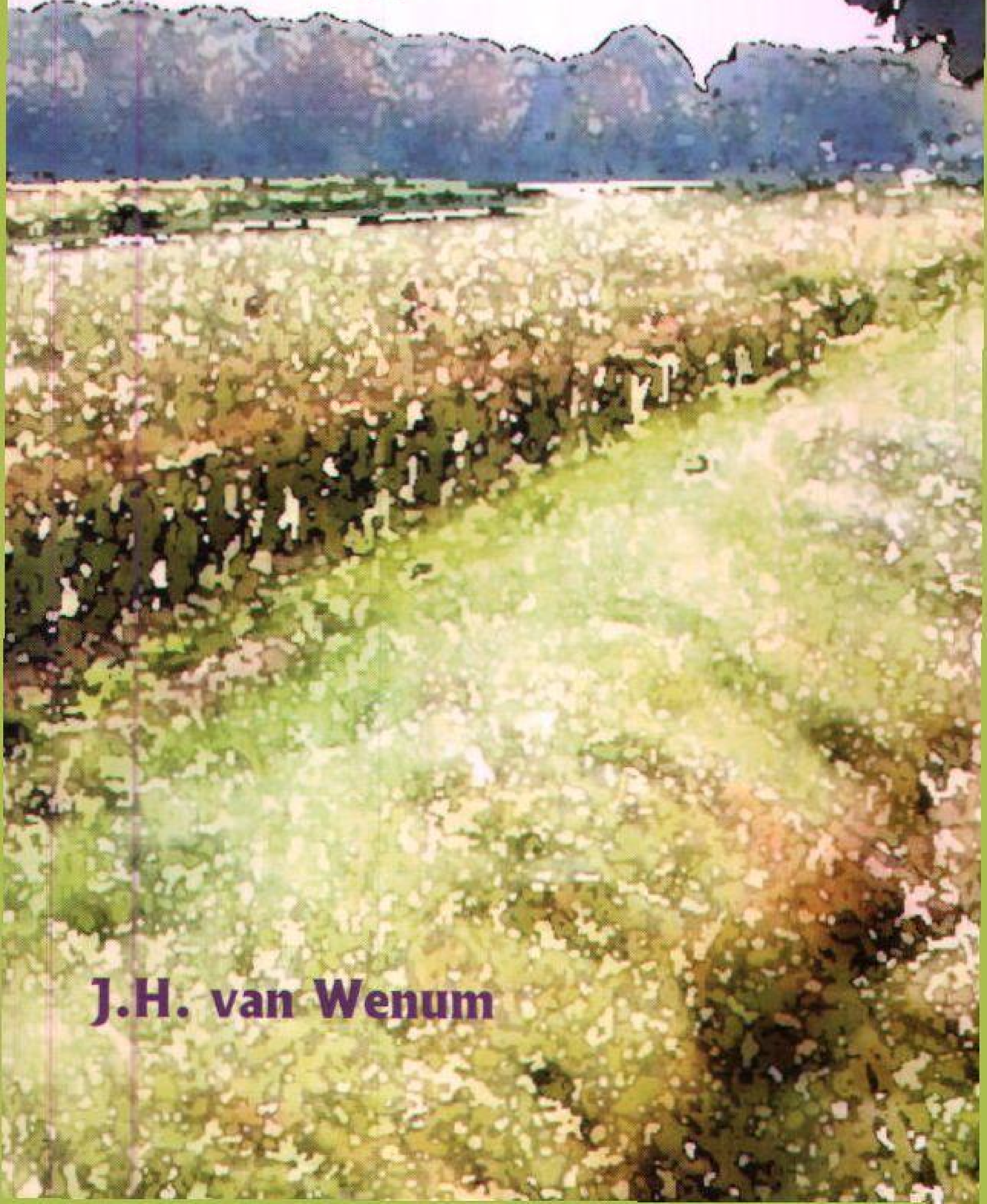
Economic analysis of wildlife conservation in crop farming



J.H. van Wenum

Stellingen

1. Bedrijfseconomisch gezien heeft, onder uniforme productie omstandigheden, het roteren van natuurbeheersactiviteiten op een akkerbouwbedrijf de voorkeur boven beheersactiviteiten op vaste locaties.
(Dit proefschrift)

2. Met een adequate vergoeding en een beheerspakket dat de risico's ten aanzien van de verspreiding van ziekten en plagen vermindert ten opzichte van de huidige beheerspakketten zijn beduidend meer akkerbouwers bereid tot agrarisch natuurbeheer.
(Dit proefschrift)

3. Met betrekking tot agrarisch natuurbeheer op het regionale niveau geeft "selective control" meer natuur tegen lagere kosten.
(Dit proefschrift)

4. Integratie van normatief en positief onderzoek, door het opnemen van modelresultaten in gedragsstudies en gedragsaspecten in modelstudies, verbetert het inzicht in de besluitvorming door agrariërs en landeigenaren ten aanzien van agrarisch natuurbeheer.
(Dit proefschrift)

5. De aanleg van nieuwe grootschalige natuurgebieden kan wanneer wordt uitgegaan van het opportunity costs principe het beste buiten de Nederlandse grenzen geschieden.

6. Neo-klassieke economen die Max Havelaar koffie kopen en daarmee inefficiëntie stimuleren zijn het bewijs van de beperktheid van hun eigen modellen.

7. De door Kahneman en Tversky (1979) geintroduceerde loss aversion theorie: "people are more avers to losses relative to the status quo than they are attracted by gains" geldt ook voor relaties.

8. Het succes van interdisciplinair onderzoek hangt niet zo zeer af van de compatibiliteit van de disciplines als wel van de compatibiliteit van de betrokken onderzoekers.

9. Grondbewerking in het donker als effectieve onkruidbestrijdingsmethode geeft aan dat parttime landbouw milieuvriendelijk produceren niet in de weg hoeft te staan.

10. Een landbouwwetenschapper heeft boerenwijsheid nodig.

11. Als ambtenaren staken gebeurt er weinig.

Stellingen behorende bij het proefschrift "Economic analysis of wildlife conservation in crop farming", Jaap van Wenum, Wageningen, 16 januari 2001.

ECONOMIC ANALYSIS OF WILDLIFE CONSERVATION IN CROP FARMING

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ECONOMIC ANALYSIS OF WILDLIFE CONSERVATION IN CROP FARMING

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ABSTRACT

Economic analysis of wildlife conservation in crop farming

Economische analyse van natuurbeheer in de akkerbouw

J.H. van Wenum, 2001

The general objective of this thesis was to present an economic analysis of wildlife conservation in Dutch crop farming. This general objective was broken down into 5 specific research objectives around which the research was organised: (1) selection and definition of appropriate indicators for wildlife in agriculture, specifically applicable at farm level, (2) definition of a wildlife production function, (3) definition of the optimal strategy for incorporating wildlife conservation measures on the farm from the economic viewpoint, (4) analysis of farmer participation in wildlife conservation programs and farmers' Willingness to Accept and (5) exploring the opportunities for a regional approach for wildlife conservation in agriculture. To achieve these objectives ecological, agronomic and (socio)-economic knowledge was used in a coherent combination of methods derived from econometrics, operations research, behavioural economics and network analysis. Random effects modelling was used to estimate wildlife production functions and estimates from this procedure were used together with agronomic and economic information in farm optimisation modelling to normatively study decision making towards wildlife management at the farm level. Decision making was also studied in a positive way by analysing factors that determine farmers' participation in wildlife programs and willingness to accept. Finally a pilot study was done to explore the possibilities of a regional network approach incorporating both normative and positive elements.

VOORWOORD

Voor een promotie-onderzoek staat in de regel vier jaar. En als je begint, denk je dat het nog een heel karwei zal zijn om die vier jaar vol te krijgen. Nu, bijna zes jaar na aanvang van het onderzoek ben ik nog steeds de mening toegedaan dat het eigenlijk best in vier jaar had gekund. Dat het toch een ietsje meer werd kwam mede door leuke nevenactiviteiten en de omzetting van een AIO-baan in een eerst gedeeltelijke en later volledige aanstelling als Universitair Docent. Verder vraagt een begeleiding op afstand, veel discipline van de onderzoeker zelf. Dat het boekje toch gereed is gekomen, is echter mede de verdienste van mijn begeleiders: promotor prof. dr. ir. Jan Renkema en copromotor dr. ir. Ada Wossink. Jan Renkema wil ik bedanken voor de kundige begeleiding, vooral waar het ging om het kritisch doornemen van de diverse hoofdstukken van het proefschrift en natuurlijk ook voor het in goede banen leiden van de klankbordgroepvergaderingen. Gedurende de vele wandelingen in de lunchpauze, hebben we ook veel andere gesprekken gevoerd, iets wat ik altijd zeer gewaardeerd heb. Ada Wossink wil ik bedanken voor de enthousiaste en efficiëntie begeleiding van het onderzoek, eerst vanuit Wageningen en daarna vanuit de USA. Een begeleiding die ik vooral ervaren heb als samenwerking. Mijn wat praktische insteek gecombineerd met jouw wetenschappelijke meer theoretische benadering bleek een prima combinatie. Verder heb ik mede dankzij jou en ook Alastair een hele plezierige tijd gehad in Raleigh, waarvoor dank.

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Het werken aan een promotie-onderzoek wordt een stuk leuker in een stimulerende omgeving. De collega's van de leerstoelgroep ABE zorgden hiervoor met koffie- en taartpauzes, hardlopen of wandelen tijdens de lunch, vrijdagse bezoekjes aan "De Vlaam" en met de meer serieus bedoelde onderzoeksbesprekingen. Allen bedankt!

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Jaap van Wenum
Wageningen, 2001

CONTENTS

CHAPTER 1

GENERAL INTRODUCTION

1.1 Background

Agriculture does not only produce food and fibre; it also helps shaping the rural environment. Increasingly, modern society values the environmental benefits which may arise as joint outputs with primary land use, including e.g. semi natural habitats and wildlife. In Western Europe, rapid changes in primary land use have jeopardised the supply of these benefits (Lowe and Whitby, 1997). The Common Agricultural Policy has been criticised for supporting these changes and over the last decade European policy makers have begun to respond to such criticism. EU-regulations 1760/87 and 2078/92 mark the acceptance that supporting farmers to conserve wildlife and countryside might help to curb overproduction. These regulations also promote a specific approach: supplementary to a distinct geographical segregation of agricultural and wildlife functions both functions should to a large extent blend within the rural environment. While nature reserves will always be important, there is a shift of attention increasingly to the preservation of biological diversity within the major forms of primary land use, particularly agriculture (Edwards and Abivardi, 1998).

In order to preserve and restore wildlife functions on agricultural land active wildlife management on farms is required. Identification of cost efficient wildlife policies depends on the relationships between current agricultural land use activities and wildlife values, and assessment of opportunity costs of foregone uses. A normative approach investigating the trade-off between wildlife and agricultural production and income is needed to this end and measurement and definition of wildlife values into appropriate indicators is essential. Despite the importance of the issue and the wide policy interest, the list of quantitative studies on economic efficiency versus wildlife trade-offs is limited. Studies at the crop level have generally focused on the positive effects of refraining from pesticide use in northern European agriculture on the abundance of flora and fauna (see Boatman, 1994 and De Snoo, 1994). Economic studies at the farm level generally involve a comparison of specific land use regimes by analysis of accounting data and/or farm level modelling (*e.g.* Van Eck *et al.*, 1987; De Boer, 1995). Spatial aspects such as site selection and connectivity have so far not been considered in these studies.

A normative approach serves well for strictly analysing trade-offs at the farm level. However, when forecasting or explaining farmers' adoption of wildlife practices and participation in wildlife programs, a normative trade-off analysis is insufficient. According to D'Souza *et al.* (1993) factors affecting adoption of sustainable agricultural practices can be grouped under human capital (e.g. age and education), structural (e.g. farm size, debt/asset ratio), institutional (e.g. participation in farm commodity programs) and environmental categories (e.g. awareness of environmental problems). Wilson (1997) following Brotherton (1989) states that both 'scheme factors' and 'farmer factors' need to be taken into consideration when attempting to understand farmer participation in agri-environmental schemes. Scheme factors include payments offered, duration and voluntary nature of the scheme. Farmer factors include various individual farm and farmer characteristics. Other studies in more recent years emphasise the importance of farmer attitudes towards the environment (Morris and Potter, 1995) and how structural and attitudinal factors interplay in the individual farmers' decision making process (*e.g.*, Falconer, 2000). Economic models based on profit maximisation fail to encompass attitudinal variables altogether whereas omission of important explanatory variables that are correlated with variables included in econometric models leads to biased estimators and to invalidation of inference procedures (Greene, 1997). To explain program participation a comprehensive utility-based approach is required that integrates normative economic and behavioural aspects, together with institutional and agronomic aspects at the farm level.

In studies of agricultural wildlife management at the farm or lower level economic and ecological criteria prevail and spatial aspects are usually disregarded. In contrast, the size and number of sites, spatial continuity and wildlife targets are the main focus of studies at the aggregate level (see for example Camm *et al.*, 1996; Pressey *et al.*, 1996). Only very recently are economic criteria receiving attention in landscape studies. For example, Polasky *et al.*, (2001), incorporate land values and a budget constraint in their study of selecting biological reserves.

1.2 Objective

The objective of the thesis is to conduct an economic analysis of wildlife conservation in crop farming for the situation in The Netherlands. Starting point for the thesis is the farm level as the interaction between wildlife, agricultural practices and income, as well as associated decision making is most pronounced at this level. Five research questions are formulated and will be answered:

1. How can wildlife be defined and what are appropriate indicators for measuring wildlife in an agricultural setting?
2. Which factors determine wildlife production of agricultural and conservation activities and how can wildlife production be modelled?
3. What is the optimal strategy for incorporating wildlife conservation measures on the farm from the economic viewpoint?
4. How do farm and farmer characteristics, as well as behavioural factors affect participation in wildlife conservation programs and Willingness to Accept?
5. What are the opportunities for a regional approach for wildlife conservation in agriculture using farm level data?

As the starting point of the study is the farm level, specific attention is paid to indicator selection and data collection and analysis at this level. Hence, the explorative regional analysis (research question 5) will also depart from the farm level. The importance and validity of a regional approach downscaled to the farm level is recognised, but will not be applied here.

1.3 Outline of the thesis

The outline of the thesis is as follows: Chapter 2 focuses on wildlife quality indicators and the specific requirements when applied to agriculture. Chapter 3 presents the functional form and estimation technique for a wildlife production function at the farm level. A random effect model is developed to capture the relationship between wildlife output, management practices, natural conditions and non-observed farm specific factors. Next Chapter 4 presents a location specific model for optimising wildlife management on crop farms using the integer programming technique Indicators and wildlife estimates where derived from the previous two chapters. Most important model outcome is a wildlife-cost frontier at the farm level. A positive behavioural approach towards agricultural wildlife conservation is presented in Chapter 5. Factors explaining farmers' participation in existing wildlife programs and contingent participation in a proposed field margin program are analysed. Chapter 6 explores the possibilities for a regional optimisation of wildlife conservation in agriculture combining normative and positive research results into a network analysis. Finally Chapter 7 concludes the thesis with a general discussion.

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CHAPTER 2

WILDLIFE QUALITY INDICATORS IN AGRICULTURE[1]

Abstract

This chapter focuses on wildlife quality indicators and the specific requirements when applied to agriculture. The value of nature and methodological principles of indicators are discussed and recently developed wildlife indicators are described. Special attention is drawn to the so-called yardstick for biodiversity developed by the Centre for Agriculture and Environment. This yardstick is an instrument specifically to quantify and value wildlife on farms. Details of this yardstick are discussed and its opportunities for use in farm management, research and policy towards wildlife conservation in agriculture are highlighted.

2.1 Introduction

The European Commission's Fifth Environmental Action Programme *(Towards Sustainability)* and the Reform of the Common Agricultural Policy in 1992 have both initiated new legal and strategic actions to integrate environmental objectives into sectoral policies. Agriculture, in particular, is considered a driving force affecting the state of, and changes in, Europe's biological and landscape diversity. The former natural vegetation of the continent of Europe has been changed virtually in its entirety under the influence of human land-use activities; large proportions of existing and future ecological values are more or less directly dependent on the way the land is utilised.

In the Netherlands the issue became a topic of public and political interest in the 1980's. It has led to new policies on land use, agricultural practices, nature and landscape, namely the National Environmental Policy Plan (NEPP) of 1989 and the Nature Policy Plan (NPP) of 1990. The main focus of the NPP is the Ecological Main Structure (EMS), an ecological network consisting of core areas (nature reserves) and nature development areas, connected by means of corridors. Outside the EMS, where agriculture is the dominant land use, NPP envisages a minimum environmental

[1] Based on: Van Wenum, J., J. Buys and A. Wossink (1998) Nature quality indicators in agriculture. In: Brouwer, F. and B. Crabtree (Eds.) Environmental indicators and agricultural policy. Wallingford, UK, CABI: 105-120.

quality, supplied by farmers and other land users. However, the implementation of this part of the NPP policy plan is still in its infancy. With regard to wildlife conservation on farms the focus is now on the development and introduction of new incentive schemes based on wildlife, supplied, in addition to the current practice, with command and control schemes (Van Paassen *et al.*, 1991; Van Harmelen *et al.* 1995; MLNV, 1995; Ter Steege *et al.*, 1996). For the NPP implementation, both regarding the EMS and the incentive schemes, the need is felt for indicators showing wildlife quality and its development over time (see also Udo de Haes *et al.*, 1993; Nature Conservation Council, 1995). Indicators are crucial instruments as they enable goal setting and impact evaluation. Moreover they can serve as a basis for conservation payments to farmers, and provide a basis for certification schemes for agricultural products (Buys, 1995; Horlings and Buys, 1997).

For farmers, indicators can be a useful guide to management decisions directed towards wildlife conservation by providing information on the ecological benefits of modifications to farming practice and of specific wildlife conservation measures. This kind of information is also useful for normative economic research on agriculture and wildlife conservation. Several economic tools are available which incorporate the relationships between agricultural output, environmental quality indicators and production techniques focusing at the field crop or whole-farm level. Partial budgeting and programming methods are the predominant methods (for an overview see Roberts and Swinton, 1996). Typically, these methods are used to gain insights into the trade-offs between income and environmental stress (see, for example, Wossink *et al.*, 1992; Foltz *et al.*, 1995; Teague *et al.*, 1995; Verhoeven *et al.*, 1995). These approaches can also be used to study the implications of wildlife conservation at the farm level. With a wildlife quality indicator, account can be taken of both the economic effects of wildlife conservation measures and the ecological consequences. Constrained optimisation procedures may be used to define best management strategies at farm level for each targeted wildlife quality index. Finally, generally accepted quality indicators facilitate communication between the parties involved.

This chapter focuses on wildlife quality indicators and the specific requirements involved when they are applied to agriculture. Special attention is drawn to the so-called *yardstick for biodiversity*, an indicator developed recently by the Centre for Agriculture and Environment (Buys, 1995).

In the second section, some conceptual considerations on nature and biodiversity and on criteria and indicators for wildlife quality are presented, and this is followed by a review of recent developments in designing indicators and on specific requirements for wildlife quality indicators when applied in agriculture. Then the recently developed yardstick for biodiversity and initial experience with its use are analysed. The chapter concludes with a discussion and outlook.

2.2 Conceptual considerations

2.2.1 Nature and biodiversity

Nature is often defined as everything that organises and sustains itself, whether associated with human action or not (see also Veeneklaas *et al.*, 1994; Nature Conservation Council, 1995). This definition implies different gradations from almost natural (e.g. jungle) to cultural (e.g. roadsides). The value of nature is often associated with the biotic (living) part of the natural environment. The term "biological diversity", usually shortened to biodiversity is commonly used to indicate the total biotic environment. Biodiversity is an umbrella term for the number, variety and variability of the living organisms in a given assemblage (Pearce and Moran, 1994). So the assemblage, or abiotic natural environment, is considered conditional for the state of the biodiversity. Hence, biodiversity is a useful representation of the quality of the natural environment.

Biodiversity may be described in terms of genes, species and ecosystems, corresponding to three fundamental and hierarchically related levels of biological organisation (Pearce and Moran, 1994). First, genetic diversity is the sum of genetic information contained in the genes of individual plants, animals and micro-organisms; secondly, species diversity considers the richness, variation or number of different species, and populations within which gene flow occurs under natural conditions; and, thirdly, ecosystem diversity relates to the variety of habitats, biotic communities and ecological processes in the biosphere as well as the diversity within ecosystems.

2.2.2 Valuing nature

Veeneklaas *et al.* (1994) summarise the two principal views that exist with respect to the value of nature:

1. Ecocentric: nature has an intrinsic value independent of its utility to mankind, and conservation of each individual species is therefore equally relevant.
2. Anthropocentric: nature has a value to humans as nature fulfils different functions:
   - production function, including production of renewable and non-renewable resources;
   - carrier function: nature as space provider for human activities;
   - information function: the availability of nature for recreation, science, education;
   - regulation function: stabilising functions of nature to environmental changes such as (water) purification capacity and resistance and retention features of nature.

In the environmental economics literature it is common to consider the total economic value of nature as the aggregate of all user and non-user motivated values. Although the user and existence point of view might be complementary, the associated values might be overlapping (Holstein, 1996). Considering the values, i.e. the functions nature performs, its interrelationships with agricultural land use are obvious. Hence it is no surprise that the main causes for the deterioration of the natural environment are related directly to agricultural practice (Terwan and Van der Bijl, 1996): loss and fragmentation, acidification, eutrophication and desiccation.

2.2.3 Wildlife quality criteria and indicators

Although the functions performed by nature can be defined, these are difficult to address in policy design. The NPP (1990) concentrated specifically on biodiversity. Three criteria of biodiversity were specified as objectives in policy development: (i) naturalness, (ii) diversity (in terms of (inter)national rarity of species) and (iii) 'characteristic features'. Naturalness not only includes the degree of human intervention but also the size and completeness of ecosystems. 'Characteristic features' seems a redundant criterion as one would expect that under more or less natural conditions (i.e. no human interference) only species and ecosystems will be present that are characteristic of these conditions (Bal *et al.*, 1995). Direct measurement using these criteria is difficult and there is a role for indicators of performance that may directly or more tangentially relate to criteria specified.

In this chapter the focus is on indicators for agricultural areas, where nature coexists with human activities (agriculture). Therefore naturalness is not a relevant criterion. Hence, in context of the criteria outlined above, nature quality indicators for agricultural areas are defined only in the context of the diversity criterion.

The Organisation for Economic Co-operation and Development (OECD) work on agriculture and the environment recognised pressure, state and response (PSR) indicators (OECD, 1994). Hence, the PSR approach is applied to (bio) diversity in agriculture. Pressure indicators are measurements of agricultural activities that cause changes in the state of biodiversity, such as the use of pesticides and fertiliser. State indicators are direct measurements of the state of biodiversity arising from these pressures, in terms of species, habitats or environmental parameters. Finally, response indicators refer to responses by farmers, government or society to changes in the state of biodiversity, such as the use of financial incentives to farmers to enhance biodiversity. According to Udo de Haes *et al.* (1997), pressure indicators are less relevant as the concept of nature quality is effect-orientated and therefore indicators of impacts in terms of wildlife species or environmental parameters are preferable. Moreover, many of the relationships between human activities and the abundance of species and habitats are complex and poorly understood. With regard to state indicators, two types can be distinguished: (i) indicators that are direct measurements of species biodiversity, and (ii) indicators that are measurements of environmental quality or biotope (habitat) presence, conditional for the presence and abundance of wildlife. Examples of both types of state indicators are discussed in the next session. Response indicators go beyond the aims of this chapter and will not be discussed.

2.3 Designing wildlife quality indicators

2.3.1 Some reflections on recently developed indicators

Recently developed indicators for nature quality have been based mainly on the diversity concept, and indicators considering both species diversity and ecosystem (biotope or habitat) diversity have been developed.

An example of a biotope-diversity-based nature quality indicator for agricultural areas is described by Wahlberg-Jacobs (1991). Arable areas in Germany are assessed in terms of the density and size of landscape elements (hedges, ditches, tree groups, etc.) as well as agricultural practices such as pesticide use, nitrogen utilisation and farming system (conventional, organic). The method also takes into account soil type and susceptibility to erosion. The resulting value indicates the wildlife tolerance of the studied area. Smeding (1995) who explored the nature quality characteristics of organic farms also measured the area and variety of biotopes. Although such methods consider the quantitative and qualitative biotope requirements for the presence of wildlife, information on the actual presence of species is not recorded. Species-diversity-based indicators usually depart from the true presence and abundance of species and give more direct information on nature quality.

Many attempts have been made to measure biodiversity by constructing species diversity indexes. Ecologists often construct biodiversity indexes as a function of species counts and the relative abundance of species (Magurran, 1988). Species richness is the simplest form of index, but it omits differences in abundance between species in a set. Economists have taken a different approach by constructing biodiversity indexes as a function of genetic distances among members of a species set (Solow *et al.*, 1993). Species richness fails to incorporate this concept.

An example of a species-diversity-based indicator that has also been used to define policy objectives is the AMOEBA approach of Ten Brink (1991). In order to examine the quality of the North Sea ecosystem and its development over time, as well as to define targets, it combines a reference population size (in the year 1930), current population size and target size of a representative selection of marine species. The focus is on reaching the reference size in a pre-determined period. Target sizes of species reflect policy objectives to move the system towards this situation. Criticism of this approach has focused on the representativeness of the species set, the unpredictability of wildlife and corresponding interpretation of the quantitative goals (De Bruin *et al.*, 1992; Van der Windt, 1996). More recently the method has also been applied to other (aquatic and terrestrial) ecosystems.

Species diversity indicators linked to wildlife protection in the nature policy of the Netherlands have also been developed. This approach determines the protection need of species in terms of international importance, rarity and population development (tendency). An example is a vascular plant species-based indicator for vegetation developed by the province of South Holland (Clausman *et al.*, 1984). Species values were based on current rarity at different scales (provincial, national and international) as well as population size development, and used, together with values for species cover, to calculate the floristic value of vegetation.

A more extensive method for valuing wildlife in agriculture, using the protection approach, has been developed recently by the Centre for Agriculture and Environment (CLM) (Buys, 1995). This yardstick for biodiversity not only involves vascular plants but also includes mammals, birds, butterflies, amphibians and reptiles. Species values based on rarity, population development and international importance are used in conjunction with census methods to calculate the wildlife value of farms.

Sijtsma *et al.* (1995), when predicting the impact of the implementation of the EMS, used a similar methodology but different species selection since non-agricultural ecosystems were involved. Census methods were not used. Criticism of the methodology focused on the representativeness of the species set which consisted of so-called goal species - principally endangered species (Buys, 1996).

Wildlife quality dependent payment systems for use in agriculture have recently been developed and implemented by Leyden University (Mibi) and the Ministry of Agriculture, Nature Management and Fisheries (LBL) (Van Paassen *et al.*, 1991; Van Harmelen *et al.* 1995; Ter Steege *et al.*, 1996). For meadow birds a system is used with species specific payments, based on the protection need of the species. For field margin schemes a system is used based on the presence of a selected number of plant species. Payment levels under both schemes are indicative of the wildlife quality on the farms.

2.3.2 Demands for agriculture

Within the agricultural sector different users have different requirements for wildlife quality indicators. Farmers require clear benefits from the assessment of wildlife quality. Wildlife quality indicators can be introduced successfully only when they are used within incentive or certification schemes. Since unlike other agri-environmental indicators (e.g. mineral accounts), they cannot be extracted from farm accounting data, and therefore require an additional labour input, it is important that assessment for wildlife quality can be performed by farmers and, from a policy perspective, indicators have greater value if they can be used in incentive schemes. Furthermore, it is important that information on wildlife quality is made available at farm scale. There is a deficiency of farm-level data as environmental information is normally gathered at a higher level (ecosystem, region). The farmer, however, bases decisions on available farming techniques and on farm economic considerations. Hence the farm level is also the level of aggregation on which the economic and the ecological interactions are most pronounced.

Taking the foregoing into account, The CLM yardstick and the Mibi/LBL-methods appear most appropriate for application in agriculture. The Mibi/LBL-methods are simple but relatively weak in terms of biotopes and species coverage. The CLM yardstick is more complex, requiring more knowledge and time, but covers a much broader range of species and also takes the whole farm into account (Horlings and Buys, 1997). The remaining part of this chapter therefore places emphasis on this indicator.

2.4 Yardstick for biodiversity

2.4.1 Basic principles

The yardstick for biodiversity is an instrument for quantifying and valuing wildlife (biodiversity) on farms. The yardstick is based on the following main principles: (1) biodiversity refers to organisms which (may) establish and sustain spontaneously and is assessed at the level of individual species; (2) at farm level a quantitative assessment of biodiversity is made of each biotope, although in the testing process the yardstick was simplified by not splitting the farm into separate biotopes – such an assessment is based on a representative selection of (groups of) species; (3) the yardstick assigns a rating to selected species expressing their significance to society; and (4) yardstick scores are made up for each species group and indicate the value of wildlife on a farm as follows:

$$Y(s)=\sum_{b=1}^{B}\sum_{p=1}^{P}C_{bp}V_{p} \qquad [2.1]$$

where:

$Y(s)$ = *yardstick score of species group s*

B = *number of biotopes on farm*

P = *number of selected species in species group s*

C_{bp} = *census units of species p in biotope b*

V_p = *value to society (rating) of species p*

The structure of the yardstick is such that it enables farmers to identify how biodiversity may be increased. The yardstick for biodiversity is applicable to all farm types (Buys, 1995).

2.4.2 Species selection

The yardstick for biodiversity comprises a limited selection of species groups which:

- enables reliable and simple census methods to be used (that can be undertaken by farmers);
- provides information on the effect of farm management on biodiversity;
- reflects biodiversity on farms.

Based on these principles, vascular plants, larger mammals, birds, amphibians, reptiles and butterflies were selected. (Butterflies were selected for their relative importance to wildlife conservation on farms despite the lack of a simple quantitative census method for the group.) From each of these groups individual species were selected taking into account the following criteria:

- presence of the species in agricultural areas;
- likelihood of encountering the species;
- correlation between the species and farm management;
- indicative value of the species for the condition of the biotope;
- recognition of the species.

These criteria resulted in a list of 199 species of vascular plants, 17 of mammals, 77 of nesting birds, 14 of wintering birds, 7 of amphibians, 2 of reptiles and 6 species of butterflies with the aim of covering the entire country and all farmland habitats (Buys, 1995).

2.4.3 Census methods

An essential aspect of the yardstick for biodiversity is a quantitative assessment of biodiversity for different biotopes on farms (Table 2.1). Quantitative census methods must be reliable, feasible for use by farmers and preferably comparable with other census projects (Buys, 1995).

Table 2.1 Census methods used by the yardstick for biodiversity

Species group	Census method
Vascular plants	CBS classification[a]
Mammals	Roost, den or number counts[b]
Nesting birds	Tally method and fledgling counts
Wintering birds	Abundance classification x length of stay
Butterflies	-
Amphibians	Egg batch/string counts or recording calling males[b]
Reptiles	number counts

[a]) CBS = Central Statistical Bureau
[b]) In accordance with species characteristics

Commonly used census methods for vascular plants are the methods of Braun-Blanquet and of Tansley (Den Held and Den Held, 1992), but these methods require specific knowledge in order that species cover can be assessed. Recently, an alternative census method has been developed by the Central Statistical Bureau (CBS) which is supposedly easier to use. Both abundance, using nine number classes, and cover of species, using percentages, are estimated. For the yardstick for biodiversity, assessing plant cover is less relevant and therefore only the abundance is estimated using this CBS-classification (Buys, 1995). During the process of testing on farms this classification was simplified and reduced to four number classes (1-10, 11-100, 101-1000, >1000 plants).

Census methods available for mammals generally depart from counting individuals or counting the number of roosts or dens (De Wijs, 1994). The yardstick for biodiversity uses both methods according to species characteristics (Buys, 1995).

Four census methods for nesting birds are available (Hustings *et al*, 1985): transect counts, the tally method, territory mapping and fledgling counts. The first three methods are based on counting occupied territories and differ in the area to be covered by the observations. The yardstick for biodiversity uses the tally method as providing sufficient quantitative information with relatively limited effort. Fledgling counts are used alongside this method to assess the breeding success of species nesting on the productive parts of the farm. The advantage of the latter method is that the results illustrate clearly the relationship with farming practice. For wintering birds transect counts and counting individuals or faeces are available methods. The first method provides only limited quantitative information and the last two are relatively time consuming; none of the methods takes into account the length of stay of the wintering birds. The yardstick therefore uses an adapted census method which includes counting individuals to assess the maximum number present during winter (in four classes) and determining the length of stay in months (Buys, 1995).

As mentioned earlier, there is no relatively easy-to-use quantitative census method for butterflies: the yardstick therefore only assesses the presence of a butterfly species (Buys, 1995).

Recommended census methods for amphibians are counting egg batches or strings and counting individuals (calling males, Stumpel and Siepel, 1993). The yardstick uses both methods (using five number classes) in accordance with species characteristics (Buys, 1995). Reptiles are usually monitored by counting individuals (Stumpel and Siepel; 1993) and the yardstick also uses this method.

2.4.4 Rating of species

Yardstick scores are made up by multiplying the number of units resulting from the farm census and a species rating score. This rating score expresses the importance society attaches to a species. The importance of a species to society depends on rarity, the degree to which it is endangered and attractiveness (NPP, 1990). The first two aspects relate to the ecological importance of species, the latter one relates to the contribution of a species to the scenery. The yardstick uses a rating system based on the ecological importance of species and this is used here, although for vascular plants a system based on the scenic value is also available (Buys, 1995).

Three aspects are important when considering the ecological importance of species (Clausman *et al.*, 1984; Bink *et al.*, 1994):

- rarity: population size (assessed regionally, nationally or internationally);
- tendency: development in population size (assessed regionally, nationally or internationally);
- international importance: the importance of the presence of a species in the Netherlands to the global survival of that species.

All three aspects were incorporated in the yardstick rating system and applied at national level. Rarity of species (R_p) was calculated by dividing the total number of topographical grid cells in the Netherlands (1677, each grid cell at 25 km^2) by the number of grid cells in which a species is found:

$$Rp = 1677/number\ of\ topographical\ grid\ cells\ with\ species\ p\ found^{2} \qquad [2.2]$$

Tendency of species (T_p) was calculated by assessing the change in (national) population size in terms of percentage:

$$T_p = \left(\frac{(past\ population\ size_p - current\ population\ size_p)}{past\ population\ size_p} \right) * 100 \qquad [2.3]$$

Different time periods were considered when assessing population changes of different species groups, mainly due to limitations in availability of data. For species with constant or increased population sizes, T_p equals 0.

For different species groups, different criteria are used to assess international importance (Bink *et al.*, 1994). The number of criteria range from 1 (mammals, amphibians and reptiles) to 4 (nesting birds). International importance of species (I_p) was incorporated in the yardstick rating by determining the relative number of criteria met by a species:

$$I_p = \frac{number\ of\ criteria\ for\ international\ importance\ species\ p\ meets}{number\ of\ criteria\ for\ international\ importance\ of\ respective\ species} \qquad [2.4]$$

The ecological rating of a species (R_p) was defined according to the following procedure (Buys, 1995):

1. Minimum values of components of ecological importance were put at 1.
2. The maximum value for international importance was put at 1.5 because of the relatively minor importance of this component to wildlife conservation at a national level.
3. Components of ecological importance were multiplied.
4. To achieve a rating range from 1 to 100 as well as distinction between (general) species values a logarithmic transformation was carried out.

[2] For nesting birds a different measure was used (because of data availability): true population / theoretical maximum population; the theoretical maximum population equals the current population of the most abundant bird species in the Netherlands (blackbird, 1 million couples).

The ecological rating of a species (V_p) was calculated as follows:

$$V_p = 18.5 * \log\left(R_p * T_p\left(1 + \left(I_p * 0.5\right)\right)\right) \quad [2.5]$$

2.4.5 Example

Table 2.2 presents an example of a yardstick result form based on a fictitious 20 ha farm with 2 biotopes: grassland (18 ha) and ditches (2 ha). Census results are used together with the ecological rating of species to make up biotope scores for each of the species groups. Biotope scores are then added up to produce farm scores for the respective species groups. Biotope and farm scores per hectare can also be calculated. It is not possible to aggregate species group scores to a single wildlife quality score per farm or per ha as both census methods and the rating method differ for the various species groups. However, when more yardstick results become available a reference level for each species group may be defined and relative scores may be introduced. This would enable comparison and aggregation of species group scores.

Table 2.2 Example of a yardstick form based on a fictitious farm with two biotopes (after Buys, 1995)

	Vascular plants	Nesting birds	Wintering birds	Mammals	Amphibians and reptiles	Butterflies	Acreage (ha)
Biotopes							
Grassland	318	170	10	10	0	15	18
Ditches	309	95	0	6	81	0	2
Farm total	627	265	10	16	81	15	20
Average score ha^{-1}	31.4	13.3	0.5	0.8	4.1	0.8	

2.4.6 Practical experience

The presented yardstick for biodiversity has only recently been developed. So far, the yardstick has been used to analyse the results of an experiment to enhance natural values on set-aside land (Buys *et al.*, 1996). In 1995 a small-scale test was carried out on four dairy farms (Buys and Ter Steege, 1996). In 1996 another eight dairy farms tested the yardstick and in 1997 and 1998 it was applied to 22 dairy and arable farms (Oosterveld and Guldemond, 1999). Limitations of the yardstick revealed by these tests are discussed below.

In the set-aside experiment, ecological researchers used the yardstick, so no information on its ease of use by farmers was obtained. However, one limitation of the yardstick did appear. It became apparent that calculating biotope scores per hectare was inaccurate for vascular plants where census classes were used rather than true numbers. This reflected the relationship between the composition of the census classes and the area of the trial fields. Differences in field size could produce differences in biotope scores per hectare despite the presence of similar census classes. This was unexpected given the actual densities of species and the field size ratio (Buys *et al.*, 1996).

The test carried out on the dairy farms had two main objectives: first, to analyse the feasibility of farmers applying the yardstick methods and second, to analyse whether the yardstick scores represent the wildlife present on farms. Both the farmer and a field biologist carried out the same assessments. To analyse the representativeness of the selected species, all species within yardstick species groups were rated and counted. Total scores obtained were compared to scores obtained when using the selected species set only.

Farmers found it relatively easy to trace and identify species with the exception of tracing vascular plants. Some 30% of the yardstick species from this group that were present were not traced. The results found by the field biologist and the farmer differed for censuses of species groups where classes were used as opposed to true numbers (vascular plants, amphibians). With respect to the representativeness of the species selection, a similar pattern of scores across farms and biotopes was obtained from the total inventory and the yardstick species inventory (Buys and Ter Steege, 1996). Similar results were obtained when the yardstick was further tested on 22 dairy and crop farms in 1997 and 1998 (Oosterveld and Guldemond, 1999).

2.5 Discussion

In order to assess wildlife quality on farms the yardstick for biodiversity uses an approach of quantifying and rating a set of representative species. The rating is highly linked to nature policy in the Netherlands towards endangered species and the requirement for species protection. From a nature consumers' perspective, however, the aesthetic value of wildlife is very important. A rating of species which incorporates aesthetic features would therefore be more appropriate. Such a rating is available for plants (Buys, 1995). It however implies subjectivity as preferences among consumers may vary.

In order to minimise practical problems in its use by farmers, the yardstick uses a limited number of species (groups) and easy census methods where applicable. However the vascular plants group still seems to pose difficulties for application of the yardstick. Possible modifications, such as limiting the census area and using other census methods should therefore be considered. For ecological research purposes, i.e. the analysis of ecological effects of wildlife conservation measures, the accuracy of the yardstick can be increased by carrying out a census of all species within a group or even other species groups, provided information that is needed to calculate a rating for these species, is available. Other census methods may also be applied. In the continuation of this thesis, research is presented in which the yardstick is applied to vascular plant species, using all species within this group, however ignoring census results of each species (similar to butterflies, see section 2.4.3). Census data are often lacking and census methods may differ significantly, complicating the calculation of yardstick scores. Within the research the yardstick is compared to a species richness measure.

Before applying the yardstick for policy purposes, and more particularly in incentive schemes, the methods have to be tested extensively. So far, incentive schemes based on wildlife output have used the presence of a limited number of vascular plant species or nest numbers of meadow birds and species-specific payments (Van Paassen *et al.*, 1991; Van Harmelen *et al.* 1995; Ter Steege *et al.*, 1996). The disadvantage of the yardstick compared to these methods is that it is time-consuming and results in higher costs for farmers (labour) and policy executors (inspection). The farmers in the tests described above (15 ha farms) needed a total of 20-30 hours to carry out the census and maintain records and found it hard to incorporate yardstick assessments into their day-to-day farm activities (Buys and Ter Steege, 1996; Oosterveld and Guldemond, 1999). Horlings and Buys (1997) acknowledge this drawback but state that a simplified version, using fewer species groups or biotopes, could enhance the potential of the yardstick for use in incentives schemes while maintaining some of its comparative advantages over other methods. Furthermore, provided the benefits to the farmer balance the efforts required in applying the yardstick in practice, they see a number of other future policy applications for:

- management agreements: adding a wildlife quality dependent premium on top of the current effort-based payments;

- tax and financial advantages to farmers in exchange for achieving minimum yardstick scores;
- farm certification;
- establishing standards for a minimum or desired wildlife quality.

With these applications in mind the focus is now on fine-tuning of the yardstick approach enabling further professionalisation of wildlife conservation by farmers.

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CHAPTER 3

THE IMPACT OF FARM HETEROGENEITY ON THE PRODUCTION OF WILDLIFE[3]

Abstract

This study presents the functional form and estimation technique for a wildlife production function at the farm level. A random effects model is developed to capture the relationship between wildlife output, management practices, regional conditions and non-observed farm specific factors. The study uses species richness and a wildlife yardstick in estimating wildlife production functions. The model was implemented for panel data of Dutch field crop farmers. Results showed that in terms of species richness, nature mix fallow was most beneficial to wildlife production when compared to other crops and fallow alternatives. When yardstick values were considered, unsprayed cereals were the most beneficial crops. Furthermore it was concluded that non-observed farm specific conditions are having a significant impact on wildlife production.

3.1 Introduction

World wide, there is growing concern about decreasing biodiversity, caused by the degradation of living conditions of plants and animals. Increasingly, modern society values wildlife biodiversity benefits that arise as joint outputs with agricultural land use (Edwards and Abivardi, 1998). To the extent that production of these public goods is not adjusted to social optima, a role for public policy is called for both in the short run and the long run. The US Endangered Species Act (ESA) and the EU Wild Birds Directive and Habitats Directive are examples of such public policies (Rosso Grossman, 1997; Brown and Shogren, 1998). In the short run, the public interest is in ensuring socially optimal use of privately controlled inputs that influence the supply of wildlife as a public good. In the long run, public policy can play an important role

[3] Van Wenum, J.H., A.G.J.M. Oude Lansink and G.A.A Wossink (2001). The impact of farm heterogeneity on the production of wildlife. Submitted *Ecological Economics*

by provision of research on new technologies that enhance wildlife productivity (e.g. organic farming systems). The focus of this paper is on the social interest of wildlife preservation specifically on the relation between agriculture and wildlife in the short run.

In agricultural areas where biodiversity has already seriously been damaged, ways have to be found for restoration and enhancement of biodiversity and this requires active wildlife management on farms. Identification of cost efficient wildlife policies depends on the relationships between current land use activities and wildlife values, and assessment of opportunity costs of foregone uses. In this task, agricultural economics has an important role to play. A first step includes the estimation of a wildlife production function, which captures the relationship between land use practices, natural conditions and wildlife at the farm level.

Effects of options for wildlife conservation management should be determined by using a production function that includes a definition and measurement of wildlife in a tangible way that enables different land use practices and site-specific conditions to be compared. A farm specific factor in wildlife production is expected, caused by non-observed factors such as past-activities on the farm, site specific conditions and management. The wildlife production function needs to account for this bias due to farm heterogeneity.

Many empirical studies focus on the economics of preventing losses in agricultural yields due to wildlife, for example the body of literature on crop protection and recent work on pre-emptive habitat destruction under the ESA (Lueck and Michael, 1999). In contrast, little work has been done on modelling the production relationship between agricultural practices and wildlife at the farm and field level. Particularly, there is little work involving non-experimental data. Usually field experiments are done, comparing species richness between different crops, water supply situations, fallow alternatives or across soil types (see *e.g.*, Adams and Cho, 1998; De Snoo, 1997). Data and relationships derived from such experimentally controlled settings do not include the complex, heterogeneous ecosystem processes which may vary across real life observations with input mixes and practices.

The first objective of the paper is to contribute to a better understanding of the appropriate functional relationship for wildlife production based on agronomic and ecological insights. Specific attention is also given to the selection of an indicator that correctly measures the wildlife output in response to wildlife management activities. A second objective is to investigate the importance of farm specific factors in wildlife production. The third objective is to show how the estimates of a wildlife production function based on panel data of Dutch field crop farmers can be used for management and policy support.

The outline of the chapter is as follows: Section 3.2 presents the theoretical framework of the research. Next, section 3.3 discusses the data and the wildlife indicators used for estimation. Section 3.4 describes the estimation procedure and section 3.5 presents results with special reference to management factors in wildlife production. The paper finishes with discussion and conclusions.

3.2 Theoretical model

Similar to yields of agricultural crops, wildlife output varies among regions, among farms and from year to year. The theoretical model presented here distinguishes production environments, growth factors (controllable and non-controllable by management) and production levels (*cf.* Wossink *et al.*, 2001). The production environment, E, at a specific site represents the setting for wildlife production and is characterised by physical factors that include climate and aspects of the soil (groundwater table, type of soil). Variation in potential wildlife production per unit of acreage in similar production environments is attributable to differences in growth factors that vary within a year and between years and that are not controllable by management. Non-controllable growth factors, Z, include: (a) weather during the growing season (solar radiation, temperature), (b) factors due to management in the past such as the level of eutrophication and desiccation of the soil, presence of vegetation remnants and extent of the flora seed bank. The non-controllable factors together determine the 'potential wildlife yield' for a specific production environment. The extent to which this potential level is achieved in practice depends on growth factors that are controllable by management, X. These include crops selected (including fallow), rotation, size and spatial pattern of field and field margins as well as nutrient management, water management and pest control. Together with the production environment and the non-controllable growth factors these factors determine 'actual wildlife yield'. To integrate the insights described the wildlife production frontier is defined as follows:

$$F(Y,X,Z;E)=0 \qquad [3.1]$$

where Y is wildlife output (actual yield), X is a vector of effective levels of management inputs, Z is a vector denoting the variable factors that are not controllable by management and E denotes the production environment. For a specific farm i ($i = i, \ldots, I$) and year t ($t=1,\ldots, T$) wildlife production may now be denoted as:

$$Y_{it} = f_i(X_{it}, Z_{it}; E_i) \qquad [3.2]$$

Assuming a common underlying technology of wildlife production over farms i ($i = i, \ldots, I$) and years t ($t=1,\ldots, T$) equation [3.2] may be reformulated as:

$$Y_{it} = \phi_i + h(X_{it}; E_i^{'}) \qquad [3.3]$$

where the parameter ϕ_i represents the i-th farm-specific effects and h is the wildlife production function. E'_i represents the observable part of the production environment E. Note that with non-experimental data, observations might be available on output Y_{it} , inputs X_{it} and on E_i but not on Z_{it}.

3.3 Empirical model

3.3.1 Data

Data on wildlife conservation in crop farming were available from different wildlife projects in the Netherlands varying in natural conditions and parcel lay out. This study used data on vascular plants collected from three different sources. In total 278 assessments were made on 49 farms.

Data were obtained from three different wildlife conservation projects. The first was obtained from field margin experiments carried out by the provincial authorities of Groningen and Drenthe in 1994. Vegetation assessments were carried out on sandy soil in both 6-meter wide unsprayed and conventional field margins of winter and spring cereals, potatoes and sugar beet. The second data set was obtained from a nation-wide project on alternative fallow field management, carried out by the Ministry of Agriculture, Nature Management and Fisheries in 1995. Inventories were made for different management options including phacelia, grass-clover, nature fallow mixtures (mixtures of more than three legumes/catch crops etc) and natural vegetation (spontaneously developed vegetation). Both margins and field centres were assessed. The third data set was obtained from two projects in the central clay area on grass-clover strips along crop fields. One project was carried out by the provincial authorities of North-Holland in 1995 and one by the Ministry of Transportation, Public Works and Water Management in 1994 and 1995. In both projects vegetation assessments were made for strips and for field edges: ditch banks and verges (Remmelzwaal and Voslamber, 1996).

Commonly used census methods for vascular plants are the methods of Braun-Blanquet, Londo and Tansley (Knapp, 1984). The projects considered in this paper either used the Londo-scale or Braun-Blanquet method to assess the vegetation. Both methods use small representative plots within which an inventory of all plant species is made. Plot sizes varied between projects and different scales were used to assess cover of plant species found.

3.3.2 Indicators

Many attempts have been made to indicate wildlife and biodiversity. The term biodiversity in the sense of the Biodiversity Convention of the UN Conference on Environment and Development in Rio de Janeiro (1992) encompasses the whole range of the genetic diversity within species, the diversity of species and higher taxa, up to ecosystem diversity, and even the diversity of ecological interactions. Clearly the Rio convention focuses on the more complex qualitative aspects of biodiversity. Quite obviously, such broad diversity of life cannot be measured in a comprehensive manner (Duelli, 1997).

The traditional scientific quantitative concept of biological diversity is based on species diversity. Indexes considering the quantitative aspects of biodiversity are often constructed as a function of species counts and the relative abundance of species (Magurran, 1988). Others have looked at evenness: biodiversity measures as a function of genetic distances among members of a species set (Weitzman, 1992; Solow, Polasky and Broadus, 1993). Species richness is the simplest form of these measures neglecting differences in abundance or genetic distance. Species richness provides an extremely useful measure of diversity if the study area can be successfully delimited in space and time and the constituent species enumerated and identified. Considering vascular plants in a farming situation, species richness therefore suits very well. Species density, for example the number of species per m^2, is the most commonly used measure of species richness and is especially favoured by botanists (Magurran, 1988). Species richness figures may easily be derived from the previously mentioned methods of Braun-Blanquet and Londo.

In this study two indicators for wildlife production are used. The species richness indicator (in terms of species density) is compared with an extensive species based indicator specifically developed for agriculture: the wildlife yardstick (see Buys, 1995; Van Wenum *et al.*, 1998). Whereas species richness considers the density of species, this yardstick provides information on the ecological or protection value of a species set. Another reason for using this indicator is the proposed use for future policy measures on agricultural wildlife conservation in the Netherlands.

The wildlife yardstick for vascular plants consists of a representative set of species. This representative set was put together for simplicity reasons. However to gain a more complete picture this study considers all plant species. To each plant species now, a rating V (0-100 points) has been assigned based on its protection need as determined by rarity, population tendency and international importance (all three at the national level).

A yardstick score per area measure (Y) now is calculated as the sum of ratings of all plant species p found for the respective area measure regardless of the number of plants per species:

$$Y = \sum_{p=1}^{P} V_p \qquad [3.4]$$

3.3.3 Model

Estimation of a wildlife production function ideally requires information about wildlife outputs, management inputs, non-controllable inputs and the production environment (theoretical model). However, the available data contain a limited amount of information. Wildlife output data are available as continuous variables in terms of species richness and wildlife yardstick values for different plot sizes. Information on management inputs is available in terms of crops or fallow variants grown on the field (discrete variable) and in terms of distance of the sample spot to the field edge (continuous). Information on the production environment is limited to the type of soil. No information is available on non-controllable management inputs.

However, the estimator that is employed is able to recover this information. A functional specification of the wildlife production function that incorporates the available information is the following:

$$Y_{it}^{W} = M_i + \sum_{j=1}^{8} \alpha_j A_{ijt} + \sum_{i=1}^{2} \beta_j S_{ij} + \gamma_1 D_{it} + \gamma_2 D_{it}^2 + \lambda \log G_{it} \qquad [3.5]$$

where Y_{it}^{W} is wildlife, measured either as species richness or as yardstick value per measurement plot. M_i is the unobservable farm specific management variable for the i^{th} farm, A_{ijt} denotes a dummy variable for agricultural management activities of the i^{th} farm at time t with j=1 (grass-clover), 2 (nature mix fallow), 3 (natural vegetation), 4 (unsprayed winter cereals), 5 (unsprayed spring cereals), 6 (potatoes), 7 (sugar beet) and 8 (phacelia fallow). The dummy variables A_{ijt} take the value 1 if activity j is present at time t at farm i and 0 otherwise. S_{ij} are regional dummy variables with j=1 (northern clay area) and 2 (central clay area) that take the value 1 if the j-th region applies and zero otherwise. The northern sand area is the reference area in this regression, i.e. S_{ij} is zero for all i,j in the northern sand area. D_{it} represents the distance from the sampling spot to the edge of the field (in meters). The quadratic specification allows for both increasing and decreasing marginal effect of distance on wildlife production.

After consultation with ecologists it was assumed that no marginal effects of distance were to be expected above 6 m (Kleijn, 1997). Sample spots at larger distances from the field edge where therefore truncated at 6 m. G_{it} represents the size of the sample spot (plot size) in m^2. In general a logarithmic relationship between plot size and species richness is assumed[4]. Ideally this relationship should be estimated for every distance to the field edge. Data availability however does not allow this and therefore both distance to the field edge and plot size were incorporated in the same regression. For yardstick values the same assumptions were made.

Table 3.1 presents the summary statistics of all observed variables in the wildlife production function.

[4]Species Richness = Constant + z log P, with z indicating the slope of the curve (Preston, 1948)

Table 3.1 Summary statistics for observed variables used in wildlife production function (N = 278 observations)

Variable	Mean	Std. Dev.	Minimum	Maximum
Species Richness	12.21	6.10	1.000	35.000
Wildlife Yardstick	88.273	53.654	0.000	266.000
Northern clay area*	0.072	0.26	0.000	1.000
Central clay area*	0.547	0.499	0.000	1.000
Grass-clover*	0.496	0.501	0.000	1.000
Nature mix fallow*	0.367	0.483	0.000	1.000
Natural vegetation*	0.014	0.119	0.000	1.000
Unsprayed spring cereals*	0.054	0.226	0.000	1.000
Unsprayed winter cereals*	0.014	0.119	0.000	1.000
Potatoes*	0.014	0.119	0.000	1.000
Sugar beet*	0.011	0.104	0.000	1.000
Phacelia*	0.029	0.167	0.000	1.000
Distance	2.698	2.049	0.000	6.000
Plot size	25.374	27.376	9.000	100.000

*) Dummy variables

3.4 Estimation

This section discusses the estimation procedure that is used in order to estimate the unobserved management variable and the structural parameters of [3.5]. Prior to this discussion, a few remarks are made about the available data.

First it should be noted that the available panel data set of Dutch farms is unbalanced, since the length of the time series differs by farm. Second, it should be noted that some variables differ across regions only, i.e. they are constant for individual farms. Therefore, the often applied fixed effects estimator cannot be used, since parameters associated with variables that vary across farms only, are incorporated in the fixed effect (Baltagi, 1995).

Estimation of the structural parameters and the unobserved management variable is achieved by applying a random effects estimator. The empirical model [3.5] is rewritten by imposing a specific error structure:

$$Y_{it}^{W} = \sum_{j=1}^{8} \alpha_j A_{ijt} + \sum_{i=1}^{2} \beta_j S_{ij} + \gamma_1 D_{it} + \gamma_2 D_{it}^2 + \lambda \log P_{it} + u_{it} \qquad [3.6]$$

where the composite error term u_{it} has the following structure:

$$u_{it} = M_i + e_{it} \quad [3.7]$$

M_i is assumed to be a random variable representing unobservable farm-specific factors that is i.i.d. $(0,\sigma_M)$ and e_{it} is i.i.d. $(0,\sigma_e)$ and M_i is independent of e_{it}. In addition, A_{ijt}, S_{ij}, D_{it} and P_{it} are assumed to be independent of M_i and e_{it} (Baltagi, 1995).

Estimation of the random effects model is done by transforming the data prior to estimation. The transformation gives the following equation:

$$\begin{aligned} Y_{it}^{W} - \theta_i \overline{Y}_i^{W} &= \sum_{j=1}^{8} \alpha_j (A_{ijt} - \theta_i \overline{A}_{ij}) + \sum_{j=1}^{2} \beta_j S_{ij} + \\ &\gamma_1 (D_{it} - \theta_i \overline{D}_i) + \gamma_2 (D_{it}^2 - \theta_i \overline{DD}_i) + \lambda (\log P_{it} - \theta_i \overline{\log P_i}) \end{aligned} \quad [3.8]$$

where θ_i is defined as $1 - \sigma_e / (T_i \sigma_M^2 + \sigma_e^2)^{0.5}$ and all barred variables indicate farm-specific means (averaged over t). From [8], it can be seen that estimating the random effects model requires a consistent estimate of σ_M and σ_e. An estimate for σ_e is obtained from the residuals of a fixed effects ('within') estimation of the model in [3.8]:

$$\sigma_e^2 = \frac{\hat{e}'\hat{e}}{n - N - K} \quad [3.9]$$

Where n, N and K are the total number of observations, the number of farms and the number of parameters to be estimated, respectively. A consistent estimate of σ_M is obtained from regression on the individual means:

$$\overline{Y}_i^{W} = \sum_{j=1}^{N} \alpha_j \overline{A}_i + \sum_{j=1}^{2} \beta_j S_{ij} + \gamma_1 \overline{D}_i + \gamma_2 \overline{D_i D_i} + \lambda \overline{\log P_i} + M_i + \overline{e}_i \quad [3.10]$$

This regression uses N observations (one observation for each farm) and it can be shown that the variance of the composite disturbance term $M_i + \overline{e}_i$ gives a consistent estimate of $\sigma_M^2 + \sigma_e^2 \overline{T}$, where $\overline{T}$ is defined as $\frac{1}{N} \sum_{i=1}^{N} \frac{1}{T_i}$ (Greene, 1998, p.337).

Estimates of the structural parameters of the wildlife production function are obtained by calculating the value of θ_i (using the estimates for σ_M and σ_e) and performing OLS on the transformed equation [3.8].

3.5 Results

Equation [3.8] was estimated using both the species richness and the wildlife yardstick value. Results of the random effects estimation can be found in Table 3.2. Parameters are significant at the critical 5% level for 38% of the species richness specification and 31% of the wildlife yardstick value specification. The R^2 of the species richness specification is 0.74 and 0.67 for the yardstick value specification. The results show that in the species richness specification, potatoes and sugar beet (α_6 and α_7) have a significant negative impact on wildlife production, which may be due to a larger use of pesticides on these crops. Other crop and fallow variants did not have a significant impact. For the yardstick specification, unsprayed spring cereals ((α_6) had a significant and positive impact on wildlife production. All other crop and fallow parameters were not significant for this specification.

Table 3.2 Random Effects estimation results of species richness and yardstick value specification

Species richness specification			Yardstick value specification		
Parameter	Estimate	t-value	Parameter	Estimate	t-value
α_1	-2.82	-0.98	α_1	38.64	1.67
α_2	3.70	1.39	α_2	32.47	1.56
α_3	2.56	0.75	α_3	40.77	1.40
α_4	0.72	0.19	α_4	61.54	2.04*
α_5	2.47	0.58	α_5	66.25	1.92
α_6	-9.96	-2.38*	α_6	-4.29	-0.13
α_7	-13.85	-3.25*	α_7	5.05	0.14
α_8	-3.38	-1.12	α_8	-7.11	-0.29
β_1	-3.20	-1.28	β_1	-21.17	-1.12
β_2	-1.83	-1.21	β_2	25.14	2.13*
γ_1	2.61	6.56*	γ_1	34.98	8.93*
γ_2	-0.40	-7.25*	γ_2	-4.97	-9.21*
λ	3.72	4.45*	λ	-2.20	-0.34

α_1-α_8) parameters associated with dummy variables of grass-clover (1), nature mix fallow (2), natural vegetation (3), unsprayed winter cereals (4), unsprayed spring cereals (5), potatoes (6), sugar beet (7) and phacelia fallow (8).
β_1-β_2) parameters associated with regional dummy variables.
γ_1-γ_2) parameters associated with distance from the field edge
λ) parameter associated with plot size
*) Significant at the critical 5% level.

Table 3.3 Differences between parameters α_i within species richness and yardstick value specifications (t-ratios in parentheses).

		Yardstick value specification							
		α_1	α_2	α_3	α_4	α_5	α_6	α_7	α_8
	α_1		-6.17	-2.13	-22.90	-27.61	42.93	33.59	45.75
S			(0.75)	(-0.09)	(-1.23)	(-1.09)	(1.75)	(1.28)	(2.99)*
P	α_2	6.15		-8.17	-29.07	-33.78	36.76	27.42	39.58
E		(7.33)*		(-0.40)	(-1.64)	(-1.38)	(1.54)	(1.07)	(2.80)*
C	α_3	5.38	-1.14		-20.76	-25.48	45.06	35.72	47.88
I		(2.32)*	(-0.53)		(-0.75)	(-0.78)	(1.42)	(1.08)	(1.91)
E	α_4	3.54	-2.97	-1.84		-4.71	65.83	56.69	68.65
S		(1.53)	(-1.34)	(-0.59)		(-0.21)	(3.08)*	(2.58)*	(3.13)*
	α_5	5.29	-1.23	-0.09	1.75		70.54	61.20	73.37
R		(1.79)	(0.42)	(-0.02)	(0.74)		(2.44)*	(2.05)*	(2.65)*
I	α_6	-7.14	-13.66	-12.52	-10.68	-12.43		-9.33	2.82
C		(-2.48)*	(4.87)*	(-3.51)*	(-4.67)*	(-3.93)*		(-0.35)	(0.10)
H	α_7	-11.02	-17.54	-16.41	-14.56	-16.32	-3.89		12.16
N		(-3.68)*	(-5.99)*	(-4.48)*	(-6.46)*	(-5.11)*	(-1.40)		(0.42)
E	α_8	-0.56	-7.07	-5.93	-4.09	-5.85	6.58	10.47	
S		(-0.35)	(-4.84)*	(-2.27)*	(-1.58)	(-1.84)	(2.11)*	(3.24)*	
S									

α_1-α_8) Parameters associated with dummy variables of grass-clover (1), nature mix fallow (2), natural vegetation (3), unsprayed winter cereals (4), unsprayed spring cereals (5), potatoes (6), sugar beet (7) and phacelia fallow (8).

*) Significant at the critical 5% level.

Table 3.3 shows differences between parameters α_i within the species richness and yardstick value specifications. For example, the first value in row one (-6.17) is the difference between α_1 and α_2 within the yardstick value specification, indicating that grass-clover and nature mix fallow do not significantly differ in their contribution to wildlife production (in terms of yardstick value). The first value in column one (6.15) gives the difference between α_1 and α_2 for the species richness specification, however within this specification nature mix fallow contributes significantly more to wildlife production than grass-clover. For the species richness specification it can be seen that sugar beet and potatoes (α_6 and α_7) are not beneficial to wildlife production as they have a significantly lower contribution than all other crop and fallow alternatives. Within the group of fallow alternatives, nature mix fallow and natural vegetation (α_2 and α_3) contribute significantly more to wildlife than grass-clover and phacelia (α_1 and α_8) for this specification. From the yardstick value specification it is clear that unsprayed cereals (α_4 and α_5) are most beneficial, contributing significantly more to wildlife production than sugar beet, potatoes and phacelia. Furthermore grass-clover and nature mix fallow score significantly higher than phacelia fallow in yardstick value terms. For both specifications it can be concluded that testing for differences between parameters gives additional information on the contribution of these parameters to wildlife production when compared to the regression results only.

Table 3.2 shows that the soil parameters (β_1 and β_2) are not significant for the species richness specification implying that soil type does not have a significant impact on species richness. However, for the yardstick value specification, β_2 (central clay parameter) is significant and positive, indicating that wildlife found in the central clay area has a higher ecological value than wildlife found on sandy and northern clay soils. The distance parameters (γ_1 and γ_2) are significant for both specifications and indicate that species richness and yardstick value (initially) increase though at a decreasing rate with distance. The plot size parameter λ is significant and positive for the species richness specification, whereas plot size does not have a significant impact on the yardstick value. The value of 3.97 for the species richness specification implies that species richness increases by 3.97% if the plot size that is used in the data collection increases by 1 square meter. The positive impact of plot size on species richness is consistent with ecological theory (Preston, 1948).

Non observed farm specific aspects might significantly contribute to wildlife production. The joint significance of all non-observed farm specific effects is tested using a Lagrange multiplier test (Baltagi, 1995, p.163). Under the null hypothesis: H_0: all $\mu_i = 0$, or equivalently $\sigma_\mu^2 = 0$, this test is asymptotically distributed as $\chi^2(1)$. The test statistic is 24.08 for the species richness specification and 84.33 for the yardstick value specification. Therefore, the null hypothesis all $\mu_i = 0$ is clearly rejected for both specifications at the critical 5% level, which takes the value 3.84. The results of these tests imply that farm specific conditions have a significant impact on the production of wildlife in terms of species richness and in terms of yardstick value.

3.6 Discussion and conclusions

The paper presents the functional form and estimation technique for a wildlife production function at the farm level. A random effects model is developed to capture the relationship between wildlife output, management practices, regional conditions and non-observed farm specific factors. The study uses species richness and a wildlife yardstick in estimating wildlife production functions. Species richness estimates do not give information on the importance for wildlife conservation of the species found, whereas the yardstick incorporates the protection need of the species therefore presenting a more appropriate estimate of wildlife production for protection purposes. The results of this study show that there is no significant relationship between yardstick values and area (plot size) while on the other hand plot size was significantly influencing species richness. Higher variability among yardstick values and the limited number of cases may have contributed to such. Furthermore the incorporation of both plot size and distance to the field edge in the same regression may play a role. However ignoring these variables in the regression model would have produced biased estimates for the regional and crop parameters.

In terms of species richness, nature mix fallow was found to be most beneficial to wildlife production when compared to other crops and fallow alternatives. When yardstick values are considered, unsprayed cereals are the most beneficial crops. Sugar beet and potatoes were found to be least beneficial, both in terms of species richness and yardstick value. This study however, considered only one species group (vascular plants). For a complete view of wildlife in farming other species groups need consideration as well.

European policy presents a trend in agricultural wildlife conservation from effort-based payments to incentives and payment schemes linked to wildlife production. For farmers the main advantage of these wildlife output based systems is that it allows them to choose their own management strategy to obtain a certain wildlife level. For policy makers the system guarantees (wildlife) value for money. As non-observed farm specific conditions are having a significant impact on wildlife production, the output based payment schemes are particularly attractive to farmers that already have favourable conditions for the presence wildlife. Rather than introducing wildlife into new areas this ensures preservation and enhancement of existing wildlife.

Further research on wildlife production functions in agriculture should focus on dynamics. In particular, incorporating the development of wildlife production over time will provide more insights into the complex factors that cause variations in wildlife production in agriculture.

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CHAPTER 4

LOCATION-SPECIFIC MODELLING FOR OPTIMISING WILDLIFE MANAGEMENT ON CROP FARMS[5]

Abstract

In order to guide conservation and restoration of wildlife in agricultural areas research is needed into the trade-off between wildlife and agricultural production and income. This study presents a location specific model for optimising wildlife management on crop farms using the integer programming technique. Available data and indicators of wildlife production are presented. Furthermore, time and location aspects of wildlife management are discussed. The model is applied to crop farming in the Netherlands. Most important model outcome is a wildlife-cost frontier at the farm level. Model outcomes show that rotating wildlife conservation practices across the farm is economically more attractive than fixed-location practices. Opportunities for use of the insights provided by model results by both policy makers and farmers are analysed.

4.1 Introduction

Agriculture does not only produce food and fibre; it also helps shaping the rural environment. Increasingly, modern society values the environmental benefits which may arise as joint outputs with primary land use, including e.g. semi natural habitats and wildlife. In Western Europe, rapid changes in primary land use have jeopardised the supply of these benefits (Lowe and Whitby, 1997). Specialisation by region and within individual farms, as well as intensification, through use of fertilisers and pesticides, have increased. Land amelioration (viz. defragmentation, exchange of land, alterations in accessibility) has also contributed to such. These developments have resulted in a loss of habitat for many wild species, and consequently a rapid decline in numbers and populations. The Common Agricultural Policy has been criticised for supporting these changes and over the last decade European policy makers have begun to respond to such criticism. EU-regulations 1760/87 and 2078/92 mark the

[5]Van Wenum, J.H., G.A.A. Wossink and J.A. Renkema (2001) Location-specific modelling for optimising wildlife management on crop farms. Submitted *Ecological Economics*

acceptance that supporting farmers to conserve wildlife and countryside might help to curb overproduction. These regulations also promote a specific approach: supplementary to a distinct geographical segregation of agricultural and wildlife functions both functions should to a large extent blend within the rural environment. While nature reserves will always be important, there is a shift of attention increasingly to the preservation of biological diversity within the major forms of primary land use (Edwards and Abivardi, 1998). This transformation of agricultural policy being an agri-food policy to more of a countryside and wildlife policy calls for investigation of the mechanisms that would help satisfy the following criteria (Lowe and Whitby, 1997): that payments are targeted to ensure cost-effectiveness; that the level and targeting are responsive to public demands; that the benefit is clearly tangible. The first step towards an effective policy to conserve and restore wildlife in agricultural areas, is investigation into the trade-off between wildlife and agricultural production and income. In this task agricultural economics has an important role to play.

The interactions between agricultural production and wildlife and associated decision making are most pronounced at the farm level. The objective of this study is to present and apply a model that enables the assessment of a wildlife-costs frontier at the farm level: i.e. the definition of best (least cost) management strategies for obtaining different wildlife production levels. Such an optimisation model has to account for both time and location specific aspects of agricultural production and wildlife. This particularly applies to crop farming where the production situation differs from year to year due to crop rotation.

Many empirical studies focus on the economics of preventing losses in agricultural yields due to wildlife, for example the body of literature on crop protection and recent work on pre-emptive habitat destruction under the ESA (Lueck and Michael, 1999). In contrast, little work has been done on modelling the production relationship between agricultural practices and wildlife at the farm and field level. Previous ecological and economic studies of wildlife management at the farm level have generally focused on the impact of land use regimes on farm income and biodiversity. For example, the positive effects of refraining from pesticide in northern European agriculture on the abundance of flora and fauna was reported by e.g. Rands (1985), Tew *et al.* (1992), Boatman (1994) and by De Snoo (1997). Economic studies at the whole farm level generally involve a comparison of specific land use regimes by analysis of accounting data and/or farm level modelling (e.g. Van Eck *et al.*, 1987; De Boer, 1995). None of the studies mentioned pays attention to the dynamic and location aspects of the joint production of agricultural outputs and wildlife. Wildlife production, however, not only depends on present management practices but also on management practices in previous year(s). Also, wildlife production depends highly on site specific biophysical conditions and on location aspects such as the distribution of conservation activities like hedgerows and unsprayed field margins in agricultural areas.

The literature on the location aspects of agricultural production and the environment generally focuses on optimal pollution control in relation with water quality of an agricultural watershed: e.g. Braden *et al.* (1989); Braden *et al.* (1991); Moxey and White (1994); Lintner and Weersink (1996). This location dimension, however, is also important in the case of positive externalities of agricultural production, i.e. wildlife. Ecologically, the spatial distribution of species is important for their changes of propagation. Economically, the 'where' question is of importance because of the advantages of selective control, i.e. protecting where it is most effective and least costly. Selective control requires identification of the most effective wildlife management options and also where to apply these. Studies in the field of site selection are virtually all carried out on a regional level and identify the smallest number or cheapest set of sites to realise targeted wildlife criteria; see Csuti *et al.* (1997); Wossink *et al.* (1999). To our knowledge, studies on the location specific aspects of wildlife preservation at the farm level have not been reported in the literature.

The outline of this chapter is as follows: section 4.2 presents an overview of the interactions between agriculture, and more specifically crop farming, and wildlife. Management options for promoting wildlife in agricultural areas are discussed. Section 4.3 presents a generic model for optimal wildlife management on crop farms. Next section 4.4 presents the requirements for implementation of the model. An application of the model for Dutch crop farming is presented in section 4.5. Finally, section 4.6 discusses opportunities for use of the model results to support decision making by farmers and policy makers.

4.2 Crop farming and wildlife

The interactions between agricultural practices and the presence and abundance of wildlife are complex. Two major developments in agricultural practice have caused a reduction in the state of wildlife the last three decades. The use of chemical inputs, in terms of pesticides and nutrients, and monocultures of crops have left little opportunities for wildlife survival. Emissions of chemical inputs to non-agricultural habitats have also contributed to such. Furthermore the number of non-agricultural habitats on the farm is reduced through field enlargement, merging of farms etc., decreasing the chances of survival for wildlife.

Research into ways of enhancing wildlife in arable farming has predominantly focused on unsprayed and/or out of crop field margins and on alternative management of fallow land (Boatman and Sotherton, 1988; De Snoo, 1997). Especially field margins receive much attention. Yields in margins, especially on headlands are often lower due to a higher pest and weed pressure, soil compaction or shady conditions (De Snoo, 1995). At the same time, wildlife abundance is higher in margins, owing to the unfavourable growing conditions for agricultural crops and the location often next to non-agricultural biotopes such as ditches or woodland. From an agricultural point of view, enhancing wildlife in field margins may cause yield reductions in the centre of the field due to weed invasion and wildlife damage. On the other hand positive impacts of unsprayed field margins are reported through biological control of pests in the fields (Boatman and Sotherton, 1988; De Snoo, 1997). However, no information is available on whether these positive effects outweigh negative agronomic effects. Fallow land offers special opportunities for wildlife as in general no chemical inputs are used. Furthermore financial compensation may be obtained through the EU-set aside scheme. However, when set aside is applied in margins a minimum width of 20 m is necessary for obtaining financial compensation (MINLNV, 2000).

Apart from alternative management of field margins and fallow land other opportunities for enhancing wildlife in crop farming are available. Winter cover crops are used in agriculture to save nutrients and for maintaining organic matter content in the soil. For wildlife these crops may provide cover and food during the winter period. Furthermore, non-agricultural habitats on the farms may receive alternative management aimed at enhancing wildlife. Ditch banks offer special opportunities for vegetation development by creating a poor nutrient situation. Rough vegetation may be created on these banks providing cover and nesting opportunities for mammals and birds.

Various wildlife-enhancing activities are thus available at the farm level, each with specific cost and wildlife features, depending on the location on the farm, crops grown, and crop rotation. Incorporating wildlife in farm modelling therefore is rather complicated. The next section presents a theoretical economic model to optimise wildlife management at the farm level taking into account the various optional activities, and the spatial and dynamic interactions.

4.3 Generic model

The theoretical model meets two criteria (Braden *et al.*, 1989): (1) it accounts for the effects of management restrictions on wildlife production at the farm level; and (2) it identifies the pattern of management activities on the farm that maximises farm income over a predefined time horizon. Index $t=1, \ldots, T$ denotes time periods and index $j=1, \ldots, J$ denotes the number of management units recognised on the farm (e.g. field margins, field centres, ditches etc.). Let f denote the production relationship between agricultural inputs and outputs, and let g denote the relationship between agricultural inputs and wildlife outputs (See also Van Wenum *et al.*, 2001).

$$Max\, Z = \sum_{t=1}^{T} \sum_{j=1}^{J} q' y_{jt} - C(y_{jt}, r_t, x_{jt}; l_j) \qquad [4.1a]$$

s. t.

$$f_{jt}(y_{jt}, x_{jt}; l_j) \leq 0 \quad \forall\ j,t \qquad [4.1b]$$

$$g_{jt}(x_{jt}, u_{jt}; l_j) \leq 0 \quad \forall\ j,t \qquad [4.1c]$$

$$\sum_{t=1}^{T} \sum_{j=1}^{J} u_{j,t} \geq N \qquad [4.1d]$$

and

$$x_{jt} \in X_{jt} \quad \forall\ j,t \qquad [4.1e]$$

Z = farm profit
C = cost function
q = vector of prices of agricultural outputs
y = vector of marketed outputs
r = vector of prices of agricultural inputs
x = vector of farm specific management activities
u = vector of wildlife production scores
X = set of all management activities
l = vector of bio-physical and other location specific characteristics (production environment)
N = wildlife production level at the farm level

The production relationship, *f(.)*, between agricultural inputs and outputs [4.1b] is location (*j*) and time (*t*) specific. Yields among locations, even within the farm and within fields vary and for multi-year cropping variants both inputs and outputs between the years as well as associated gross margins may differ. The production relationship, (g), between agricultural inputs and wildlife outputs [4.1c] again is location and time specific. Wildlife varies across locations and across time. The latter specifically counts for multi-year fallow where wildlife may develop or change over time.

Solving the equation set yields x^*, the vector of agricultural management activities including management restrictions that satisfies the requirement for wildlife conservation as expressed by N. Varying N gives a wildlife conservation costs frontier $Z(N)$ for the total farm studied, that is the change (decrease) in farm profit, Z, associated with producing specific levels of N.

4.4 Implementation of the generic model

4.4.1 Agricultural production function

Implementation of the generic model requires information on the production relationship between agricultural inputs and outputs, $f(.)$, see equation [4.1b]. The production level of agricultural outputs, in terms of marketed product(s) per hectare, y, is determined by the production environment, l, and by production techniques and methods applied as expressed by the activity set, X. The activity set is predominantly determined by the farming strategy applied, i.e. organic, integrated or conventional farming and by farm specific constraints such as the availability of labour and machine equipment. Given a farm specific activity set and known production environment, different input/output relationships for various crops can be estimated. Data to such may be obtained from farm accountancy data networks and/or experimental stations.

4.4.2 Wildlife production function

Implementation of the generic model further requires information on the relationship, $g(.)$, between agricultural inputs x_{jt}, and wildlife results u_{jt}, see equation [4.1c]. The production environment, l, at a specific site represents the setting for wildlife production and is characterised by (bio)physical factors that include climate and aspects of the soil (groundwater table, type of soil). Furthermore wildlife production is determined by site specific factors not controllable by management: (a) weather during the growing season (solar radiation, rainfall, temperature), (b) factors due to management in the past such as the level of eutrophication and desiccation of the soil, presence of vegetation remnants and extent of the flora seed bank. The biophysical factors together with the non-controllable factors (production environment) thus determine the 'potential wildlife yield' on a farm. The extent to which this potential level is achieved in practice depends on growth factors that are controllable by management, x. These include crops selected (including fallow), rotation, size and spatial pattern of field and field margins as well as nutrient management, water management and pest control. Together with the production environment these factors determine 'actual wildlife yield' (Turner *et al.*, 2000). Whereas agricultural outputs are easy to quantify and measure in terms of marketable yields y_{jt}, wildlife results, that is u_{jt} in equation [4.1c], are much more difficult to assess. A direct measurement of the presence and abundance of all wildlife on a farm is not feasible; therefore indicators of wildlife production have to be used.

Within the OECD work on agriculture and the environment, *p*ressure, *s*tate and *r*esponse indicators are recognised (PSR-framework, see OECD, 1994). Recently this framework has been applied to agriculture and biodiversity. Pressure indicators are measurements of agricultural activities that cause changes in the state of biodiversity such as the use of pesticides and fertiliser. State indicators are direct measurements of the state of biodiversity arising from these pressures, in terms of species, habitats or environmental parameters. Finally response indicators refer to responses by farmers, government or society to changes in the state of biodiversity, such as the use of financial incentives to farmers to enhance biodiversity.

Obviously for solving the normative generic model and implementation of the wildlife production function, indicators of the state of biodiversity, are needed. Main requirement for equation [4.1c] of the generic model to be implemented is an indicator applicable at the farm level to provide a complete picture of the state of wildlife. Furthermore the relationship of the indicator outcomes with farm management practices has to be clear.

Many attempts have been made to indicate wildlife and biodiversity. The term biodiversity in the sense of the Biodiversity Convention of the UN Conference on Environment and Development in Rio de Janeiro (1992) encompasses the whole range of the genetic diversity within species, the diversity of species and higher taxa, up to ecosystem diversity, and even the diversity of ecological interactions. Clearly the Rio convention focuses on the more complex qualitative aspects of biodiversity. Quite obviously, such broad diversity of life cannot be measured in a comprehensive manner (Duelli, 1997).

The traditional scientific quantitative concept of biological diversity is based on species diversity. Indexes considering the quantitative aspects of biodiversity are often constructed as a function of species counts and the relative abundance of species (Magurran, 1988). Others have looked at evenness: biodiversity measures as a function of genetic distances among members of a species set (Weitzman, 1992; Solow, Polasky and Broadus, 1993). Species richness is the simplest form of these measures neglecting differences in abundance or genetic distance. Species richness provides an extremely useful measure of diversity if the study area can be successfully delimited in space and time and the constituent species enumerated and identified.

In this study two indicators for wildlife production are used, considering vascular plants only. The species richness indicator (in terms of species density) is compared with an extensive species based indicator specifically developed for agriculture: the wildlife yardstick (see Buys, 1995; Van Wenum *et al.*, 1998). Whereas species richness considers the density of species, this yardstick provides information on the ecological or protection value of species. Another reason for using this indicator is its application in proposed future measures on agricultural wildlife conservation in the Netherlands. The use of two indicators also enables analysis of the impacts of indicator choice on the selection of optimal management strategies (Eiswert and Haney, 2001).

The wildlife yardstick for vascular plants consists of a representative set of species. This representative set was put together for simplicity reasons. However to gain a more complete picture this study considers all plant species. To each plant species now, a rating V (0-100 points) has been assigned based on its protection need as determined by rarity, population tendency and international importance (all three at the national level).

A wildlife score per area measure (U) now is calculated as the sum of ratings of all plant species r found for the respective area measure regardless of the number of plants per species (Van Wenum *et al.*, 2001):

$$U = \sum_{r=1}^{R} V_r \qquad [4.2]$$

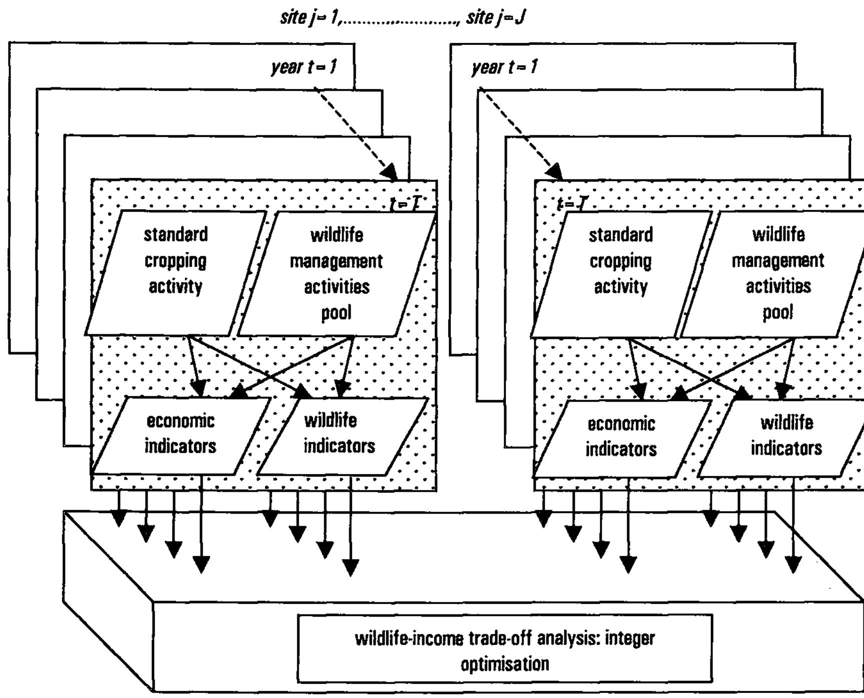
When species richness is considered, a wildlife score per area measure, U, simply is the number of species found.

To implement the indicators into the generic model, data are needed on the presence of plant species for all management activities X given site characteristics l. Research into the relation between agricultural management and wildlife however, usually takes into account a limited number of management options. Also assessments for consecutive years are scarce. Furthermore research is carried out on different locations, with inconsistent location specific conditions hampering a comprehensive analysis. Van Wenum *et al.* (2001) however, presented a functional form and estimation technique for a wildlife production function at the farm level. A random effects model was developed to capture the relationship between wildlife output, management practices, regional conditions and non-observed farm specific factors. The study used species richness and the wildlife yardstick (both considering vascular plants) in estimating wildlife production functions.

4.4.3 Optimisation procedure

A schematic representation of the optimisation procedure is presented in Fig. 4.1. In order to model and optimise wildlife management, the farm is divided in spatial units ($j=1, ..., J$). In a conventional farming situation, different crop fields and non-productive biotopes such as woodland and ditches can be observed. Management on crop fields or within a non-productive biotope type will normally be uniform. Incorporating wildlife management options may result into more activities per field and thus an increase in the number of spatial units to be recognised. Each recognised spatial unit is assumed to have uniform conditions and management. Therefore it is necessary to formulate the model in an integer context. Management activities now are integers forcing the model to select only one management activity per spatial unit j per year t.

Scheme 4.1 Optimisation procedure



Solving the equation set from section 4.3 may require a model of considerable size due to the integer context of the problem. Also other factors may affect model size: (1) the length of the planning period, (2) the number of management units (sites) recognised, (3) the number of management alternatives to each unit and (4) the combinatorial complexity of the problem. We discuss these aspects in more detail.

Ad (1): Decisions regarding incorporating wildlife management are made on the tactical and strategic level. For the present study a planning horizon of one rotation (usually lasting 4 years) is considered appropriate. Impacts of past activities influencing wildlife and or agricultural production in following years can therefore be incorporated in the model.

Ad (2): Without specific attention being paid to wildlife management, an individual field (including margins, headlands and centre) will generally be treated uniformly. However with the introduction of wildlife management alternatives, management on field margins may be different from the field centre. Moreover distinction between headlands and longitudinal sides should be made for their differing agronomic and economic features. Besides non-agricultural habitats need separate consideration.

Ad (3): When all available activities may be applied on each site the selection problem is huge. Therefore it looks appropriate to define an optimal baseline situation, considering crops and whole fields only. After this baseline run, for each site the standard crop activity is known and wildlife management alternatives may be defined for new optimisations to be carried out.

Ad (4): Combinatorial aspects have to do with the influence of past on present activities on sites, and with activities on certain sites influencing the wildlife or agronomic situation on other sites. Furthermore farm level constraints on top of site constraints add to the combinatorial character of the model.

4.4.4 Model output

The most important outcome of the model is a wildlife-cost frontier at the farm level. For each wildlife production level N, the associated set of management activities that maximises farm income is defined. Due to the nature of the applied LP-model, this frontier is a piecewise linear function where each step corresponds to a particular basic solution to the income-maximising problem. This means that the objective function is not continuously differentiable. So rather than $\partial Z/\partial N$ (see section 4.3), $\Delta Z/\Delta N$ needs consideration, where ΔN is a discrete change in wildlife production.

4.5 Application of the model

4.5.1 Representative farm

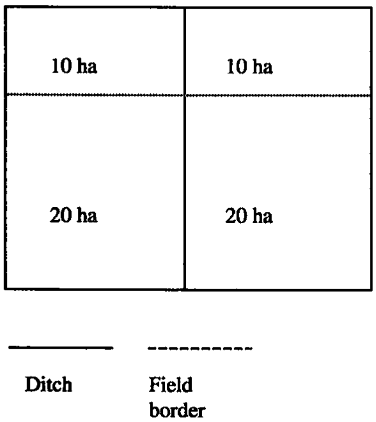
A representative crop farm type was chosen for a first application of the model as presented in Scheme 4.1. The crop farm is representative for the central clay area in the Netherlands. Parcellation of farms in this area is relatively simple and the number of crops grown on these farms is limited. Therefore this area is ideal for a first application of the model. The cropping plan of the farm is presented in Table 4.1.

Table 4.1 Cropping plan of representative farm

Crop	Acreage (ha)
ware potatoes	20
winter wheat	20
sugar beet	12
Onion	6
Fallow black	2
TOTAL	60

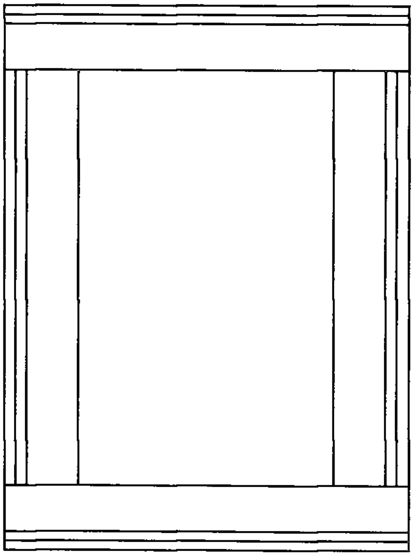
The representative farm has a cropping plan based on a 3-year rotation with one-third of the acreage planted with potatoes. The farm consists of two blocks of 30 ha, typically for the considered region. The blocks are subdivided into 4 fields (2 of 20 ha and 2 of 10 ha respectively). A graphical representation of the farm is presented in Scheme 4.2.

Scheme 4.2 Spatial layout of the representative farm



For the initial situation crops were assigned to the fields for 4 years for the representative farm. Each field was subdivided into 13 spatial units enabling the introduction of 3 m, 6 m and 20 meter wide margins on each side of the field. A decomposed field is presented in Scheme 4.3.

Scheme 4.3 Decomposed field



4.5.2 Model input

For each spatial unit in the model next to the baseline cropping activity, alternative wildlife management activities were offered in the optimisation procedure. Application of unsprayed cereals was restricted to 3 and 6 meter margins whereas fallow alternatives may also be applied in 20-meter margins and on whole fields. Furthermore fallow alternatives may be applied for 1 year or for 2, 3 or 4 years consecutively on the same field or margin. No other permanent cropping variants were offered as most crops require rotation to prevent yield losses from soil born diseases. Table 4.2 gives an overview of all considered alternatives.

Table 4.3 presents gross margins for all available crop and wildlife management activities for both margins and field centre. Gross margins for the field centre were obtained from PAV (1997). For field margins lower yields were assumed and gross margins were calculated accordingly. Yield reductions were obtained from De Snoo (1995), Schoorlemmer (1998) and Van Bemmelenhoeve Research Farm. For fallow variants, Table 4.3 presents gross margins for the first year. With the exception for natural vegetation gross margins for the 2nd to 4th year are higher as seed costs are only applicable in the first year.

Table 4.2 Overview of cropping and wildlife activities and sites applicable

Activity	Field margin			Field centre	Whole field
	3m	6m	20m		
ware potatoes				x	x
winter wheat				x	x
Sugar beat				x	x
onion				x	x
Seedgrass				x	x
Phacelia fallow	x	x	x	x	x
grass-clover	x	x	x	x	x
nature mix fallow	x	x	x	x	x
Natural vegetation	x	x	x	x	x
Unsprayed winter cereals	x	x			
Unsprayed spring cereals	x	x			

Table 4.3 Gross Margins (NLG/ha) of cropping variants and wildlife activities for spatial field units

Activity	Spatial field unit					
	0-3m		3-6m		6-20m	centre
	head land	length side	Head land	length side		
ware potatoes	4596	5626	5626	6141	6656	6656
winter wheat	2203	2585	2585	2776	2967	2967
Sugar beat	3838	5084	5084	5707	6330	6330
onion	2178	4250	4250	5286	6322	6322
Seedgrass	1292	1640	1640	1814	1987	1987
Phacelia fallow*	-175	-175	-175	-175	-175	-175
grass-clover*	-140	-140	-140	-140	-140	-140
nature mix fallow*	-205	-205	-205	-205	-205	-205
Natural vegetation*	-8	-8	-8	-8	-8	-8
Unsprayed winter cereals	2008	2302	2302	2449	2596	2596
Unsprayed spring cereals	1303	1577	1577	1714	1851	1851

*) Gross margin in first year, excluding EU-MacSharry premium for set aside land. Premium is only applicable for set aside fields or set aside field margins with a minimum width of 20m.

Wildlife scores, U, for each activity were obtained from Van Wenum et al. (2001): Species richness and wildlife yardstick values of vascular plants were estimated using a random effects procedure. The following model was estimated:

$$U_{it} = M_i + \sum_{k=1}^{8} \alpha_k A_{ikt} + \sum_{i=1}^{2} \beta_k S_{ik} + \gamma_1 D_{it} + \gamma_2 D_{it}^2 + \lambda \log P_{it} \qquad [4.3]$$

where U_{it} is wildlife, measured either as species richness or as yardstick value. M_i is an unobservable farm specific management variable for the i^{th} farm, A_{ikt} denotes a dummy variable for agricultural management activities of the i^{th} farm at time t with k=1 (grass-clover), 2 (nature mix fallow), 3 (natural vegetation), 4 (unsprayed winter cereals), 5 (unsprayed spring cereals), 6 (potatoes), 7 (sugar beet) and 8 (phacelia fallow). The dummy variables A_{ikt} take the value 1 if activity k is present at time t at farm i and 0 otherwise. S_{ik} are regional dummy variables with k=1 (northern clay area) and 2 (central clay area) that take the value 1 if the j-th region applies and zero otherwise. The northern sand area is the reference area in this regression, i.e. S_{ik} is zero for all i,k in the northern sand area. D_{it} represents the distance in meters from the sampling spot to the edge of the field. The quadratic specification allows for both increasing and decreasing marginal effect of distance on wildlife production.

For this study wildlife scores (both species richness based and yardstick based) were calculated for each activity. Equation 3 was used to this end with the following input parameters: central clay region, plot size of 100 m^2 and distance to the field edge of 1,5 m for the 3 meter margins, 4,5 meter for the 3-6m margin units and 6 meter for the central units of the field. Table 4.4 presents the wildlife scores for the different activities. An average farm specific factor *M* was assumed (value 0). No distinction was made between multi-year and one year fallow variants because wildlife data for more permanent activities were lacking. Therefore, wildlife scores were assumed to be constant over years.

Table 4.4 Wildlife scores of cropping variants and wildlife activities for spatial field units (management factor = 0)

Activity	Species richness			Yardstick value		
	0-3m	3-6m	6-20 m/ field centre	0-3m	3-6m	6-20 m/ field centre
Ware potatoes	-0.7	-1.3	-3.1	66.5	67.1	47.4
Winter wheat	-2.7	-3.2	-5.0	71.2	71.7	52.1
Sugar beat	-4.6	-5.2	-7.0	75.9	76.4	56.7
Onion	-2.7	-3.2	-5.0	71.2	71.7	52.1
Seedgrass	-2.7	-3.2	-5.0	71.2	71.7	52.1
Phacelia fallow	5.9	5.3	3.5	63.7	64.3	44.6
Grass-clover	6.4	5.8	4.0	109.5	110.0	90.3
Nature mix fallow	12.9	12.3	10.5	103.3	103.9	84.1
Natural vegetation	11.8	11.2	9.4	111.6	112.2	92.5
Unsprayed winter cereals	9.9	9.3	7.6	132.4	132.9	113.2
Unsprayed spring cereals	11.7	11.1	9.3	137.1	137.6	117.9

4.5.3 Model results

The baseline situation was calculated using the cropping plans of Table 4.1. Total Gross margins (per year) for the baseline situation of the representative farm is NLG 313998. In the baseline situation the species richness indicator valued -262 per year and the wildlife yardstick valued 3288 per year. A stepwise increase of wildlife scores was imposed and Total Gross Margins were obtained through optimising the model. Wildlife cost frontiers for the farm using species richness indicator and yardstick values are presented in Fig. 4.1 and Fig. 4.2 respectively. No big leaps in the frontier, characteristic for integer optimisations, are observed. The considered four-year period and the large number of spatial units recognised, give the model a large number of opportunities to keep the step width limited.

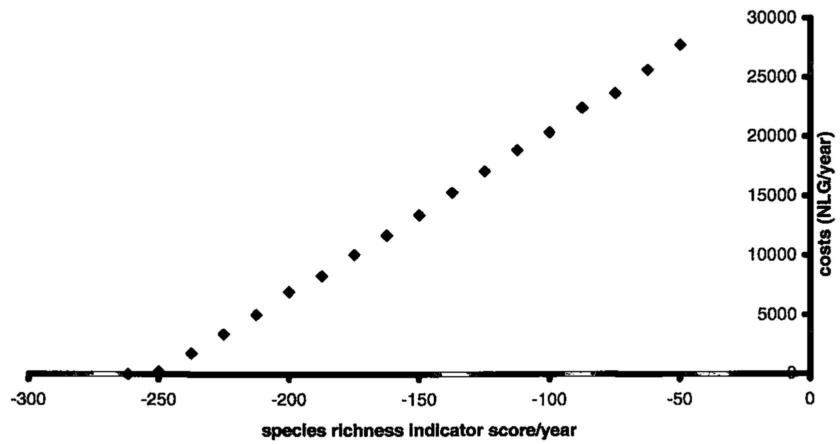


Fig. 4.1 Wildlife-cost frontier (species richness indicator based) for the representative farm

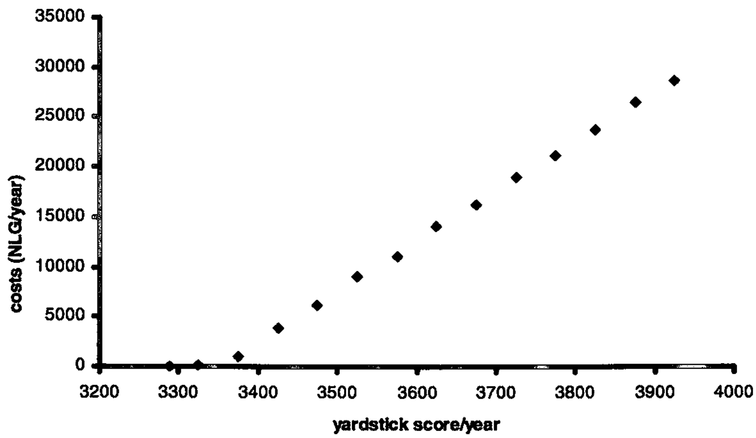


Fig 4.2. Wildlife-cost frontier (yardstick score based) for the representative farm

When a stepwise increase of species richness indicator or yardstick values is imposed Total Gross Margin for the representative farm is dropping. When small increases are imposed conventional cereal margins are replaced by unsprayed cereal margins. Table 4.5 and 4.6 show results of optimisations when larger increases are imposed. Optimisation 1 in both tables results into a similar cost level and the same accounts for optimisation 2. This therefore enables both indicators to be compared on their resulting management strategies for the farm. Furthermore it helps understanding the species richness and yardstick score levels by showing the activities that are replaced and by comparing the indicator values of the replaced and the new activity. From Table 4.5 it is clear that optimisation 1, using the species richness indicator, predominantly leads to replacing wheat fields and margins by unsprayed margins and natural vegetation. When the yardstick is used, for a comparable cost level, also margins of other crops such as ware potatoes were replaced (Table 4.6, optimisation 1). This indicator therefore results in a larger network of field margins at a similar cost level. This pattern was also visible for other optimisations at slightly higher and lower cost levels.

Table 4.5 Results of 2 optimisations with imposed species richness scores (replaced crops in parentheses)

Year	Optimisation 1 Imposed species richness score: -175/year Costs: NLG 10090/year	Optimisation 2 Imposed species richness score: -113/year Costs: NLG 18850/year
Year 1	+ 0.54 ha 6m margins wheat unsprayed (wheat) +1.80 ha 20m nature mix fallow (wheat, fallow black)	+20 ha natural vegetation (wheat)
Year 2	+1.50 ha 6m margins wheat unsprayed (wheat) +0.80 ha 20m nature mix fallow (fallow black)	+20 ha natural vegetation (wheat)
Year 3	+20 ha natural vegetation (wheat)	+ 0.50 ha 6m margins wheat unsprayed (wheat) + 0.30 ha 3m margins wheat unsprayed (wheat)
Year 4	+ 1.08 ha 6m margins wheat unsprayed (wheat)	+ 0.24 ha 6m margins wheat unsprayed (wheat) + 0.40 ha 3m margins wheat unsprayed (wheat)

A further increase in indicator values up to the levels of optimisation 2 in Table 4.5 and 4.6 reduces the differences in results between both specifications. Both optimisations show outcomes where wheat fields are replaced by natural vegetation and margins are altered to unsprayed variants. An interesting result of the optimisations is that no multi-year fallow alternatives are used. It can therefore be concluded that rotating wildlife activities across the farm is more attractive than permanent activities.

Furthermore at high wildlife levels crops with low gross margins, especially the cereals, are replaced and the more intensively grown crops are not affected. Intensive cropping plans with low proportion of cereals will therefore result into higher costs for enhancing wildlife levels. Crops like potatoes and sugar beet that have higher gross margins and a higher use of inputs (fertiliser and crop protection agents) than cereals will then have to be replaced by wildlife activities resulting in higher costs to obtain similar wildlife levels.

Table 4.6 Results of 2 optimisations with imposed yardstick scores (replaced crops in parentheses)

Year	Optimisation 1 Imposed yardstick score: 14300/year Costs: NLG 11110/year	Optimisation 2 Imposed yardstick score: 14900/year Costs: NLG 18950/year
Year 1	+0.60 ha 3m margins barley unsprayed (sugar beet, onion, ware potatoes) +20 ha natural vegetation (wheat)	+1.08 ha 6m margins wheat unsprayed (wheat)
Year 2	+ 1.5 ha 6m margins wheat unsprayed (wheat) +0.24 ha 3m margins barley unsprayed (ware potatoes)	+1.02 ha 6m margins wheat unsprayed (wheat) +0.56 ha 3m margins wheat unsprayed (wheat)
Year 3	+1.08 ha 6m margins wheat unsprayed (wheat) +0.24 ha 3m margins barley unsprayed (ware potatoes)	+20 ha natural vegetation (wheat)
Year 4	+1.08 ha 6m margins wheat unsprayed (wheat) +0.60 ha 3m margins barley unsprayed (sugar beet, onion, ware potatoes)	+20 ha natural vegetation (wheat)

4.6 Discussion

The model presented gives farmers more insight and a better understanding in selecting best management practices to obtain different wildlife production levels. Furthermore the model outcome gives policy makers information on costs associated with different wildlife production levels. Incentive development and cross compliance instruments may therefore benefit from the model outcome. However before using the outcomes for policy design a study on the acceptance of the proposed wildlife activities is necessary as perceptions and preferences among farmers towards wildlife conservation may vary.

Model results indicate that rotating of wildlife activities across the farm, mainly following the cereal crops is most attractive: wildlife scores are thus obtained at lowest cost. The model however assumed uniform conditions across the farm, whereas in practice conditions between fields and also within fields may significantly differ opening opportunities for permanent coverage with wildlife activities. Furthermore wildlife scores for permanent fallow activities were held constant. With a positive wildlife development over time, multi-year fallow also becomes more attractive. However, multi-year fallow implies that also crops with high gross margins will be replaced and that this type of wildlife activities therefore will be costly. Connectivity of wildlife activities was not considered in this study. However by forcing the model to leave field centres in tact and allowing only margins to change to wildlife activities, the spread across the farm, and the chances for connectivity, would be better with increasing wildlife levels.

A bottleneck of the model presented is the availability of data on the relationship between agricultural practices and wildlife indicators. More data from ecological research under various conditions will increase the reliability of the model outcome. The model is further restricted by including a fixed spatial arrangement of fields and non-agricultural elements on the farm, limiting the number of wildlife options to be considered. Another drawback of the model is its limitation to wildlife as the sole externality of farming. If other environmental externalities had been included, such as pesticide use, the focus would probably shift from cereal replacement to replacing potatoes and sugar beet (margins) being crops with higher pesticide use.

A linear relationship was assumed between indicator values and acreage: two hectares of a certain activity with a certain wildlife indicator score (either species richness based or yardstick based) had twice the score of one hectare of the same activity. Within farms this linear relationship may hold, however on the regional level the wildlife value of yet another hectare of the same activity may have a lower value to wildlife. Further research in this field is advised. The same counts for the development of wildlife over time, especially for multi-year activities as data in this field are scarce.

Many of the private initiatives currently taken to enhance wildlife in agricultural areas depart from co-operation of farmers on a regional level. When considering an analysis on a regional scale spatial connections e.g. linking of important ecological objects (ecological networks) needs special attention (Lintner en Weersink, 1996; Wossink *et al.*, 1997). An optimisation to be carried out on a regional scale may well lead to different contribution efforts by farmers to meet the regional determined wildlife objectives. Equity among participants therefore also needs special attention (Önal *et al.*, 1998). The model presented here does not account for these two aspects. However, the farm specific outcomes of the model may well serve as a basic input for aiding decision making on a regional scale. In this respect Walpole and Sinden (1997), offer an interesting approach using farm level benefit-cost ratios and GIS predictive modelling, to aid land degradation management on a regional scale. Such an approach would also offer great potential for supporting regional wildlife management decision making.

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CHAPTER 5

WILDLIFE CONSERVATION BY FARMERS: ANALYSIS OF ACTUAL AND CONTINGENT PARTICIPATION[6]

Abstract

This chapter examines actual and contingent participation of Dutch crop farmers in wildlife conservation programs. Probit and tobit modelling were used to analyse the effect of farm and farmer characteristics and farmer attitudes on participation. The optimal bid offer was derived from a referendum CV survey for a proposed field margin program. Actual participation was highest for organic farmers and farmers facing area specific restrictions. Contingent participation was strongly affected by bid offer. Furthermore, specialisation, integrated farming, off farm income sources, risk perception and ditch length positively influenced contingent participation. The CVM-experiment suggested that up to 60 percent participation might be achieved with appropriate bid offers. Implications of the results for policy are discussed.

5.1 Introduction

Recently, wildlife conservation on agricultural land receives much attention from policy makers in the European Union. Land use policies are being developed that pursue both environmental and wildlife objectives. These policies provide incentives to landowners and farmers to maintain the current situation or to convert land to more environmentally benign uses.

In a conservation program determined through (textbook) market interaction, farmers compete for a given conservation budget with self-defined practices through a tender approach. The competition ensures efficient allocation of the budget. Real world conservation schemes rarely take this approach. Instead these programs are basically a combination of incentive based policies and command and control in the sense that fixed amounts are offered for a limited number of approved conservation practices. A judgement of what can be expected from using such policies requires a representation of both the command and control and the incentive component and a

[6]Van Wenum, J.H. and G.A.A.Wossink (2001) Wildlife Conservation by farmers: analysis of actual and contingent participation. Submitted *European Review of Agricultural Economics*

detailed representation of the activities these real world policies intend to target (Schwabe and Smith, 1998). Specifically, a lack of participation in these programs may be due to either the incompatibility of the approved conservation practices with farm and farmer characteristics or to an inappropriate amount of incentive payments. This paper examines both aspects.

Exactly which factors influence farmers' participation in voluntary agri-environmental schemes is not yet fully understood. The literature on the adoption of sustainable agricultural practices is used to guide the formulation of the research model for this study. According to D'Souza *et al.* (1993) factors affecting adoption of sustainable agricultural practices can be grouped under human capital (e.g. age and education), structural (e.g. farm size, debt/asset ratio), institutional (e.g. participation in farm commodity programs) and environmental categories (e.g. awareness of environmental problems). Wilson (1997) following Brotherton (1989, 1991) states that both 'scheme factors' and 'farmer factors' need to be taken into consideration when attempting to understand farmer participation in agri-environmental schemes. Scheme factors include payments offered, duration and voluntary nature of the scheme. Farmer factors include various individual farm and farmer characteristics. Other studies in more recent years emphasise the importance of farmer's attitudes towards the environment (Morris and Potter, 1995) and how structural and attitudinal factors interplay in the individual farmers' decision making process (*e.g.*, Falconer, 2000). Economic models based on profit maximisation (*e.g.*, Van Wenum *et al.*, 1998) fail to encompass attitudinal variables altogether whereas omission of important explanatory variables that are correlated with variables included in econometric models leads to biased estimators and to invalidation of inference procedures (Greene, 1997). In order to handle this problem a comprehensive utility-based approach to explain program participation is required that integrates normative economic and behavioural aspects, together with institutional and agronomic aspects.

The contingent valuation method (CVM) is the predominant method for analysing opportunities for (new) incentive schemes. Typically consumers' Willingness To Pay (WTP) is measured for the non-market goods associated with agriculture, viz. species conservation, water quality and pastoral landscapes. In contrast, few studies have addressed the concomitant supply side of these environmental goods. Bonnieux and Rainelli (1995) estimated a value for agricultural landscape using the WTA-concept. Farmers were asked what is the minimum amount they needed to be paid in order to implement a specific change in their farming practices. Cooper (1997) also used CVM to estimate the minimum incentive payment farmers would require in order to adopt more environmentally friendly 'best management practices' (BMPs). To predict adoption the CVM data were combined with actual market data on enrolment in BMPs, furthermore Cooper considered intensity of adoption by estimating the acres enrolled as a function of the incentive payment. Only a limited number of farm and farmer characteristics were used in the regressions and none of the studies mentioned, incorporated farmer attitudes and perceptions.

The purpose of this chapter is twofold. Firstly, we intend to assess the factors that explain farmers' participation in existing conservation programs. Secondly, we analyse farmers' contingent participation in a new wildlife conservation program and the acreage enrolled as a function of the incentive payment. A range of farm and farmer characteristics including attitudes and perceptions were considered. Measuring respondent attitudes, as recommended by the US NOAA (National Oceanic and Atmospheric Administration) may help interpret valuation questions (Arrow *et al.*, 1993; Kotchen and Reiling, 2000). Eliciting attitudes toward the environment is expected to enhance CVM, in particular.

The outline of this chapter is as follows. The next section presents the theoretical background followed by an application for field crop farmers in the Netherlands. The new incentive scheme consisted of a voluntary nation-wide field margin program with carefully selected vegetation and management opportunities, reducing the risk of weed and pest problems. The chapter finishes with conclusions and special attention for policy implications.

5.2 Theoretical background

5.2.1 Actual participation

The decision to participate in an existing wildlife conservation program takes the form of a binary variable, which suggests that either a logit or probit model is appropriate. Both type of models relate the dependent and independent variables non-linear, however based on two different cumulative distribution functions (CDF) of the random variable. Whereas the logit model is based on the logistic CDF the probit model is based on the normal CDF. In this study the following probit model is proposed to explain actual participation in conservation programs:

$$a_i^* = \beta_0 + \sum_{j=1}^{k} \beta_j x_{ij} + u_i \qquad [5.1]$$

$$u_i \sim \text{IN}(0, 1)$$

with:

$a_i =$ 1, if $y^*>0$ (farmer i adopted wildlife conservation measures), and

$a_i =$ 0, otherwise

$x_{ij} =$ vector of explanatory variables j ($j=1, ...,k$) for farmer i: farm and farmer characteristics, behavioural aspects

5.2.2 Contingent participation

To analyse contingent participation in a proposed wildlife conservation program both the WTA (receiving compensation for a loss) and WTP format (paying something for a foregone gain) may be used. Respondents however, will be far less familiar with the notion of paying for a foregone gain causing far greater uncertainty and variability in answers to WTP questions than occurs with WTA questions (see also Turner *et al.*, 1994). Therefore WTP was avoided in favour of WTA.

While it is possible to directly elicit farmers' minimum WTA to adopt a conservation practice, the referendum approach, as recommended by the US NOAA Panel, is likely to be preferable (Arrow *et al.* 1993). The dichotomous choice (DC) form of CVM was used to take the referendum approach. Under DC-CVM, the respondent is prompted to provide a yes or no response to a bid amount contained in the valuation question, where the bid amount is varied across the respondents. Compared with eliciting the WTA in an open-ended fashion, this method is particularly likely to reveal accurate statements of value as the format reduces the ability of the respondent to purposely bias the study results (Hoehn and Randall, 1987; Cooper, 1997).

When using the referendum approach, CV responses are binary variables, therefore one needs a statistical model appropriate for a discrete dependent variable. Logit and probit models play a key role in the analysis of discrete CV data. A probit model is proposed to explain farmers' contingent participation. To this end the bid amount is incorporated in the model as an explanatory variable.

$$c_i^* = \beta_0 + \sum_{j=1}^{k} \beta_j x_{ij} + u_i \qquad [5.2]$$

$u_i \sim \text{IN}\ (0, 1)$

with:

$c_i =$ 1, if $z^*>0$ (farmer i accepts the offer), and

$c_i =$ 0, otherwise

$x_{ij} =$ vector of explanatory variables j ($j=1, \ldots, k$) for farmer i: farm and farmer characteristics, behavioural aspects, bid amount

5.2.3 Intensity of participation

When participation in a wildlife conservation program is considered a binary decision, all participants are treated the same neglecting quantity differences among participants: intensity of the participation decision. In case of wildlife programs the maximum intensity is reached when the total available acreage is used for conservation. Intensity of participation is therefore defined as the proportion of the total available acreage that is used for conservation. Since this variable has a censored distribution (values between 0 and 100) a tobit model is proposed to explain intensity of participation:

$$y_i^* = \beta_0 + \sum_{j=1}^{k} \beta_j x_{ij} + u_i \qquad [5.3]$$

with:

$$y_i = \begin{cases} \beta_0 + \sum_{j=1}^{k} \beta_j x_{ij} + u_i & \text{for participating farmers} \\ 0 & \text{for farmers who are not participating} \end{cases}$$

$u_i \sim \text{IN}\ (0, \sigma^2)$

and

$y_i =$ extent of participation farmer i in incentive program

$x_{ij} =$ vector of explanatory variables j ($j=1, \ldots, k$) for farmer i: farm and farmer characteristics, behavioural aspects, bid amount

The relationship between the expected value of all observations, $E[y_i|x_i]$, and the expected conditional value above the limit, $E\left[y_i^* \,\middle|\, x_i\right]$, is given by: $E[y_i|x_i]=$

$\Phi\left(\frac{\beta' x_i}{\sigma}\right) E\left[y_i^* | x_i\right]$, where Φ is the cumulative density function of the standard normal distribution, σ is the standard deviation of the error term, and β and x_i are the vector of coefficients and explanatory variables, respectively. For the purpose of the estimation we are not only interested in β but merely in the marginal effect of all observations. Greene, 1997, p. 965) shows that this marginal effect can be decomposed in two parts:

$$\frac{\partial\left[y_i | x_i\right]}{\partial x_i} = \beta\beta \times \left[\Phi_i\left(1 - \frac{\phi_i \varphi}{\Phi_i \Phi}\left(\frac{\beta' x_i}{\sigma\sigma} + \frac{\varphi\phi_i}{\Phi\Phi_i}\right)\right) + \varphi\phi_i\left(\frac{\beta' x_i}{\sigma\sigma} + \frac{\varphi\phi_i}{\Phi\Phi_i}\right)\right] \quad [5.4]$$

where ϕ is the density function of the standard normal distribution, $\Phi_i = \Phi\left(\frac{\beta' x_i}{\sigma}\right)$ and $\phi_i = \phi\left(\frac{\beta' x_i}{\sigma}\right)$. The relation in equation [5.4] shows that the total effect of a change in x_i consists of two parts: (1) the change in y_i of those above the limit, weighted by the probability of being above the limit, and (2) the change in the probability of being above the limit, weighted by the expected value of y_i if above.

5.3 Application

5.3.1 Review of wildlife conservation in the Netherlands

Until recently, in the Netherlands as well as in other EU member states, wildlife conservation mainly focused on farming areas located within or alongside the so-called Ecological Main Structure (EMS): An ecological network of nature reserves and interconnecting zones. Farmers in and near these areas receive subsidies for a variety of conservation management practices, ranging from extensive cereal growing to the development and maintenance of landscape elements.

The majority of farmers in the EU are located outside the EMS. EU-Regulations 1760/87 and 2078/92 mark the acceptance, that instead of the traditional distinct geographical segregation of agricultural and wildlife functions as in the EMS, both functions should to a large extent blend within the rural environment. Besides, also in ecological circles attention is shifting towards the preservation of wildlife within the major forms of primary land use in addition to nature reserves and other protected areas (Edwards and Abivardi, 1998). In the Netherlands, policy towards these so-called white areas has taken two forms. Incentives for pastoral farming have been introduced which focus on meadow bird protection and alternative ditch bank management. For crop farming areas, conservation activities concentrate on fallow land and field margins. The Dutch Ministry of Agriculture, Fisheries and Nature Management developed an incentive for enhancing wildlife on land that has been set-aside as part of the EU-support regulations for cereal crops. Payments are offered to cover the extra cost associated with the wildlife management including seed and seeding costs of special mixtures of dicotyledons. In 2000, only 190 crop farmers participated with a total of 500 hectares consuming approximately 25% of the total budget available. In addition, provincial authorities developed incentives for field margins. Management opportunities vary across provinces and payments vary accordingly. Participating in these national and regional schemes has been disappointing. Only a limited number of farmers participate in the cost share programs offered for the areas outside the EMS. The most recent data provided by the Central Bureau of Statistics show that 3,3% of specialised crop farmers are involved in activities concerning wildlife conservation (LEI/CBS, 2000). These include both farmers involved in EMS related activities and activities regarding the areas outside the EMS. This number is much lower than for land based animal husbandry (cattle, sheep, etc.) where 8,2% of the farms is involved in wildlife conservation.

Following the literature on the adoption of sustainable agricultural practices as described in the introduction, it is hypothesised that participation in wildlife conservation programs is affected by farm and farmer characteristics, attitudes and scheme factors. More specifically a strong influence of the following factors is expected:

- The production environment on the farm. When less favourable conditions exist on the farm, gross margins of crop production will be smaller and other activities such as wildlife management are relatively more attractive.
- Farm size. Small farms usually grow a larger proportion of (labour) intensive high-returning crops and will therefore be less attracted to wildlife activities. Morris and Potter (1995) found that, when looking at participation in agri-environmental schemes in the United Kingdom, it was the younger farmers with the largest more economically buoyant farms who tended to find schemes attractive.
- Successor situation. It is assumed that farmers without successor are less production oriented and more willing to adopt conservation oriented farming (Potter and Lobley, 1992).

- Familiarity with conservation programs. Not all farmers are aware of the regulations and incentives available for wildlife conservation, hampering participation.
- Societal commitment of farmer. Farmers that are more sensitive to what society wants are expected to be more open to wildlife conservation activities.
- Innovativeness of the farmer. Innovative farmers that like to try new production methods are expected to be less hesitant towards wildlife conservation.
- Risk attitude towards wildlife conservation practices. From other studies (Van der Meulen *et al.*, 1996; Buys *et al.*, 1996) it is known that the perceived risk of weed infestation and spread of pests and diseases is a major factor for not participating in wildlife conservation programs

It is hypothesised that wildlife schemes that have lower weed and disease risk features than existing programs may increase participation rates. Furthermore we expect a higher participation rate and a higher intensity of participation, in terms of the area used for the practice, when payment levels increase.

5.3.2 Survey

A survey was compiled and pre-tested with 8 crop farmers. After minor adaptations the survey was mailed to 1000 farmers from three important crop farming regions in the Netherlands: the provinces of Groningen, Drenthe and Flevoland. 278 questionnaires were returned. After removing questionnaires from test farms, non-crop farms as well as incomplete questionnaires, 250 remained for analysis.

The survey consisted of six parts: (1) general information about the farmer and the farm, (2) detailed information on the production environment of the farm (parcellation, ditches, woodrows), (3) farmer attitudes: towards society, towards agricultural wildlife conservation (risk perception and valuation of positive externalities), and innovativeness by scoring statements on 5-point Likert scales, (4) familiarity with existing wildlife conservation programs, (5) actual adoption of wildlife measures in terms of alternative field margin and fallow management, and (6) contingent participation: a fictitious field margin practice was introduced to the farmer (Table 5.1) and his or her Willingness to Accept was analysed using the referendum approach. As discussed in section 2, the dichotomous choice (DC) form of CVM was used to take this approach. Bid amounts in this study varied between

NLG 1000[7] and 5000 per hectare. Average gross margins (excluding costs for contract work) for a cropping plan with cereals, potatoes and sugar beet range from NLG 3000 for sandy soils to NLG 5000 for the top clay soils (PAV, 2000). However, near the field edge, yields and associated gross margins are significantly lower than for the field centre (De Snoo, 1994).

Table 5.1 Field margin package offered in contingent valuation experiment

1.	Field margins of 3 m
2.	No chemical spraying and fertilising of margins between 1 January and 1 October. Incidental knapsack spraying to control problem weeds is allowed
3.	Margins have to be sown with a mixture of at least 3 different dicotyledons such as clovers, phacelia etc. (seed costs: approx. NLG 150/ hectare)
4.	The regulation is not valid for margins of whole fields that have already been set aside
5.	The margins do not count for the MacSharry set aside scheme.
6.	Sowing before 15 May and no tillage until 1 October
7.	A maximum of one cutting is allowed
8.	Minimum length of 500 meters
9.	Participation is voluntary and stopping is allowed after every year
10.	Variable premium amounts (NLG /ha)

Table 5.2 presents descriptive statistics of the data set as well as variable definitions. Variables INNOV, SOCIE, NVALUE and RISK reflecting farmers attitudes were measured on a 5 point Likert scale and converted to dummy variables (agree/disagree) because Likert scales are non-metric variables. Likert values 1 and 2 were converted to 0 (disagree) and Likert values 3, 4 and 5 were converted to 1 (agree). No multicollinearity was found among variables used. From the data it was concluded that farmers from the Province of Flevoland had the highest response rate. Furthermore there was a positive response bias towards larger farms and 'wildlife oriented' farms. 20 Percent of the respondents participated in a wildlife program for at least one year during the period 1997-1999, whereas in 1999 for the whole of the Netherlands 3,3 % of specialised crop farms and 5,9% of mixed crop farms employed wildlife activities (LEI/CBS, 2000).

[7] Netherlands Guilders: NLG 2.20371= EURO 1.

Table 5.2 Description and summary statistics of dependent and independent variables

Variable	Description	Mean	St. dev.	Min.	Max.
PART	Experience with wildlife oriented field margin and fallow land practices in past 3 years (1if yes, 0 if no)	0.20	0.40	0.00	1.00
CPART	Willingness to Accept offered field margin package (1 if yes, 0 if no)	0.52	0.50	0.00	1.00
INTPA	Percentage of field margins on the farm used for offered fictitious field margin package	24.94	34.91	0.00	100.00
AGE	Age of eldest farm manager	49.67	10.19	26.00	77.00
SUCC	Successor (1 if present or not yet known, 0 if no)	0.78	0.41	0.00	1.00
FTYP	Farm type (0 if crop, 1 if mixed crop)	0.31	0.46	0.00	1.00
OINC	Number of non-farm income sources	0.40	0.53	0.00	2.00
LABF	Labour force (FTE)	1.67	1.07	0.13	13.70
TOTHA	Farm size (Ha)	66.69	53.96	8.00	460.00
PMETI	Integrated Production method (1 if yes, 0 if no)	0.14	0.35	0.00	1.00
PMETO	Organic production method (1 if yes, 0 if no)	0.04	0.19	0.00	1.00
SHREN	Percentage of short term rented land (max 1 yr.)	6.46	13.01	0.00	69
CEREA	Percentage of cereals in crop rotation	25.72	17.75	0.00	100
PROVF	Province Flevoland (1 if yes, 0 if no)	0.49	0.50	0.00	1.00
PROVG	Province Groningen(1 if yes, 0 if no)	0.30	0.46	0.00	1.00
RESTR	Number of area specific restrictions applicable to the farm (e.g. drinking water area, Ecological Main Structure)	0.18	0.43	0.00	2.00
STYP	Soil type (0=sandy, 1=clay)	0.61	0.49	0.00	1.00
FISIZ	Average field size (Ha)	7.05	4.88	1.00	45.00
DITCH	Ditch length per ha (m)	95.52	60.08	0.00	595.24
WOOD R	Woodrows per ha (m)	11.68	35.72	0.00	357.14
YEARS	No. of years actual participation	0.46	0.98	0.00	3.00
FAM1	Familiarity with nature fallow regulations (1 if yes, 0 if no)	0.49	0.50	0.00	1.00
FAM2	Familiarity with field margin regulations (1 if yes, 0 if no)	0.66	0.48	0.00	1.00
INNOV	I like to try new ideas on my farm (0=disagree, 1=agree)	0.58	0.49	0.00	1.00
SOCIE	I want to know how society thinks about my farm (0=disagree, 1=agree)	0.40	0.49	0.00	1.00
NVALU	Cropping set aside land with a nature fallow mix is good for the image of agriculture (0=disagree, 1=agree)	0.60	0.49	0.00	1.00
RISK	Cropping field margins with a nature fallow mix will cause more weed problems on my farm(0=disagree, 1= agree)	0.76	0.43	0.00	1.00
BID	Bid amount in CV question (cents/m2)	30.40	13.73	10.00	50.00

5.3.3 Empirical model

Given the theoretical model and the hypotheses formulated, the empirical application focuses on both actual and contingent participation in wildlife programs. Data were obtained from different geographic regions. Soil type, crop rotation but also parcellation characteristics are different for these regions enabling a wide range of conditions to be studied.

Actual participation (PART) was considered a binary choice: farmers participating in at least one wildlife program in one of the last three years were regarded participants. The wildlife programs considered were, provincial field margin programs, the nation-wide program for fallow land, and programs linked to the Ecological Main Structure. Explanatory variables included farm and farmer characteristics and farmer attitudes:

$$PART_i^* = \beta_0 + \beta_1 AGE_i + \beta_2 SUCC_i + \beta_3 FTYP_i + \beta_4 OINC_i + \beta_5 LABF_i + \beta_6 TOTHA_i + \beta_7 PMETI_i + \beta_8 PMETO_i + \beta_9 SHREN_i + \beta_{10} CEREA_i + \beta_{11} PROVF_i + \beta_{12} PROVG_i + \beta_{13} RESTR_i + \beta_{14} STYP_i + \beta_{15} FISIZ_i + \beta_{16} DITCH_i + \beta_{17} WOODR_i + \beta_{18} YEARS_i + \beta_{19} FAM1_i + \beta_{20} FAM2_i + \beta_{21} INNOV_i + \beta_{22} SOCIE_i + \beta_{23} NVALU_i + \beta_{24} RISK_i + u_i$$

$$\text{With} \quad \begin{cases} PART_i = \quad 1 & \text{if } PART^*>0 \\ 0 \quad \text{otherwise} & \end{cases}$$

For variable definitions see Table 5.2.

Contingent participation (CPART) was considered a binary choice. Table 5.1 presents the proposed field margin program that was offered to farmers. The proposed program was derived from the existing scheme to encourage conservation practices on set aside land (MINLNV, 2000). This scheme was set up to encourage conservation practices on MacSharry set aside land. Farmers are compensated for seed costs associated with specific nature fallow mixtures. The program however is only available for full field application. Compared to existing field margin programs, usually predominantly consisting of unsprayed cereals, weed and disease risks of this specific program (see Table 5.1) are lower, increasing compatibility with ordinary farming practices. In addition to the variables of the actual participation regression, bid offer and actual participation (0/1) were used to explain contingent participation:

$$CPART_i^* = \beta_0 + \beta_1 AGE_i + \beta_2 SUCC_i + \beta_3 FTYP_i + \beta_4 OINC_i + \beta_5 LABF_i + \beta_6 TOTHA_i + \beta_7 PMETI_i + \beta_8 PMETO_i + \beta_9 SHREN_i + \beta_{10} CEREA_i + \beta_{11} PROVF_i + \beta_{12} PROVG_i + \beta_{13} RESTR_i + \beta_{14} STYP_i + \beta_{15} FISIZ_i + \beta_{16} DITCH_i + \beta_{17} WOODR_i + \beta_{18} YEARS_i + \beta_{19} FAM1_i + \beta_{20} FAM2_i + \beta_{21} INNOV_i + \beta_{22} SOCIE_i + \beta_{23} NVALU_i + \beta_{24} RISK_i + \beta_{25} PART_I + \beta_{26} BID_i + u_i$$

$$\text{With} \quad \begin{cases} CPART_i = 1 & \text{if } CPART^*>0 \\ 0 & \text{otherwise} \end{cases}$$

Intensity of participation (INTPA) was measured as the proportion of the total margin length that farmers intended to use for the proposed conservation practice (see Table 5.1). Since this explanatory variable has a censored distribution (values between 0 and 100) a tobit model was estimated with:

$$INTPA_i = \beta_0 + \beta_1 AGE_i + \beta_2 SUCC_i + \beta_3 FTYP_i + \beta_4 OINC_i + \beta_5 LABF_i + \beta_6 TOTHA_i + \beta_7 PMETI_i + \beta_8 PMETO_i + \beta_9 SHREN_i + \beta_{10} CEREA_i + \beta_{11} PROVF_i + \beta_{12} PROVG_i + \beta_{13} RESTR_i + \beta_{14} STYP_i + \beta_{15} FISIZ_i + \beta_{16} DITCH_i + \beta_{17} WOODR_i + \beta_{18} YEARS_i + \beta_{19} FAM1_i + \beta_{20} FAM2_i + \beta_{21} INNOV_i + \beta_{22} SOCIE_i + \beta_{23} NVALU_i + \beta_{24} RISK_i + \beta_{25} PART_i + \beta_{26} BID_i + u_i$$

for those participating and

$$INTPA_i = 0$$

for those not participating in the proposed field margin program.

5.4 Empirical results

5.4.1 Actual participation

The effects of the explanatory variables on the probability that a farmer participates in an existing wildlife management program are presented in Table 5.3. PMETO and RESTR are significant at the 1% and 5% level respectively, indicating that organic farmers as well as farmers that face area specific restrictions are more likely to participate. Furthermore PROVG was significant at the 5% level indicating that participation of farmers from the Province of Groningen was lower compared to the reference area (Province of Drenthe). FAM2 (Familiarity with field margin regulations) was nearly significant at the 5% level (P=0.07), with the magnitude of the coefficient indicating an effect on participation.

An often-used goodness of fit measure for probit models is the pseudo-R^2 introduced by McKelvey and Zavoina (1975). The value for the participation model was 0.46 with 84 % of the farmers classified correctly. These values are satisfactory.

Table 5.3 Probit estimates for parameters explaining actual participation in conservation programs

Variable	Coefficient	P-value
Constant	-1.4897*	0.018
AGE	-0.0028	0.117
SUCC	-0,3192	0.230
FTYP	0.1604	0.498
OINC	0.1149	0.578
LABF	0.0012	0.865
TOTHA	0.0035	0.104
PMETI	-0.0661	0.825
PMETO	1.7278**	0.003
SHREN	-0.0026	0.410
CEREA	-0.0004	0.387
PROVF	-0.1178	0.663
PROVG	-0.6040*	0.043
RESTR	0.4898*	0.029
STYP	-0.0073	0.717
FISIZ	-0.0014	0.143
DITCH	0.0014	0.490
WOODR	0.0039	0.326
FAM1	0.2341	0.333
FAM2	0.4936	0.071
INNOV	-0.1175	0.465
SOCIE	0.1660	0.454
NVALU	0.1190	0.459
RISK	-0.1584	0.511

*) Significant at the 5% level
**) Significant at the 1% level
***) Significant at the 1‰ level

5.4.2 Contingent participation

The effect of the explanatory variables on the probability that a farmer would accept the bid offered in the contingent valuation experiment is presented in Table 5.4. A highly significant effect ($P<0.001$) of bid offer (BID) on contingent participation was found. Fig. 5.1 presents the relationship between bid offer and percentage of acceptors of the offer. It is clear that from NLG 3000 per hectare onwards participation rates remain fairly constant at levels around 60%.

Factors FTYPE and OINC are significant at the 1% level indicating that specialised crop farmers and farmers with non-farm income sources are more willing to accept the offer. Also ditch length per ha had a significant affect on contingent participation ($P<0.05$). Contrary to actual participation, contingent participation was not affected by production method and province.

74% of the farmers was classified correctly by the model and McKelvey-Zavoina's R^2 was 0.55. This model has a better goodness of fit than the actual participation model, but the percentage of farmers classified correctly is lower.

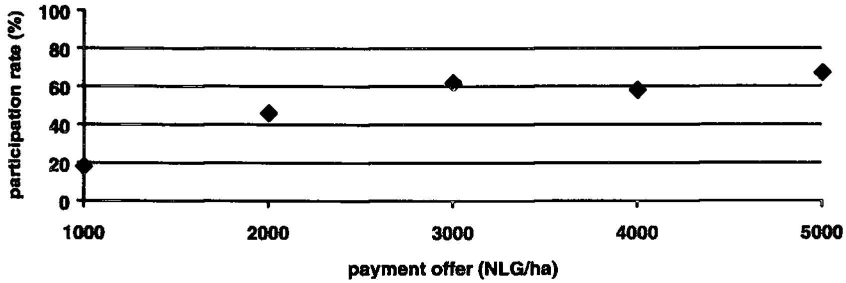

Fig. 5.1 Participation rate in proposed field margin program for different payment levels

Table 5.4 Probit estimates for parameters explaining contingent participation in proposed field margin program

Variable	Coefficient	P-value
Constant	-2.2655***	<0.001
PART	0.8595	0.228
AGE	-0.0007	0.700
SUCC	-0.1148	0.632
FTYP	-0.7521**	0.006
OINC	0.5607**	0.002
LABF	-0.1851	0.164
TOTHA	0.0044	0.071
PMETI	0.4410	0.107
PMETO	0.7903	0.155
SHREN	0.0029	0.304
CEREA	0.0001	0.888
PROVF	0.2292	0.397
PROVG	0.3955	0.270
RESTR	0.3751	0.129
STYP	0.5119	0.172
FISIZ	-0.0026	0.511
DITCH	0.0042*	0.025
WOODR	0.0010	0.744
YEARS	-0.3006	0.301
FAM1	-0.0211	0.923
FAM2	0.4134	0.065
INNOV	0.0011	0.666
SOCIE	0.0003	0.898
NVALU	-0.0001	0.819
RISK	0.0037	0.250
BID	0.0355***	<0.001

*) Significant at the 5% level
**) Significant at the 1% level
***) Significant at the 1‰ level

5.4.3 Intensity of participation

A tobit regression was carried out to determine the effect of the explanatory variables on the intensity of participation. Intensity was defined as the proportion of the total acreage of field margins on the farm that was offered for the new incentive program. The value of McKelvey-Zavoina's R^2 for this model was 0.32. Table 5.5 shows the effect of the explanatory variables on intensity of participation. All factors that were significantly affecting contingent participation also significantly affect intensity of participation. In addition, PMETI and RISK are significant at the 5% level, indicating that integrated farmers and farmers that have a lower perception of weed risks are willing to devote a higher proportion of their field margins to the proposed program.

Table 5.5 Tobit estimates for parameters explaining intensity of participation in proposed field margin program

Variable	Normalised Coefficient	P-value	Marginal effect decomposition	
			Intensity of participation	Participation
Constant	-128.9992***	<0.001		
PART	44.4881	0.243	9.624	14.901
AGE	-0.0046	0.972	-0.001	-0.001
SUCC	-18.9175	0.189	-2.521	-5.358
FTYP	-32.0288*	0.012	-4.752	-9.477
OINC	28.1477**	0.009	6.340	9.521
LABF	-4.1679	0.485	-0.657	-1.262
TOTHA	0.0753	0.548	0.015	0.024
PMETI	33.9998*	0.035	6.796	11.145
PMETO	28.0392	0.352	5.197	8.982
SHREN	0.1942	0.377	0.036	0.062
CEREA	0.0094	0.764	0.001	0.002
PROVF	27.7271	0.081	6.379	9.425
PROVG	0.3529	0.985	0.064	0.112
RESTR	24.7128	0.053	4.888	8.076
STYP	1.6590	0.917	0.307	0.531
FISIZ	-0.0001	0.999	-0.001	-0.001
DITCH	0.2574*	0.026	0.073	0.090
WOODR	-0.1874	0.257	-0.032	-0.058
YEARS	-10.5054	0.499	-1.731	-3.233
FAM1	9.4507	0.464	1.880	3.093
FAM2	1.9498	0.886	0.363	0.625
INNOV	-0.1428	0.835	-0.025	-0.045
SOCIE	0.1368	0.901	0.024	0.043
NVALU	0.0123	0.704	0.002	0.003
RISK	-21.0476*	0.038	-2.737	-5.899
BID	0.3476***	<0.001	0.077	0.117

*) Significant at the 5% level
**) Significant at the 1% level
***) Significant at the 1‰ level

Table 5.5 furthermore presents a decomposition of the marginal effects for all observations in the change in intensity for participants and the change in the probability of becoming a participant evaluated at mean x_i (see table 5.2). The estimates show that marginal changes in the explanatory variables increase the participation more than it does the intensity of participation. For RISK (risk perception of weeds) the relative change in the probability of participation is higher, in particular. In contrast, ditch length per ha (DITCH), regional constraints (PROVF) and non-farm income sources (OINC) show relatively large marginal effects for the intensity of participation. These results can be used to draw economic implications for improvement strategies for wildlife management on Dutch crop farms.

5.5 Discussion and conclusions

A survey of Dutch crop farmers was conducted to analyse actual participation in wildlife conservation programs and contingent participation in a proposed field margin program. Probit and tobit modelling were used to analyse the effect of farm characteristics and farmer attitudes on participation.

Participation in existing wildlife programs was highest for organic farmers, as well as for farmers that face area specific restrictions. Contingent participation was highest for specialised crop farmers and for integrated farmers. Furthermore it was concluded from the CVM-experiment that participation rates around 60% may be achieved with a bid offer above NLG 3000 per hectare for the proposed field margin program.

The expected positive influence of farm size and successor absence on participation in wildlife conservation programs was not confirmed by the survey and neither was familiarity with existing conservation programs. The hypothesised relation between societal commitment and innovativeness of the farmer on the one hand and willingness to participate in wildlife programs on the other hand was not observed for any of the models. Risk perception of wildlife measures did not significantly affect the decision to participate in neither the actual programs nor the proposed program. It did however significantly affect the intensity of participation in the proposed program. The hypothesis that the production environment would influence participation was confirmed by the survey. Area specific restrictions significantly and positively affected actual participation.

Overall the conclusions regarding the importance of the location of the farm (area specific institutional constraints) correspond very well with the findings of

Wilson (1997) for the ESA UK and of Kristensen *et al.* (2001) for landscape activities in Denmark. These studies found that the location factor (local socio-economic and biophysical environment) is a more important factor for understanding farmer involvement than a large range of farm and farmer characteristics.

This study observed an interesting difference in participation in existing wildlife programs and contingent participation, which might be due to scheme factors and farm and farmer factors. Scheme factors include the bid amount, and lower weed risks of the proposed field margin program as opposed to existing conservation programs. Whereas the existing programs attracted organic farmers, familiar with weed and disease risks, the proposed program predominantly attracted integrated farmers, stressing the importance of scheme factors. Farmer factors in the first place include the bias towards wildlife-orientation in the sample. Familiarity with existing programs on the other hand was not found to significantly influence actual or contingent participation. Furthermore actual participation did not significantly affect contingent participation (including intensity).

The study results regarding contingent participation suggest that participation rates in wildlife programs could be enhanced through conservation schemes that reduce the risks of weed and disease when compared to existing programs and that have adequate financial compensation features. Co-operation between policy makers and the farming community to discuss agronomically appropriate incentive schemes that are also adapted to the local circumstances therefore may well result in an increased participation in conservation schemes.

Finally it should be noticed that in spite of the high amounts of money that were offered for the proposed program in this study, 40 % of the farmers were still unwilling to participate. From the comments written by farmers on the returned survey forms it was clear that the perceived governmental interference with farming, was a major factor for not participating. Further research into the motives of these farmers is advised.

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CHAPTER 6

CO-ORDINATING ECONOMIC, BEHAVIOURAL AND SPATIAL ASPECTS OF WILDLIFE PRESERVATION IN AGRICULTURE[8]

Abstract

This chapter addresses the supply side of wildlife preservation and restoration in agriculture at the regional level. First, it is shown how network design modelling can be used for economic optimal spatial selection of unsprayed field margins creating a wildlife corridor in the landscape. Second, the compatibility of field margin management with farmers' perceptions is analysed by using the results of conjoint analysis in the spatial optimisation. The theoretical model is implemented by means of a GIS model and an empirical example for Dutch field crop farming is added to illustrate the approach.

6.1 Introduction

Increasingly, modern society values the environmental benefits that arise as joint outputs with primary land use, including semi-natural habitats and wildlife. In Western Europe, rapid changes in primary land use have jeopardised the supply of these benefits (Lowe and Whitby, 1997). The Common Agricultural Policy has been criticised for supporting these changes and recently European policy makers have begun to respond to such criticism. A growing EU commitment, underpinned by article 130R of the Maastricht Treaty, to integrating environmental considerations into agricultural policy, has strengthened the appearance of environmental cross-compliance (ECC) on the policy agenda (Spash and Falconer, 1997). Underlying ECC, is the principle of farmers providing protection and enhancement of the rural environment in return for support payments; see for example Russell and Fraser (1995). Regulations 1760/87 and 2078/92 mark the acceptance that supporting farmers to conserve wildlife and countryside might also help to curb over production. Supplementary to the traditional distinct geographical segregation of agricultural and wildlife functions, both functions should to a large extent

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[8] Based on: Wossink, A., J. van Wenum, C. Jurgens and G. de Snoo (1999) Co-ordinating economic, behavioural and spatial aspects of wildlife preservation in agriculture. *European Review of Agricultural Economics* 26(4) 443-460

blend within the rural environment. Besides, also in ecological circles attention is shifting towards the preservation of wildlife within the major forms of primary land use in addition to nature reserves and other protected areas (Edwards and Abivardi, 1998).

While it is easy to assert that conservation of wildlife is an objective of agricultural land use, it is less obvious how to achieve that objective in farming practice. Difficulties are manifold (see Lowe and Whitby, 1997; Önal, 1997; Slangen, 1997). There is particularly a need for (a) definition and measurement of wildlife in a tangible way that enables wildlife-enhancing qualities associated with different land use alternatives to be compared and gains and losses to be assessed, and (b) provision of information to farmers and policy makers that allows optimal choices to be made and effective policy incentives to be developed. In this context, 'optimal' means cost-efficient, so that those wildlife targets set by public demand are met at minimum cost. Identification of the best combination of land uses depends on the knowledge of production relationships between land use activities and wildlife values, and assessment of opportunity costs of foregone uses. In this task, agricultural economics has an important role to play.

Despite the importance of the issue and the wide policy interest, the list of quantitative studies on economic efficiency versus wildlife trade-offs is limited. Previous studies at the crop level have generally focused on the positive effects of refraining from pesticide use in northern European agriculture on the abundance of flora and fauna (see Boatman, 1994 and De Snoo, 1994, 1995). Economic studies at the farm level generally involve a comparison of specific land use regimes by analysis of accounting data and/or farm level modelling (*e.g.* Van Eck *et al.*, 1987). Previous studies at the regional level generally focus on site selection and identify the smallest number of reserve sites to realise targeted wildlife criteria (see for example Camm *et al.*, 1996; Pressey *et al.*, 1996). The economic aspects of site selection, however, are not considered in these studies.

The objective of this chapter is twofold. First, the economics of the joint spatial production of agricultural output and wildlife at the regional level are addressed. Second, it is shown how farmers' perceptions and preferences can be incorporated in the economic analysis of land use and wildlife.

Economic studies of the spatial aspects of agricultural production and the environment generally are on water quality of agricultural watersheds (*e.g.* Braden *et al.*, 1989, 1991; Moxey and White, 1994; Önal *et al.*, 1998). The spatial dimension, however, is also important in the case of the positive externalities of agricultural production. Economically, the 'where' question is of importance because of the advantages of selective control, *i.e.* protecting where it is most effective and least costly. Selective control requires identification of the most effective wildlife conservation methods and also where to apply these. Ecologically, the spatial distribution of species is important for their changes of propagation and dispersion. This chapter makes two specific contributions to the literature on spatial modelling. It is shown how network design modelling can be used to address the spatial aspect of wildlife conservation in agriculture and a geographical information system (GIS) is used in the empirical application. Despite their significant potential for environmental economic research, GIS techniques are still seldom used for this purpose in practice (Fletcher and Phipps, 1991; Moxey, 1996).

The second objective is to address farmers' perceptions and preferences. Close involvement of farmers with wildlife management requires the acceptability of conservation practices to farmers to be carefully considered. Interview techniques based on the theoretical insights of behavioural economics enable an ex ante assessment of the impact of farmers' (the consumers of agricultural technology) perceptions on their choices among management practices. The use of interviews is common in fields such as marketing research and analysis of consumer behaviour, but these techniques have not been widely reported in the agricultural economics literature (Adesina and Baidu-Forson, 1995; Wossink *et al.*, 1997).

The most important outcome of the approach presented in this chapter is a wildlife-cost frontier for an agricultural area. This frontier gives farmers and policy makers information for the design of an effective and acceptable policy, and information on the costs associated with different wildlife targets.

The chapter is organised as follows. Section 6.2 presents the general model. In section 6.3 the implementation of the theoretical model is considered. Section 6.4 provides an empirical example based on the situation of the Haarlemmermeer, an area near Amsterdam, the Netherlands. Section 6.5 discusses the results and provides conclusions.

6.2 General model

The theoretical model meets three criteria. It accounts for the effects of management restrictions on wildlife, it identifies the pattern of land use for optimal wildlife conservation at minimal cost, and it is specified at a highly disaggregated level to account for the heterogeneity of the natural environment.

Three categories of wildlife-orientated land use activities can be considered on fields in an agricultural region: (1) along the field (*i.e.* crop edges), (2) within the field, and (3) in between two crops in the rotation (fallow land, stubble field). In this study the focus is on the first category only.

The total pattern of field margins in an agricultural region can be considered as a set of edges linking points (nodes) where edges connect; see Hillier and Lieberman (1990: 336). More specifically, this can be denoted as an undirected graph $H = (T; A)$ where T is the set of nodes and A is the set of edges. Let edges be denoted as $(i,j) \in A$ with $i,j \in T$ *and* $i \neq j$. The sequence of nodes in (i,j) does not indicate a restricted direction within the edge because H is an undirected graph. Some edges may have a high value for wildlife conservation already (*e.g.* because of hedgerows), let these be identified by $(i,j) \in A^0 \subseteq A$.

Furthermore, let z_{ijm} denote wildlife output and let c_{ijm} denote the cost per unit of wildlife output for land use activity $m \in M$ on edge (i,j). The objective now is to maximise total wildlife output, N, at minimal regional cost of wildlife conservation, C, while meeting the restrictions that edges selected for conservation must be spatially connected in a wildlife corridor and that the edges $(i,j) \in A^0$ must be included. The edges $(i,j) \in A^0$ may include loops, so cycles must be allowed for in the analysis.

For the design of the optimal corridor, $x_{ijm} \in \{0,1\}$ and $y_i \in \{0,1\}$ are introduced. The variable $x_{ijm} = 1$ if edge (i,j) with land use activity m is included in the corridor and $x_{ijm} = 0$ otherwise. Similarly, y_i indicates whether or not node i is included in the corridor. A most demanding wildlife conservation policy will require the corridor to be complete. For the design of less demanding efficient policies the option of spatial gaps, G, in the corridor is introduced.

Finding the wildlife corridor leads to the mathematical formulation of a network problem. The optimal corridor will contain the edges $(i,j) \in A^0$ together with other edges $(i,j) \in A \setminus A^0$ that might, but need not, be chosen. That is, the optimisation is a modified

Steiner Tree Problem[9] (see Magnanti and Wong, 1984: 11; Engeval *et al.*, 1998) with the possibilities of cycles and gaps:

$$C^*(N): \quad Min \sum_{(i,j)\in A} \sum_{m\in M} c_{ijm} x_{ijm} \qquad [6.1a]$$

s.t

$$\sum_{i\in T} \sum_{m\in M} x_{itm} \geq y_t \ , \ t\in T \qquad [6.1b]$$

$$N \equiv \sum_{(i,j)\in A} \sum_{m\in M} z_{ijm} x_{ijm} \qquad [6.1c]$$

$$l_{ij} \geq G \ x_{ijm} \quad \forall \ (i,j)\in A \ , \ m\in M \qquad [6.1d]$$

$$\sum_{m\in M} x_{ijm} = 1 \quad \forall \ (i,j)\in A^0 \qquad [6.1e]$$

$$x_{ijm} \in \{0,1\} \quad \forall \ (i,j)\in A \ , \ m\in M \qquad [6.1f]$$

$$y_i \in \{0,1\} \quad \forall \ i\in T \qquad [6.1g]$$

$$m\in M \subseteq S \qquad [6.1h]$$

where

c_{ijm} = cost per unit wildlife output for land use activity $m\in M$ and edge $(i,j)\in A$;

x_{ijm} = binary variable that indicates whether edge $(i,j)\in A$ with land use activity $m\in M$ is included in the corridor;

y_i = binary variable that indicates whether node $i\in T$ is included in the corridor;

z_{ijm} = wildlife output on edge $(i,j)\in A$ with land use activity $m\in M$;

l_{ij} = length of edge $(i,j)\in A$;

N = spatially correlated wildlife for the total area;

G = maximum length of gaps allowed in the corridor;

[9] In the standard Steiner Tree Problem, the problem is to find a tree that spans a subset T^0 of the nodes T at minimal total edge cost.

A = set of edges;

A^0 = set of edges that has to be included in the optimal corridor;

T = set of all nodes;

M = set of wildlife oriented land use activities complying with farmers perceptions;

S = set of all wildlife oriented land use activities.

Equations [6.1b]-[6.1d] represent finding the wildlife corridor. Constraint [6.1b] ensures connectivity (see *e.g.* Engevall *et al.*, 1998: 13) whereas equation [6.1c] defines total spatially correlated wildlife, N. Solving the equation set will provide $C^*(N)$, that is the total cost C associated with wildlife preservation N. Systematically relaxing the requirement of completeness of the corridor by allowing gaps of increasing length will result in an efficient wildlife-cost frontier for the total region. The solution will also provide $F \in A \times M$, that is the set of edges $(i,j) \in A$ and practices $m \in M$ on these edges that satisfy the corridor requirement. By means of $C^*(N)$ and F, field specific proposals for a cost efficient increase of natural values can be given.

Selective control means that differences in c_{ijm} are taken into account. Ignoring these differences in corridor design leads to minimisation of the total number of edges included in the corridor or to minimisation of the total length of the corridor. Such procedures preclude identification of the most effective locations of wildlife conservation and would likely lead to an overestimation of the cost of conservation and to inappropriate policy proposal.

What is optimal for the entire region may not be optimal for individual farmers. Önal *et al.* (1998) point out that besides environmental and economic objectives also equity is important. When losses to farmers are not fully compensated it may be necessary to spread the impact of the policy option as fairly as possible among the farmers involved improving acceptability. It is assumed here that a compensation scheme is set up. Instead, another aspect of acceptability of wildlife conservation policy is addressed — namely by assessing the subset of wildlife conservation activities, $M \subseteq S$, acceptable to farmers and compatible with the prevailing farm organisation.

6.3 Implementation

To implement the theoretical model described in 6.2, it is necessary to address the full range of issues that arise in applied production economics, including the assessment of the set of wildlife conservation activities, the measurement of wildlife quality and the spatial aspect of wildlife conservation. The integration of these issues is now described.

6.3.1 Assessment of the activity set

Field margin management offers special opportunities to integrate economic, ecological and environmental aspects. In arable fields the largest number of plant species is found in the outer few meters of the crop. Crop edges are also more attractive for fauna than the field centre (De Snoo, 1995). At the same time, unsprayed field margins are of special importance for reducing pesticide concentration in surface water (De Snoo and Wegener Sleeswijk, 1993). In economic terms, crop edges are less valuable than the field interior. Management of the edges often requires additional effort, for instance in the case of wedge-shaped fields, and the yields from the edges are often lower. So, field margin management offers special opportunities for integrating economic, ecological and environmental aspects. In the Netherlands, many experiments on unsprayed field margins have been performed in which yield reductions and savings on pesticides and the occurrence of species are assessed (see *e.g.* De Snoo, 1994, 1995).

Unsprayed crop edges can be achieved in many different ways (such as unsprayed margins in regular crops, unsprayed cereal margins alongside non-cereal crops, grass strips alongside crops, fallow strips alongside crops). This total set, *S*, of potential ways of achieving unsprayed crop edges has to be reduced to a subset, *M*, considering the acceptability of the activities to farmers. Perception and conjoint analysis are suitable techniques for this assessment (Van der Meulen *et al.*, 1996). Both methods are based on survey data and provide insights into the subjective perceptions of farmers regarding unsprayed crop field margins to be gained (Churchill, 1991).

In a perception analysis, respondents are first asked to state their preference for a range of types of unsprayed crop edges. Next, they are asked to indicate their degree of agreement or disagreement with a large number of features of unsprayed crop edges. The individual scores of the features are measured on a five-point Likert scale (Churchill, 1991). These scores enable those features particularly relevant to the acceptability of unsprayed crop edges to be assessed. Next, levels are distinguished for each of these most relevant attributes and all possible combinations of the attribute levels are assessed (profiles). Finally, in the conjoint analysis respondents are asked to give a rank to these profiles. By using regression analysis to the standardised and normalised reversed rank data[10], the utility score for each attribute level is estimated[11]. Total utility is given by (Hair *et al.*, 1987: 609):

$$u_n = \sum_{k=1}^{K} u_{kn} w_{kn} \qquad [6.2a]$$

where:

u_n = overall utility for n-th respondent;

w_{kn} = the attribute level of the k-th attribute facing the n-th respondent.

By means of the outcomes of the ranking of the profiles the relative importance RI_{kn} of the attribute k to the individual respondent n can be assessed:

$$RI_{kn} = range_{kn} \Big/ \sum_{k=1}^{K} range_{kn} \qquad [6.2b]$$

where $range_{kn}$ equals the utility score for respondent n of the k-th attribute minus the utility score of the least preferred level of the k-th attribute (Hair *et al.*, 1987: 608).

The outcomes of the perception and the conjoint analysis are used in a region specific assessment of the set of land management practices M.

6.3.2 Measuring wildlife output

While agricultural output of a land management activity $m \in M$ is easy to quantify and measure in terms of the marketable yield, wildlife output is much more difficult to assess. The general understanding is that a direct measurement of wildlife is not feasible and that indicators are to be used instead.

[10] The estimates of utility within each attribute therefore have a mean of zero and differences between values are proportional to differences in desirability.

[11] The utility model is estimated for each individual respondent. This differs from contingent valuation and contingent ranking were a number of individuals is asked about their stated preferences for one set of alternatives, and a representative utility model is estimated for the relevant population.

Within the OECD work on agriculture and the environment a distinction is made in *p*ressure, *s*tate and *r*esponse (PSR) indicators (OECD, 1994). Recently, this PSR-framework has been applied to agriculture and wildlife. Pressure indicators are measurements of agricultural activities that cause changes in the conditions affecting wildlife, such as the use of pesticides and fertiliser. State indicators are direct measurements of the wildlife-enhancing conditions due to the pressures they place on species or biotopes. Finally, response indicators refer to reactions by farmers and governmental or societal organisations to changes in wildlife conditions, such as the use of financial incentives to enhance wildlife conservation in agriculture. The normative model outlined in section 6.2, requires the use of a state indicator measuring wildlife associated with agricultural land use. Several of these indicators have recently been developed both for diversity in terms of species and in terms of diversity of biotopes (see Van Wenum *et al.*, 1998, for an overview). For the purpose of this study, a species-based state indicator was considered most suitable because it departs from the observed presence and abundance of species and gives direct information on the relationship between land use and wildlife.

The most extensive operational species-based state indicator currently available for agriculture in the Netherlands is used here. This so-called yardstick for wildlife comprises a limited selection of species groups, which (Buys, 1995): (a) enables reliable and simple observation methods to be used (that can be undertaken by farmers), (b) provides information on the effect of farm management on wildlife, and (c) reflects wildlife on farms. Based on these three principles, vascular plants, mammals, birds, butterflies, amphibians and reptiles have been selected for the yardstick. Individual species are included from each of these groups using the following criteria: presence of the species in agricultural areas, likelihood of encountering the species, correlation between the species selected and farm management, indicative value of the species for the conditions of the biotope, and recognition of the species (Buys, 1995).

For each included species, the yardstick assigns an ecological value (0-100) that is based on rarity, population development and the importance of a species' presence in the Netherlands to the global survival of that species (see Van Wenum *et al.*, 1998, for a detailed description of the rating system). The ratings are used together with the observed presence of the species on the farm. Total scores are calculated separately for each species group, summed over the biotopes (such as farmland, grassland, ditches, and yard) on the farm. So, yardstick scores can not be aggregated over species groups, which implies that the design of the corridor and the assessment of the associated wildlife-cost frontier are specific to each species group.

Non-spraying of herbicides and insecticides along the edge of arable crops promotes the abundance of flowering plants and many insect groups that feed on flowering plants. The presence of insects in turn is a key factor for the survival of bird species. Given their crucial intermediate role in the food chain, insects were selected as indicative for total wildlife output. This was further reduced to butterflies since this is the only group of insects recognised in the wildlife yardstick used to measure wildlife output. Data on butterfly observations on unsprayed edges in various crops provided the information for the calculation of $z_{ijm} \quad \forall \ (i, j) \in A, \ \ m \in M$.

6.3.3 Cost of wildlife conservation

To assess the trade-offs between income and environmental pollution for land management activities, partial budgeting and programming techniques are the predominant normative methods (Roberts and Swinton, 1996). These approaches also can be used to analyse the financial implications of wildlife conservation at the farm level. Partial budgeting was used here to assess the costs c_{ijm} of each unsprayed edge $(i,j) \in A$ for the various crops/cropping practices $m \in M$:

$$c_{ijm} = [(1-b_m)p_m q_{ijm} + SAV_{ijm}]/z_{ijm} \qquad [6.3a]$$

where

$p_m =$ price output of management practice $m \in M$ (NLG/kg);

$q_{ijm} =$ yield in interior of field (kg ha^{-1}) for edge $(i,j) \in A$ with management practice $m \in M$;

$b_m =$ yield reduction management practice $m \in M$ without pesticide use compared with pesticide use;

$SAV_{ijm} =$ savings on pesticide cost (NLG/ha) for edge $(i,j) \in A$ with management activity $m \in M$.

In the Netherlands, data are available from experiments on unsprayed crop edges in which yield reductions and savings on pesticides and the occurrence of species are assessed (see De Snoo, 1994, 1995).

6.3.4 Integration and spatial aspects

To test the effects of unsprayed crop edges on biodiversity, it is necessary to develop an empirical analogue to the theoretical model presented in section 6.2 accounting for the insights of sections 6.3.1-6.3.3. The program ECONET4 was specially developed for this study and is based on the GIS package ECONET for optimisation of ecological corridor designation (Jurgens, 1992, 1993, 1994). ECONET4 addresses the modified Steiner Tree problem as presented in section 6.2 and simulates the costs, land management and spatial consequences. Given the position of existing unsprayed field margins, or other eco-objects, ECONET4 determines where additional ones have to be located to create the most efficient network. Traditionally GIS models use distances to select network links. One particular advantage of ECONET4 is the option of using edge weights on the interval [1,9], which enables the most cost-effective locations for the connecting elements to be identified.

The Steiner Tree problem is known to be NP-hard and therefore computationally elusive (any algorithm that computes an exact solution to a NP-hard problem requires an amount of computing time which increases at least exponentially with the size of the problem). ECONET4 searches for the optimal corridor by an enumeration procedure combining: (a) an optimal algorithm for the calculation of shortest or least cost links between nodes based on Dijkstra (1959), Floyd (1962) and Yen (1972, 1973), and (b) calculation of the minimum spanning tree from the shortest/least cost connections as found in (a). ECONET can only be used when digitised network data are provided.

6.4 Empirical example

For a first application of the approach the geographically simple research area of the Haarlemmermeer[12] was chosen. Three steps were taken for the application: (1) digitisation, *i.e.* co-ordinate information of the fields in the area was captured for GIS processing, (2) determination of the baseline situation regarding the distribution (frequency/spatial) of crops and field margin management, and (3) assessment of the ecological network.

The research area was made up of 36 farms each of 20 ha. The most common rotation on the farms is: winter wheat followed by potatoes, a second winter wheat crop,

[12] An area of reclaimed land, known as a polder.

and finally sugar beet (De Snoo, 1995: 143). All farms were assumed to have this cropping pattern of 50 percent wheat (WW), 25 percent potato (POT) and 25 percent sugar beet (SB). Each farm covers four adjacent fields of 5 ha with field size 500 x 100 meters. The parcel layout as presented in Fig. 6.1 captures the actual situation in the Haarlemmermeer. The spatial distribution of the crops over the fields of a farm, however, was assessed by random selection. Next, field margin management in the basic situation had to be assessed. Since there was no empirical information about this, each farm was attributed a type of field margin management out of four options (all margins sprayed, POT unsprayed, WW unsprayed, POT and WW unsprayed) also by means of random selection, see Fig. 6.2.

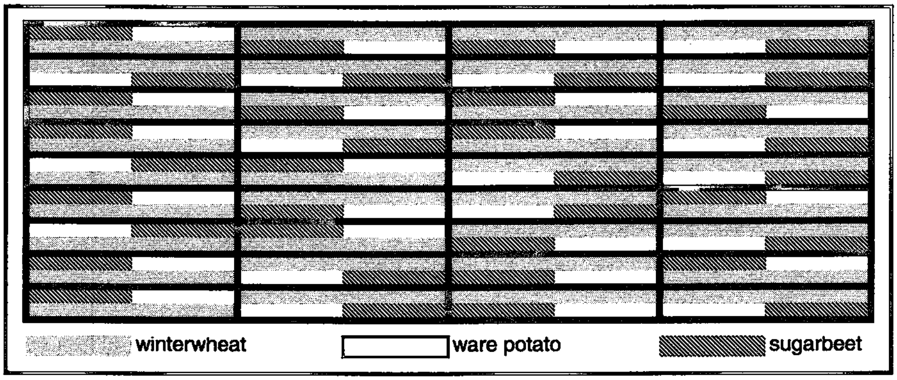

Fig. 6.1 Random cropping pattern in the research area (36 parcels of 20 ha)

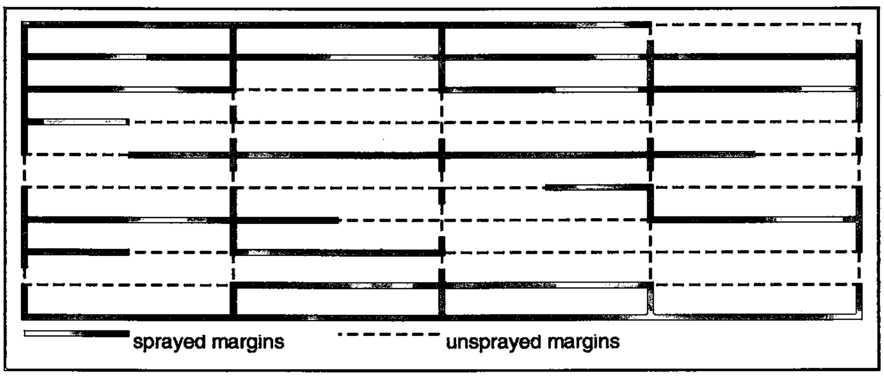

Fig. 6.2 Baseline situation (A)

To assess the region-specific set of acceptable types of unsprayed crop edges, results of earlier work were used in which farmers from four different regions in the Netherlands were interviewed (Van der Meulen *et al.*, 1996). The perception analysis in this study showed that farmers in the Haarlemmermeer region preferred margins in regular crops compared to grass strips or fallow strips. Grass strips are not compatible with the machinery available on these highly specialised crop farms and would require additional labour. Fallow strips are not acceptable because for these farmers it is very important to achieve a minimum yield from the unsprayed crop edge. The perception analysis also showed that (considering all respondents) four attributes were most relevant for the compatibility of unsprayed crop edges with the farm organisation: the width of the margin, the type of compensation payment scheme for implementing the unsprayed crop edges, guidance and whether the margin should be included in the rotation. In the conjoint analysis for each attribute two or three levels were distinguished (Table 6.1, rows marked with an asterisk).

Table 6.1 Results of the conjoint analysis

Attribute and Attribute levels	overall (n=31)		Gelderland (n=9)		Zeeland and Groningen (n=12)		Haarlemmermeer (n=10)	
	import. (in %)	utility	import. (in %)	Utility	Import. (in %)	utility	Import. (in %)	utility
Width:	46.15		45.90		40.00		66.20	
* 3 metres		-0.84		-1.70		-0.83		0.00
* 6 metres [a]		-0.21		0.89		0.11		-1.74
* defined by the farmer		1.05		0.81		0.72		1.74
Payment system:	23.48		20.66		35.36		4.23	
* conditional		0.48		0.58		0.69		0.11
* result		-0.48		-0.58		-0.69		-0.11
Guidance:	17.00		31.48		16.07		5.28	
* frequent		-0.35		-0.89		-0.31		0.14
* infrequent		0.35		0.89		0.31		-0.14
Location in the field:	13.36		1.97		8.57		24.30	
* fixed		-0.28		-0.06		-0.17		-0.64
* rotation		0.28		0.06		0.17		0.64

Legend) The three columns relate to three regions from which respondents were selected.

[a]) For the first two regions the utility scores for 6 meter edges is higher than for 3 meters edges, which might seem counter-intuitive. Crop farming in these regions is extensive and parcel layout is rather inefficient. Farmers indicated that they would locate unsprayed edges in field sections that were less productive (because of trees, hedgerows or inefficiency of machinery use).

Source: Van der Meulen *et al.* (1996).

Among groups of respondents significant differences were found in the preconditions, relating to differences in the intensity of farming and parcel structures. For the farmers in the Haarlemmermeer, both the width and the location of the crop edge are very important (see Table 6.1, last column under 'importance'). For these two features, the utility scores indicate that farmers would prefer an unsprayed margin of variable width but not wider than 3 metres and included in the rotation. Also from the bio-ecological point of view, 3 metres is considered an acceptable width for an unsprayed crop edge. Following the outcomes of the perception and conjoint analysis, 3-metre wide unsprayed edges were used in the crop rotation of wheat, potato and sugar beet in the first application for the Haarlemmermeer region.

Data to assess the costs and benefits of unsprayed[13] crop margins were available from an experiment in the area during 1990-1994 (De Snoo, 1995). All field margins are located alongside a ditch, so the non-sprayed margins in the crops on two adjacent fields were combined (see Table 6.2). For each of these edges, the wildlife output for butterflies, z_{ijm}, and cost per unit of wildlife output, c_{ijm}, were assessed, see equation *(3a)*. The latter had to be translated to integers in the [1, 9] interval for reasons to do with the GIS model used (Table 6.2).

Table 6.2 Costs and wildlife output of field margins combinations, Haarlemmermeer.

Field margin in	Costs (NLG/km) (1)	Wildlife score[a] (per km) (2)	Costs per unit wildlife score(1:2)	Weight[b]
WW/WW	112	1868	0.06	1
WW/POT	1	1652	0.00	1
WW/SB	565	1293	0.44	3
POT/POT	-110	1436	-0.08	1
POT/SB	455	1077	0.42	3
SB/SB	1019	718	1.42	9

[a]) Measured by means of the wildlife yardstick described in section 6.3.2 and in Buys (1995).
[b]) The ECONET model uses impedance values 1-9.
Source: Timmerman and Vijn (1996).

For the base line situation A, the total costs for the area of 720 ha sum to NLG[14] 606 being the total of the costs for the different types of field margins included (see Tables 6.3 and 6.4). Total wildlife output for the baseline situation was a yardstick score of 18017 (see section 6.3.2).

[13] No use of herbicides or insecticides. Fungicides were allowed.

[14] Netherlands Guilders: NLG 2.20371 = EURO 1.

Table 6.3 Costs and wildlife score for each of the strategies for the case study area

Strategy (gap width)	Additional Costs (NLG)	Additional wildlife points
A = Baseline	606	18017
BI	+781	+2780
BII (100)	+668	+2521
BII (200)	+328	+1415
CI	+440	+3283
CII (100)	+429	+3096
CII (200)	+327	+3024
CII (400)	+270	+2730
CII (600)	+112	+1868

Next, four strategies for network design were considered:

- (BI) The strict ecological point of view: non-spraying on the corridor calculated by ECONET4 using l_{ij} $\forall\ (i,j) \in A \backslash A^0$ for edge weights.
- (BII) The weak ecological point of view: allowing gaps (of 100, 200, 400 or 600 metres) in the corridor assuming that the species are able to bridge these gaps.
- (CI) The principle of selective control: non-spraying on the corridor calculated by ECONET4 using c_{ijm} $\forall\ (i,j) \in A \backslash A^0$ and $m \in M$ for edge weights.
- (CII) a modified version of CI in which gaps in the corridor are allowed.

Strategy BI implies the assessment of the corridor of shortest length connecting all the unsprayed margins present in the baseline situation. Fig. 6.3 shows the ECONET4 results for strategy BI. The double lines indicate where additional unsprayed field margins have to be established. Additionally 300 m of WW/WW-margin, 900 m of WW/SB, 400 m of WW/POT, 200 m SB/SB, 100 m SB/POT and 100 m POT/POT-margin are required. The additional costs sum to NLG 780 and the additional wildlife value is 2780 points (Table 6.3, strategy BI).

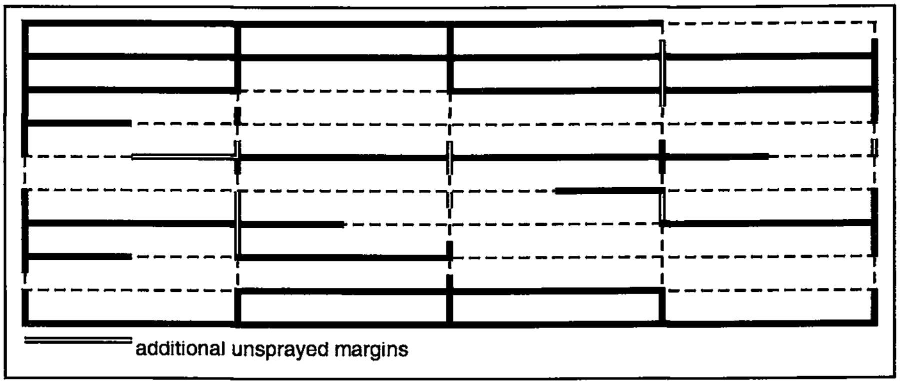

Fig. 6.3 Assessment according to the ecological point of view (BI)

In the same way the costs and wildlife values of the other strategies were calculated. Strategy BII (100), for instance, shows how many metres of extra margin is required when the species (butterflies in this case) can cross a 100 metres wide sprayed edge. Less effort is required to establish a network ensuring dispersion of butterflies but at the same time less extra wildlife value is added. When the species can travel 100 m, the unsprayed margins need not be connected in a closed network. Results for comparable calculations for a range of 200-600 m are given in Table 6.3. ECONET4 printouts for these strategies are available on request.

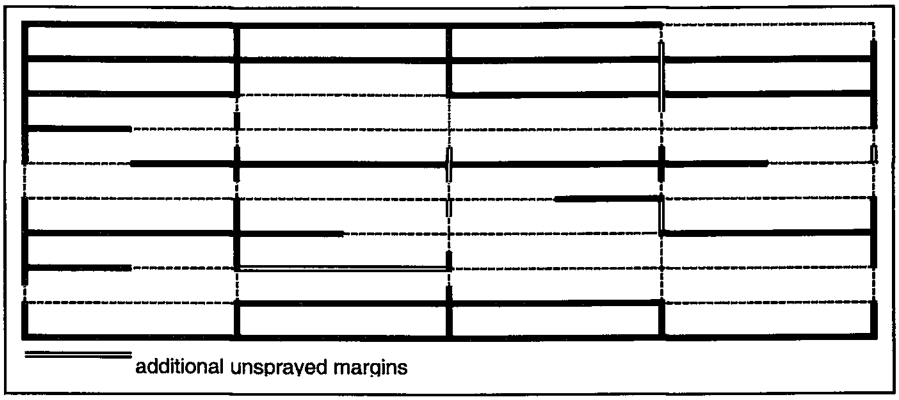

Fig. 6.4 Assessment according to the principle of selective control (CI)

Fig. 6.4 depicts the outcome for strategy CI. Not the minimum distance but the costs per unit of wildlife for each of the optional connecting edges are decisive in the assessment of the ecological network. The results of strategies CI and CII show the advantages of selective control. Strategy CI costs 44 percent less than BI, whereas its wildlife score is 18 percent higher. Strategy CII (100) costs 36 percent less than BII (100) and produces

23 percent more wildlife. In the determination of the spatial pattern, field margins with sugar beet (W/SB, POT/SB and SB/SB) are particularly avoided. This is in line with the high costs of unsprayed margins in sugar beet (Table 6.2).

The calculations in Table 6.3 are depicted by means of an efficiency frontier relating the costs and wildlife output levels resulting from the optimal solutions to each of the scenarios discussed (Fig. 6.5).

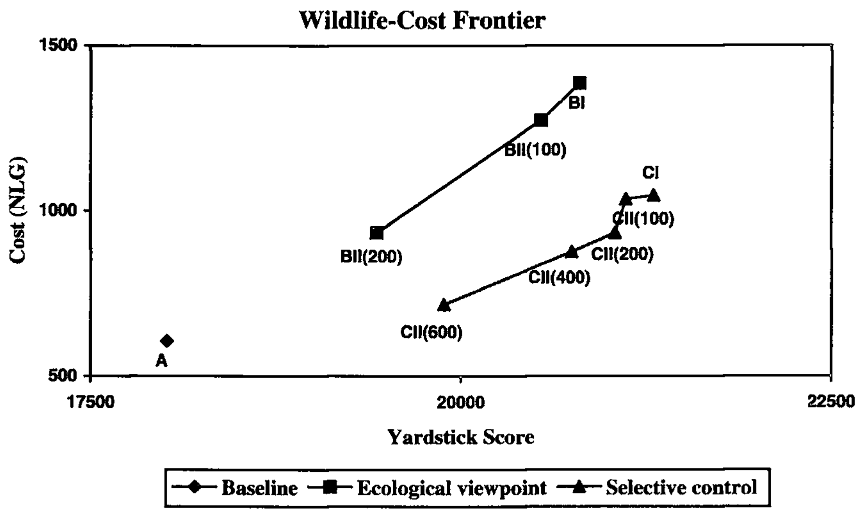

Fig. 6.5 Wildlife-cost frontier for the area (720 ha, random field margin management at the farm level) for two strategies (B, C) for establishing ecological networks

From the results on total costs and wildlife scores given in Table 6.3 and shown in Fig. 6.5, marginal costs were derived for strategies B and C. Table 6.4 reports the efficient levels of wildlife production for the respective strategies at selected marginal benefit levels. With selective control (strategy B) achieving the total potential increase in wildlife output would cost NLG 0.15 per unit of wildlife whereas with the ecological strategy this would be NLG 0.30 per unit.

Table 6.4 Summary marginal wildlife-cost frontiers

Strategy	Additional wildlife points	Percentage of potential wildlife output achieved for marginal costs (NLG) per unit of wildlife score of				
		0.10	0.15	0.20	0.25	0.30
Strategy BI	+2780	-	-	-	51	100
Strategy CI	+3283	57	100			

As the results suggest, the failure to account for selective control creates an upward bias in predictions of the cost of wildlife preservation. Moreover, the outcomes have implications for programme design. The optimal plans under strategy B and C involve different farmers; compare Fig. 6.3 and 6.4. Moreover, with strategy C the emphasis is mainly on unsprayed edges in wheat, whereas with strategy B all the three crops are involved.

The application to the Haarlemmermeer demonstrates the use of the approach of this study and its outcomes. The results reported here might be specific to the regions studied in this chapter. One may ask, for instance, whether and how the uniform field structure has affected the outcomes. Decisive for the results is not so much the uniformity of fields but their size and shape. A large number of small fields would enable shorter, cheaper ecological corridors to be constructed because the grid structure in ECONET4 would be much more detailed. How simulation of the crop distribution has affected the outcomes is an empirical question. If in reality more wheat is grown, the costs of wildlife preservation were over-estimated. If, on the other hand, in practice more sugar beet is grown, then the outcomes are under-estimated.

6.5 Concluding remarks

The wildlife cost frontier shows the advantages of selective control in wildlife conservation: more wildlife at lower costs. The approach further enables the spatial identification of ecologically desirable field margin management and which farmers should participate. The information on management options available, and the associated cost and ecological benefits can be very valuable for farmers and policy makers. Specifically, it would support the discussions regarding environmental cross-compliance, for example to impose a 'wildlife quality minimum' for agricultural areas in the Netherlands. It would also support private initiatives by groups of farmers, the 'wildlife co-operatives'.

The study shows that when research and extension programmes in wildlife preservation are being developed, it is important that an early attempt is made to obtain information on farmers' perceptions regarding land use activities that enhance wildlife, and on spatial constraints (parcel layout) affecting management options. Such an analysis should be conducted at the regional or sub-regional level to account for differences in farm situation and farmers' objectives (*see* Van Wenum and Wossink, 2001).

In this study only field margins were considered. To obtain a complete picture, two other options for wildlife conservation and restoration need to be included: management practices within the field, and fields left fallow in the rotation. Partial budgeting was used to capture the cost at the farm level. This technique has the advantage of simplicity but may leave out the influence of fixed, allocable inputs. The analysis can be performed using a more sophisticated method such as linear programming to obtain a more detailed assessment of the costs. Another issue for further research would be to consider the trade-off between the acceptability of the attribute levels as assessed in the conjoint analysis, and the operational costs at the regional level. It would be particularly interesting to consider different widths of the unsprayed crop edges.

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CHAPTER 7

GENERAL DISCUSSION

7.1 Introduction

The general objective of this thesis was to present an economic analysis of wildlife conservation in Dutch crop farming. This general objective was broken down into 5 specific research objectives around which the thesis was organised:

1. Selection and definition of appropriate indicators for wildlife in agriculture, specifically applicable at farm level (Chapter 2)
2. Definition of a wildlife production function (Chapter 3)
3. Definition of the optimal strategy for incorporating wildlife conservation measures on the farm from the economic viewpoint(Chapter 4)
4. Analysis of farmer participation in wildlife conservation programs and farmers' Willingness to Accept (Chapter 5)
5. Exploring the opportunities for a regional approach for wildlife conservation in agriculture (Chapter 6)

This chapter summarises and discusses the main outcomes of the previous chapters and will reflect on the research organisation and methods used. Implications for further research and for policy towards agricultural wildlife conservation are discussed.

7.2 Summary of main outcomes

The objectives formulated in the previous section required methods and information from different disciplines and sources. Ecological, agronomic and (socio)-economic knowledge was used in a coherent combination of methods derived from econometrics, operations research, behavioural economics and network analysis. Random effects modelling was used to estimate wildlife production functions and estimates from this procedure were used together with agronomic and economic information in farm optimisation modelling to normatively study decision making towards wildlife management at the farm level. Decision making was also studied in a positive way by analysing factors that determine farmers' participation in wildlife programs and willingness to accept. Finally a pilot study was done to explore the possibilities of a regional network approach incorporating both normative and positive elements.

In order to study decision making with regard to wildlife management in crop farming, indicators of wildlife play a crucial role. Assessing wildlife quality in a uniform and tangible way enables comparison of agricultural and wildlife activities, of farms and of regions. Furthermore indicators can be used for ecological-economic trade-off analyses at different scales and their potential is even greater when they can be used for incentive-based policies. In this thesis a new wildlife indicator was introduced that was specifically developed for use in farming and for incentive development. This so-called yardstick for biodiversity (or wildlife yardstick) was used throughout the thesis and compared to a conventional simple species richness measure.

A first application of the indicators was in modelling the relationship between wildlife output, management practices, regional conditions and non-observed farm specific factors. Both species richness and the wildlife yardstick were used in estimating wildlife production using random effects modelling. Only one species group was considered (vascular plants). When considering the selected management activities on their wildlife production, it was found that in terms of species richness, nature mix fallow was most beneficial to wildlife production when compared to other crops and fallow alternatives. When wildlife yardstick values were considered, unsprayed cereals were the most beneficial crop. Sugar beet and potatoes were found to be least beneficial, both in terms of species richness and yardstick value. Furthermore it was concluded that non-observed farm specific conditions are having a significant impact on wildlife production.

A next step in the research was to define the optimal strategy for incorporating wildlife activities in crop farming from the economic viewpoint. The estimated wildlife production function was used to generate wildlife coefficients of conventional cropping activities and alternative wildlife management activities. These activities and their wildlife coefficients (species richness and wildlife yardstick values) as well as their costs features served as input for a location-specific optimisation model at the farm level using the integer programming technique. Most important model outcome was a wildlife-cost frontier at the farm level. Model outcomes showed that rotating wildlife conservation practices across the farm is economically more attractive than fixed-location practices when uniform field conditions and constant wildlife production for multi-year wildlife practices are assumed. Furthermore model outcomes indicated that intensity of farming increases costs for enhancing wildlife levels.

The farm level analysis simply considers trade-offs between costs and wildlife output. Ecological aspects such as connectivity of wildlife activities across the farm were ignored. Furthermore no attention was paid to farmers' willingness to accept for the proposed wildlife activities. To accommodate the shortcomings of the farm level analysis two additional studies were carried out.

First, actual participation in existing wildlife conservation programs and contingent participation in a proposed field margin program were analysed. Probit and tobit modelling were used to analyse the effect of farm and farmer characteristics and farmer attitudes on participation. Actual participation was highest for organic farmers and farmers facing area specific restrictions. Contingent participation in a proposed field margin program was strongly affected by bid offer. Furthermore, specialisation, integrated farming, off farm income sources, risk perception and ditch length positively influenced contingent participation. The CVM-experiment suggested that up to 60 percent participation might be achieved with appropriate bid offers.

The importance of connectivity of wildlife activities and the creation of ecological networks is most pronounced at the regional level. The second additional study therefore departs from this level and addresses, in addition to costs, connectivity of measures and acceptability by farmers. It was shown how network design modelling can be used for economic optimal spatial selection of unsprayed field margins creating a wildlife corridor in the landscape. The approach showed the advantages of selective control in wildlife conservation: more wildlife at lower costs by enabling the spatial identification of ecologically desirable field margin management and which farmers should participate.

7.3 Discussion of data, methods and results

In order to conduct a comprehensive research with information and methods from many disciplines, simplifications and various assumptions are unavoidable.

The research presented used vascular plant indicators (species richness and wildlife yardstick) to estimate wildlife production in agriculture. Data availability and farm scale applicability were the predominant factors underlying this species group choice. The methodological principles of these indicators were analysed as well as its opportunities for use in farm management, research and policy towards nature conservation in agriculture. Research on the one hand and management and policy use on the other hand have different demands for simplicity of the yardstick, which may hamper an accurate scientific assessment of wildlife. On the other hand, data to make full use of the wildlife yardstick were lacking and therefore simplifications of the yardstick were welcomed.

Farm planning was used to find the optimal set of practices to reach a certain level of wildlife production based on a species richness indicator and on yardstick values for different practices. The set of wildlife practices considered was limited. Only set aside and field margin activities were considered that were easy to incorporate in crop farming enabling an immediate return to farming activities. Not included were quality improvement of existing landscape elements on the farm and more permanent types of wildlife activities such as forestation or wetland development. Apart from the distinction between various margins and field centre, all other farm conditions were considered uniform. Each field therefore was treated in the same way, whereas in reality large differences in for example soil conditions may exist, influencing the model outcomes. So, when agronomic and ecological features of fields or within fields differ significantly, different results may be obtained and fixed location practices on parts of the farm where less favourable conditions exist may be attractive. A linear relationship was assumed between indicator values and acreage: two hectares of a certain activity with a certain wildlife indicator score (either species richness based or yardstick based) had twice the score of one hectare of the same activity. Within farms this linear relationship may hold, however on the regional level the wildlife value of yet another hectare of the same activity may have a lower value to wildlife. Factors such as critical wildlife population sizes for survival, minimal habitat sizes needed for wildlife species and connectivity of ecological objects or nature reserves have to be taken into account when studying wildlife management at this scale.

Farm level optima were based on normative results ignoring farmer attitudes and preferences. Model outcomes therefore represent the trade-off between costs and wildlife only. Forecasting adoption of practices or participation in wildlife programs requires knowledge of preferences and attitudes of farmers. Risk attitude, government involvement and other factors may result into a discrepancy between model outcome and real life outcomes. Combining normative and positive research therefore puts the model outcomes in perspective. In this research for example 40% of farmers were unwilling to participate in an offered field margin program that was highly profitable form a normative economic viewpoint.

7.4 Future Research

This study focused on agricultural wildlife conservation in crop farming using vascular plant indicators as a proxy for total wildlife. Other species groups however may require different wildlife activities and therefore results may significantly differ. Research into this matter is advised. Contrary to crop farming, pastoral farming is of a more significant importance to wildlife conservation in the Netherlands being the sole biotope for meadow birds. Defining optimal strategies for conserving and enhancing meadow bird protection on different scales may help decision making in this field.

The regional pilot study offers a window of opportunities for future research: A further development of the explored network method, including analysis of farmers willingness to participate from surveys, site specific conditions for wildlife and agricultural production from GIS may aid decision making towards wildlife conservation at the regional level.

An interesting but difficult area of research is the integration of regional and farm scale research. Regional goals and farm level goals may differ and iterative procedures translating results from regional to farm level and vice versa may enhance decision making towards wildlife conservation. In fact this procedure more or less simulates the negotiation process between farmers on the one hand and regional policy makers on the other hand. Integration of positive and normative research in an iterative way, by incorporating attitudes and preferences into modelling studies and using modelling results in behavioural studies increases understanding of decision making in wildlife management by farmers and landowners. No examples of both types of integration (farm-region and model-behaviour) are yet available from literature.

7.5 Policy implications

Segregation or integration of agricultural and wildlife functions forms a major discussion point in policy making towards wildlife conservation. From an ecological viewpoint segregation is advised when certain wildlife functions cannot be fulfilled within agriculture or when they can be more efficiently fulfilled outside agriculture. Integration on the other hand is advised when certain wildlife functions can only or more easily be realised within agriculture, for example in the case of meadow birds in pastoral farming or for the development of arable flora in crop farming. However when only general abundant wildlife functions are realised within agricultural regions, as is the case in many programs, the question arises whether this justifies the amounts of public funds involved. Targeting of the money therefore, using specific programs for enhancing specific wildlife values is advised. Wildlife or nature however does not only have an intrinsic (existence) value and therefore conservation of species is not the only aim when considering agricultural wildlife conservation. In the anthropocentric view as expressed by Veeneklaas *et al.* (1994) nature also has a (user) value to humans for fulfilling different functions: production, carrier, information (including recreation) and regulation functions of nature are distinguished. In the environmental economics literature it is common to consider the total economic value of nature as the aggregate of all user and non-user motivated values. Although the user and existence point of view might be complementary, the associated values might be overlapping (Holstein, 1996).

Taking into account all the functions wildlife represents, for enhancing general types of wildlife an integral policy on environmental issues in agriculture is advised. An increased environmental quality, through for example reduced pesticide and nutrient use, may very well increase the conditions for the abundance of general types of wildlife. The EU income support system for cereal producers may be used for not only providing income but also for providing environmental and natural values at low costs. Flexibility of the current regulations on set aside, especially with regard to minimum acreage and width of margins offers major opportunities, for farmers throughout the European Union.

7.6 Conclusions

This thesis presented an economic analysis of wildlife conservation in Dutch crop farming. An enumeration of the most important conclusions of the research is listed below.

- Taking into account the requirements for the use of wildlife indicators in agriculture: farm-scale applicability, necessary assessments performable by farmers and possibilities for application in incentive schemes, species based indicators such as the yardstick for biodiversity are most appropriate.
- In terms of vascular plant species richness, nature mix fallow is most beneficial to wildlife production when compared to other crops and fallow alternatives. In terms of yardstick values of vascular plant species, unsprayed cereals are the most beneficial crops to wildlife production when compared to other crops and fallow alternatives.

- Non-observed farm specific conditions are having a significant impact on wildlife production.
- Farm model optimisation for crop farming indicates that with uniform production conditions across the farm, rotating of wildlife activities, mainly following the cereal crops is most attractive from the economic viewpoint: wildlife scores are thus obtained at lowest cost.
- Participation in existing wildlife programs for crop farming in the Netherlands was found to be highest for organic farmers, as well as for farmers that face area specific restrictions.
- Contingent participation in a proposed field margin program was found to be highest for specialised crop farmers and for integrated farmers.
- CVM suggests that farmer participation rates around 60% may be achieved with a bid offer above NLG 3000 per hectare for a proposed field margin program that reduces the risks of weed and disease when compared to existing programs.
- Selective control in wildlife conservation at the regional level gives more wildlife at lower costs by enabling the spatial identification of ecologically desirable field margin management and which farmers should participate.
- When research and extension programmes in wildlife conservation are being developed, it is important that an early attempt is made to obtain information on farmers' perceptions regarding land use activities that enhance wildlife, and on spatial constraints (parcel layout) affecting management options.
- Integration of positive and normative research in an iterative way, by incorporating attitudes and preferences into modelling studies and using modelling results in behavioural studies increases understanding of decision making in wildlife management by farmers and landowners.
- Taking into account all the functions wildlife represents, for enhancing general types of wildlife an integral policy on environmental issues in agriculture is advised. The EU income support system for cereal producers may be used for not only providing income but also for providing environmental and natural values at low costs.

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SUMMARY

Agriculture does not only produce food and fibre; it also helps shaping the rural environment. Increasingly, modern society values the environmental benefits which may arise as joint outputs with primary land use, including e.g. semi natural habitats and wildlife. In order to preserve and restore wildlife functions on agricultural land active wildlife management on farms is required. Identification of cost efficient wildlife policies depends on the relationships between current land use activities and wildlife values, and assessment of opportunity costs of foregone uses. A normative approach investigating the trade-off between wildlife and agricultural production and income is needed to this end and measurement and definition of wildlife values into appropriate indicators is essential. A normative approach serves well for strictly analysing trade-offs at the farm level. However, when forecasting or explaining farmer adoption of wildlife practices and participation in wildlife programs, behavioural aspects of farming need consideration as well.

The objective of the thesis was to make an economic analysis of wildlife conservation in crop farming in the Netherlands. Starting point from the thesis was the farm level as the interaction between wildlife, agricultural practices and income, as well as associated decision making is most pronounced at this level. The general objective was broken down into 5 specific research objectives around which the thesis was organised:

1. Selection and definition of appropriate indicators for wildlife in agriculture, specifically applicable at farm level
2. Definition of a wildlife production function
3. Definition of the optimal strategy for incorporating wildlife conservation measures on the farm from the economic viewpoint
4. Analysis of farmer participation in wildlife conservation programs and farmers' Willingness to Accept
5. Exploring the opportunities for a regional approach for wildlife conservation in agriculture

The above formulated research objectives required methods and information from different disciplines and sources. Ecological, agronomic and (socio)-economic knowledge was used in a coherent combination of methods derived from econometrics, operations research, behavioural economics and network analysis.

Chapter 2 focuses on wildlife quality indicators and the specific requirements when applied to agriculture. The value of nature and methodological principles of indicators are discussed and recently developed wildlife indicators are described.

Special attention is drawn to the so-called yardstick for biodiversity (wildlife yardstick) developed by the Centre for Agriculture and Environment. This yardstick is an instrument specifically to quantify and value wildlife on farms. It consists of a representative set of species for which a quantitative assessment is used together with an ecological species rating. These ratings are based on rarity, trend in population size and international importance of the species. Details of the yardstick are discussed and its opportunities for use in farm management, research and policy towards wildlife conservation in agriculture are highlighted.

Chapter 3 presents the functional form and estimation technique for a wildlife production function at the farm level. A random effects model is developed to capture the relationship between wildlife output, management practices, regional conditions and non-observed farm specific factors. The study uses species richness and the wildlife yardstick (both based on vascular plants) in estimating wildlife production functions. The model was implemented for panel data of Dutch field crop farmers. Results showed that in terms of species richness, nature mix fallow was most beneficial to wildlife production when compared to other crops and fallow alternatives. When yardstick values were considered, unsprayed cereals were the most beneficial crops. Furthermore it was concluded that non-observed farm specific conditions are having a significant impact on wildlife production.

Chapter 4 presents a location specific model for optimising wildlife management on crop farms using the integer programming technique. Available data and indicators of wildlife production are presented. Furthermore, time and location aspects of wildlife management are discussed. Most important model outcome is a wildlife-cost frontier at the farm level. Model outcomes show that rotating wildlife conservation practices across the farm is economically more attractive than fixed-location practices under the assumptions of uniform production conditions across the farm and constant wildlife production for multi-year wildlife practices.

Chapter 5 examines actual and contingent participation of Dutch crop farmers in wildlife conservation programs. Probit and tobit modelling were used to analyse the effect of farm and farmer characteristics and farmer attitudes on participation. The optimal bid offer was derived from a referendum CV survey for a proposed field margin program. Actual participation was highest for organic farmers and farmers facing area specific restrictions. Contingent participation was strongly affected by bid offer. Furthermore, specialisation, integrated farming, off farm income sources, risk perception and ditch length positively influenced contingent participation. The CVM-experiment suggested that up to 60 percent participation might be achieved with appropriate bid offers. Implications of the results for policy are discussed.

Chapter 6 addresses the supply side of wildlife conservation and restoration in agriculture at the regional level. First, it is shown how network design modelling can be used for economic optimal spatial selection of unsprayed field margins creating a wildlife corridor in the landscape. Second, the compatibility of field margin management with farmers' perceptions is analysed by using the results of conjoint analysis in the spatial optimisation. The theoretical model is implemented by means of a GIS model and an empirical example is added to illustrate the approach. The approach shows the advantages of selective control in wildlife conservation: more wildlife at lower costs by enabling the spatial identification of ecologically desirable field margin management and which farmers should participate

Chapter 7 summarises the main outcomes of the thesis and critically reviews the research organisation and methods used. Furthermore this chapter suggests opportunities for future research and discusses implications for policy. Further research includes the use of other species groups than vascular plants for indicating wildlife. These species groups may have different requirements for wildlife activities and therefore results may significantly differ. Furthermore, future research in the explored network modelling is advised, including analysis of farmers' willingness to participate from surveys and site specific conditions for wildlife and agricultural production from GIS. This may aid decision making towards wildlife conservation at the regional level. An interesting but difficult area of research is the integration of regional and farm scale research. Regional goals and farm level goals may differ and iterative procedures translating results from regional to farm level and vice versa may enhance decision making towards wildlife conservation. In fact this procedure more or less simulates the negotiation process between farmers on the one hand and regional policy makers on the other hand. Integration of positive and normative research in an iterative way, by incorporating attitudes and preferences into modelling studies and using modelling results in behavioural studies increases understanding of decision making in wildlife management by farmers and landowners.

For a further justification of enhancing general types of wildlife an integral policy on environmental issues in agriculture is advised. An increased environmental quality, through for example reduced pesticide and nutrient use, may very well increase the conditions for the abundance of general types of wildlife. The EU income support system for cereal producers may be used for not only providing income but also for providing environmental and natural values at low costs. Flexibility of the current regulations on set aside, especially with regard to minimum acreage and width of margins offers major opportunities, for farmers throughout the European Union.

SAMENVATTING

Maatschappelijk gezien krijgt de rol van de landbouw als producent van landschaps- en natuurwaarden steeds meer erkenning. Voor het herstel en behoud van natuurwaarden in het agrarisch gebied is actief natuurbeheer door boeren en landeigenaren gewenst. Voor het vaststellen van effectieve en kostenefficiënte vormen van natuurbeheer is inzicht nodig in de relatie tussen huidig grondgebruik, natuurwaarden, en het verlies aan agrarisch inkomen wanneer natuurbeheersmaatregelen worden doorgevoerd. Met een normatieve aanpak is het verband te bepalen tussen agrarische productie en inkomen enerzijds en natuurwaarden anderzijds. Het gebruik van indicatoren ter bepaling van natuurwaarden is hiervoor essentieel.

Een normatieve aanpak is beperkt tot de uitruil tussen inkomen en natuur. Voor het analyseren van de adoptie van natuurbeheersmaatregelen of het voorspellen van deelname in beheersprogramma's, moeten ook gedragsmatige aspecten worden meegenomen. Het doel van dit promotie-onderzoek was om een economische analyse te maken van agrarisch natuurbeheer in de Nederlandse akkerbouw. Het uitgangspunt van het onderzoek was het bedrijfsniveau, omdat de interacties tussen akkerbouw (management) activiteiten, inkomen en natuurwaarden en de daarmee samenhangende besluitvorming het meest duidelijk zijn op dit aggregatieniveau. Het onderzoek was opgesplitst in 5 specifieke doelen:

1. Het selecteren en vaststellen van indicatoren voor natuurwaarden in de landbouw die op bedrijfsniveau toepasbaar zijn.
2. Het vaststellen van een natuurproductiefunctie.
3. Het vaststellen van de economisch optimale strategie voor het inpassen van natuurbeheersmaatregelen op bedrijfsniveau.
4. Het analyseren van de deelname van akkerbouwers in natuurbeheersprogramma's en het onderzoeken van de acceptatie van natuurbeheersmaatregelen op het bedrijf.
5. Het verkennen van de mogelijkheden van een regionale aanpak voor agrarisch natuurbeheer met behulp van op bedrijfsniveau verkregen resultaten

Aan elk van deze doelen werd een hoofdstuk in het proefschrift gewijd. De aldus geformuleerde onderzoeksdoelen vereisten het gebruik van informatie en methodieken uit verschillende disciplines en bronnen. Ecologische, agronomische en (socio-) economische kennis werd gebruikt in een in dit proefschrift ontwikkeld

systeem van methodes uit de econometrie, operationele analyse, gedragseconomie en netwerk analyse.

In Hoofdstuk 2 werd de aandacht gericht op indicatoren voor natuurkwaliteit en de specifieke eisen die de landbouw en het bedrijfsniveau hieraan stellen. De waarde van natuur en de methodische principes van indicatoren werden besproken en recent ontwikkelde indicatoren werden beschreven. Er werd speciale aandacht gegeven aan de Natuurmeetlat, een indicator die ontwikkeld is door het Centrum voor Landbouw en Milieu. De natuurmeetlat is een instrument dat speciaal ontwikkeld is voor het kwantificeren en waarderen van natuur op landbouwbedrijven en bestaat uit een representatieve soortenset. Van elke soort wordt de hoeveelheid vastgesteld en deze wordt vermenigvuldigd met een soortspecifieke ecologische score. De scores zijn gebaseerd op zeldzaamheid, ontwikkeling van de populatiegrootte en internationale betekenis van de soort. Details van de meetlat werden besproken en speciale aandacht werd gegeven aan de mogelijkheden om de meetlat te gebruiken voor bedrijfseconomische doeleinden, en voor onderzoek en beleid met betrekking tot agrarisch natuurbeheer.

Hoofdstuk 3 presenteerde de specificatie en schattingsmethode voor een natuurproductiefunctie op bedrijfsniveau. Een random-effects model werd ontwikkeld om de relatie te schatten tussen natuurproductie, management activiteiten, regionale omstandigheden en niet waargenomen bedrijfsspecifieke factoren. De natuurproductie werd gemeten als de soortenrijkdom en als de score volgens de natuurmeetlat (beiden ingevuld voor vaatplanten). Het random-effects model werd toegepast voor paneldata van Nederlandse akkerbouwers. Uit de resultaten bleek dat wanneer natuurproductie in termen van soortenrijkdom werd uitgedrukt, natuurbraakmengsels de hoogste natuurproductie opleverden ten opzichte van andere gewassen en braakvarianten. Wanneer natuurmeetlatscores werden beschouwd scoorden onbespoten granen het hoogst. Een belangrijke conclusie van het onderzoek was dat niet waargenomen bedrijfsspecifieke omstandigheden een significant effect op de natuurproductie hadden.

Hoofdstuk 4 presenteerde een locatiespecifiek model voor het optimaliseren van natuurbeheer op akkerbouwbedrijven. Het model maakt gebruik van de integer programmeringstechniek. Beschikbaarheid van gegevens en indicatoren met betrekking tot natuurproductie werden besproken. Daarnaast werden tijd- en locatie-aspecten van agrarisch natuurbeheer geanalyseerd. De belangrijkste uitkomst van het model was een natuur-kostencurve op bedrijfsniveau. Modeluitkomsten gaven aan dat het roteren van natuurbeheersactiviteiten economisch aantrekkelijker is dan natuurproductie op vaste locaties. Hierbij werd overigens uitgegaan van uniforme productie omstandigheden op het bedrijf en een constante natuurproductie bij meerjarige beheersactiviteiten.

Hoofdstuk 5 analyseerde de deelname van akkerbouwers aan bestaande natuurbeheersprogramma's en de belangstelling voor deelname aan een nieuw fictief akkerrandenprogramma. Probit en tobit modellen werden gebruikt om het effect van ondernemers- en bedrijfskenmerken en van attitudes op deelname te onderzoeken. Een zogenaamd referendum CV (Contingent Valuation) onderzoek werd uitgevoerd om inzicht te krijgen in de benodigde vergoeding voor deelname aan het fictieve akkerrandenprogramma. Uit het onderzoek bleek dat met name biologische akkerbouwers en akkerbouwers die te maken hadden met regionale beperkingen (o.a. nabijheid ecologische hoofdstructuur, waterwingebied) deelnamen aan bestaande programma's. Deelname aan het nieuwe fictieve programma had de belangstelling van gespecialiseerde akkerbouwers, geïntegreerde bedrijven, van bedrijven met meerdere inkomensbronnen en bedrijven met relatief veel sloten. Naarmate de boeren het risico van ziekten, plagen en onkruiden hoger inschatten nam de belangstelling af. De hoogte van de vergoeding had een significante invloed op deelname aan het nieuwe fictieve programma. Uit het onderzoek werd geconcludeerd dat met een zorgvuldig gekozen akkerrandenpakket en een adequate vergoeding tot 60% van de akkerbouwers mogelijk zou willen deelnemen. Overleg tussen beleidsmakers en de agrarische sector ter bepaling van agronomisch verantwoorde en aan de locale situatie aangepaste beheerspakketten, zou daarom het aantal beherende boeren kunnen doen toenemen.

Hoofdstuk 6 behandelde de aanbodkant van agrarisch natuurbeheer op het regionale niveau. Ten eerste gaf het hoofdstuk weer hoe netwerkmodellen kunnen worden gebruikt voor de economisch optimale selectie van corridors van onbespoten akkerranden in een agrarisch landschap. Ten tweede werd in de ruimtelijke optimalisatie de perceptie van ondernemers ten aanzien van de inpasbaarheid van akkerrandenbeheer meegenomen door middel van de resultaten van eerder uitgevoerd conjoint onderzoek. Het theoretisch model werd toegepast door middel van een GIS model en een empirisch voorbeeld voor de Haarlemmermeer illustreerde de aanpak. De aanpak liet duidelijk de voordelen zien van een selectieve, op economische inzichten gebaseerde aanpak in agrarisch natuurbeheer: door een economisch optimale ruimtelijke planning van ecologisch gewenst akkerrandenbeheer, en het vaststellen van de participerende akkerbouwers opgenomen in het regionale plan werd meer natuur gerealiseerd tegen lagere kosten.

Hoofdstuk 7 gaf een samenvatting van de belangrijkste resultaten van dit proefschrift en gaf een kritische beschouwing ten aanzien van de organisatie van het onderzoek en de gebruikte methodes. Daarnaast gaf dit hoofdstuk suggesties voor verder onderzoek en werden beleidsimplicaties geanalyseerd. Het belang van verder onderzoek gebruikmakend van gegevens met betrekking tot andere soortgroepen dan vaatplanten werd onderkend. Andere eisen die deze soortgroepen stellen aan activiteiten op het akkerbouwbedrijf zouden tot andere conclusies kunnen leiden. Daarnaast wordt verder onderzoek geadviseerd naar de verkende netwerk modellering, rekening houdend met de bereidheid van boeren tot deelname aan natuurbeheersactiviteiten en het meenemen van locatiespecifieke omstandigheden voor zowel landbouw- als natuurproductie. Dit kan de besluitvorming ten aanzien van natuurbeheer op het regionale niveau ten goede komen.

Een interessant maar moeilijk onderzoeksgebied is de integratie van regio- en bedrijfsniveau. Regionale - en bedrijfsdoelen kunnen verschillen en iteratieve procedures die resultaten van regio- naar bedrijfsniveau vertalen en vice versa zouden de besluitvorming ten aanzien van agrarisch natuurbeheer kunnen verbeteren. In feite simuleert een dergelijke procedure min of meer het onderhandelingsproces tussen boeren aan de ene kant en regionale beleidsmakers aan de andere kant. Integratie van positief en normatief onderzoek op een iteratieve manier, door het opnemen van gedragsmatige aspecten in modelstudies en het gebruik van modeluitkomsten in gedragsstudies, verbetert het begrip van beslissingen ten aanzien van agrarisch natuurbeheer door boeren en landeigenaren.

Een verbeterde milieukwaliteit, door bijvoorbeeld een verminderd gebruik van gewasbeschermingsmiddelen en nutriënten, zal waarschijnlijk leiden tot verbeterde omstandigheden voor natuurproductie. Het EU inkomensondersteunende stelsel voor graanproducenten zou hiervoor gebruikt kunnen worden, door niet alleen te voorzien in inkomen maar ook door het creëren van natuur- en milieukwaliteit tegen lage kosten. Flexibiliteit van de huidige braakreguleringen, met name waar het gaat om eisen ten aanzien van minimumbreedte en -oppervlakte biedt veel mogelijkheden voor agrarische ondernemers in de hele Europese Unie.

RELATED PUBLICATIONS

Articles in refereed journals

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CURRICULUM VITAE

Jacob Hendrik van Wenum werd op 23 augustus 1970 geboren te Kootwijkerbroek. In 1988 behaalde hij zijn diploma Voorbereidend Wetenschappelijk Onderwijs (VWO) aan het Johannes Fontanus College te Barneveld. Hierna werd begonnen met de studierichting Landbouwplantenteelt aan de toenmalige Landbouwuniversiteit in Wageningen. In 1994 studeerde hij af met als hoofdvakken Graslandkunde en Agrarische Bedrijfseconomie. Zijn stage bracht hij door op het AgResearch Grasslands Institute in Palmerston North, Nieuw-Zeeland. Na de studie werkte hij kort als toegevoegd onderzoeker bij achtereenvolgens de leerstoelgroepen Agronomie en Agrarische Bedrijfseconomie van Wageningen Universiteit en als Agronomist bij een bedrijf in zaaizaden in het Verenigd Koninkrijk.

December 1995 volgde een nieuwe Wageningse aanstelling als Assistent in Opleiding bij de leerstoelgroep Agrarische Bedrijfseconomie op het onderzoeksproject: "Economische analyse van agrarische natuurproductie in een duurzame akkerbouw" waarvan dit proefschrift het resultaat is. Een gedeelte van het onderzoek werd uitgevoerd in de Verenigde Staten bij the Department of Agricultural and Resource Economics van North Carolina State University.

Sinds 1999 is Jaap van Wenum als Universitair Docent (UD) verbonden aan de leerstoelgroep Agrarische Bedrijfseconomie van Wageningen Universiteit. Als parttime akkerbouwer verliest hij daarbij de praktijk niet uit het oog.

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