CROSS-COMPLIANCE ASSESSMENT TOOL

Policy-oriented research:
Scientific support to policies SSP

Specific Targeted Research Project (STREP)

Deliverable(s): 4.3.1: Design and development of land use change, landscape and biodiversity impact generator as a component of the CCAT analytical tool, prototype 1

Due date of deliverable: [31-07-2007]

Actual submission date: [31-04-2008]

CROSS-COMPLIANCE ASSESSMENT TOOL
EC contract number 44423-CCAT
Project duration: January 2007-December 2009

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"This publication has been funded under the CCAT project, EU 6th Framework Programme, Priority 8.1 (European Commission, DG RTD, contract no. 44423-CCAT). Its content does not represent the official position of the European Commission and is entirely under the responsibility of the authors."

Project co-funded by the European Commission within the Sixth Framework Programme (2002-2006)		
Dissemination Level		
PU	Public	
PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the consortium (including the Commission Services)	X
CO	Confidential, only for members of the consortium (including the Commission Services)	

Contents

1 Introduction

1.1 Summary

In this report the details of the assessment of a selection of standards on land use, landscape and biodiversity for prototype 1 of the CCAT integrated assessment tool are described. The approach in general terms was already given in D 2.3, but this report underpins the approach with a literature review on the relationship between farming and landscape/biodiversity and related general principles of Cross Compliance, and it goes into more details as to the approach of the proposed assessments.

In accordance with D2.3 we are proposing to perform the following assessments in prototype 1:

- An expert qualitative estimate of the effectiveness of standards for biodiversity and landscape;
- Assessments of impacts induced by predicted land use changes as a consequence of Cross Compliance;
- Impact assessments on habitat quality derived from environmental indicators.

The expert qualitative estimate of the effectiveness of standards will concentrate on the selected SMRs and GAECs as specified in D2.3, Section 1.2.2 (Tabel 1.1), and will focus on those that target the preservation of landscapes and biodiversity (Birds and Habitats Directives, and GAECs targeted on habitat/landscape preservation, including e.g. measures against soil erosion).

For the other two impact assessment types only the effects of the standards addressed by the CAPRI/MITERRA prototype 1 models can be taken into account, since these form the input for these assessments. In relation to the SMRs this will only include the effects of the Nitrate Directive and for GAECs relevant standards will be included for as far as assessable with the CAPRI and MITERRA model in prototype 1. These GEAC standards are:

1) Soil erosion –
   a. minimum coverage
   b. minimum land management
   c. maintain terraces
2) Soil organic matter –
   a. standards for crop rotation
3) Minimum level of maintenance –
   a. minimum livestock stocking density and appropriate regimes
   b. protection of permanent grassland
   c. retention of landscape features
   d. avoiding encroachment of unwanted vegetation
   e. maintenance of olive groves

1.2 Outline of the report

In the next chapter first a literature review is given of the interrelationship between farming and landscape and biodiversity and how this relationship evolved in the last decades of the Common Agricultural Policy. At the end it will also discuss the general principles of Cross Compliance in relation to landscape and biodiversity. This chapter gives the state- of-play used for the further assessment approaches discussed in rest of report.

In Chapter 3 a detailed description is given of how the expert qualitative estimate will be performed of the effectiveness of standards under the Birds and Habitats Directives and all GEAC standards that specifically target the preservation and/or improvement of biodiversity and landscape quality. Chapter 4 gives a detailed explanation of the approach to assess impacts induced by predicted land use changes as a consequence of Cross Compliance. In Chapter 5 the approach is explained to assess impacts on habitat quality derived from environmental indicators that provide the pressures from which the direction of effects on landscape and biodiversity can be predicted.

2 State-of-play between farming, landscape and biodiversity

2.1 Introduction

In this Chapter a short literature based overview is given of the main principles of the relationship between farming, landscape and biodiversity and how this relationship evolved in the last decades under the Common Agricultural Policy (CAP). It also discusses further how this relationship has been addressed particularly by the standards included in the Cross Compliance policy.

2.2 Farming and biodiversity/landscape

Land use change is a fundamental form of global pressure affecting landscape and biodiversity (Sala et al., 2000, UNEP, 2002) and agriculture is one of its main drivers (Tilman et al., 2001). Agriculturally related land use changes include the conversion of complex natural ecosystems to simplified managed ones and the intensification of resource use, including application of more agrochemicals and a generally higher input and output (Matson et al., 1997). The expansion and intensification of agriculture is causing acknowledged impacts on biodiversity all around the world (Donald, 2004). In fact, farming is already the greatest extinction threat to birds (the best known taxon), and its adverse impacts look set to increase, especially in developing countries (Green et al., 2005).

In the European Union (EU), with agriculture standing as the most extended single land use type (163,7 million hectares of UAA, covering more than 40 % of the total area of EU-25; European Commission, 2007) highly diverse and characteristic agricultural landscapes have evolved under varied soil and climate conditions and farming traditions. Thus a substantial proportion of total biodiversity can be expected to be associated with farming, which highlights the considerable importance that the effects of agriculture might have on biodiversity (Tucker & Evans, 1997; OECD, 2001; EEA, 2005). This is also the main reason for introducing the concept of High Nature Value (HNV) farming and targeting these types of systems and associated areas in the most recent Rural Development program of the EU (e.g. Cooper at al., 2007; Beaufoy *et al.* 1994; Anger et al., 2002; Bignal and McCracken, 1996; de Miguel, 1999; Nagy, 2002; Andersen et al., 2003).

The interactions between agriculture and biodiversity are complex and diverse, and this complexity is reflected in the range of services that biodiversity provides to society, involving (OECD, 1996):

- Facilitation of the functioning of ecosystems, such as nutrient cycling, protection and enrichment of soils, pollination, regulation of temperature and local climates, and watershed filtration;

- Provision of source of most food and fibre products, including the basis for crop and livestock genetic resources, their improvement, and the development of new resources;
- Offer of a range of scientific, health/medicinal, cultural, aesthetic, recreational and other intangible (and non-monetary values) and services from biodiversity richness and abundance.

In this sense, the understanding of the combined ecological and social functions of agricultural biodiversity, the determination of its contribution to ecosystem goods and services and value for society at large, and the evaluation of options for the sustainable use and conservation of biodiversity across the agricultural landscape, have been pointed as the main challenging tasks to be solved (Jackson et al., 2007).

Drawing on the United Nations Convention on Biological Diversity[1] definition, biodiversity, as it relates to agriculture, can be considered in terms of three levels:

- Genetic diversity: the diversity of genes within domesticated plants and livestock species and wild relatives.
- Species diversity: the number and population of wild species (flora and fauna) affected by agriculture, including soil biota and the effects of non-native species on agriculture and biodiversity.
- Ecosystem diversity: the ecosystems formed by populations of species relevant to agriculture or species communities dependent on agricultural habitats.

Although the interdependencies between these three levels of diversity are obvious, due to practical reasons we will refer here mainly to the second and third levels, i.e. species diversity and ecosystems (landscape) diversity. Among the former, our review will be referred mainly to the wild flora and fauna species related to agricultural activities, covering a) wild species using agricultural land as habitat ranging from marginal use to complete dependence on agro-ecosystems, and b) wild species that use other habitats but are affected by farming activities. Only to some extent we will refer as well to the so called “life-support-system” (European Commission, 2001), particularly to pest controlling species and pollinators but not specifically to cryptobiota, including soil micro-organisms or earth worms.

As to the ecosystem level of agricultural biodiversity, it has been said to be manifest through (OECD, 2001):

- Changes in farming practices and systems.
- Changes in land use between agricultural and other land uses.
- The interaction between agriculture and adjacent ecosystems.

The landscape dimension implied in the functioning of these processes offers an overarching perspective to investigate and understand the dynamics and functions of species diversity *within* agricultural landscapes (Jackson et al., 2007). In this sense, it is widely recognized that agricultural lands can provide more suitable habitats for native wildlife and birds than do fragmented and extensively modified urban or suburban lands (e.g. Blann, 2006), often serving as a buffer between natural areas

[1] http://www.cbd.int/

and more highly altered landscapes, providing food, shelter, and habitat which allow movement and exchange of plant and animal populations. And further, agricultural landscapes that are composed of a mosaic of well-connected early and late successional habitats may also be more likely to harbour biota that contribute to regulating and supporting services for agriculture, compared to simple landscapes (Bengtsson et al., 2003; Swift et al., 2004; Tscharntke et al. 2005).Therefore, landscape emerges as a fundamental scale to analyze biodiversity in relation to agriculture (Hoffman, 2000).

However, apart from considering the agricultural landscape level in the sense of understanding the dynamics and functions of species biodiversity, we will also consider them in relation to amenity, aesthetic and cultural values of agricultural landscapes.

Along the 1990s, landscape has moved on to the international agenda as a focus for public policy. The inclusion of "cultural landscapes" within the scope of the World Heritage Convention (Rossler, 1995), the treatment of landscapes as a separate issue in the Dobris Assessment (Stanners & Bourdeau, 1995), the identification of the landscapes as an action theme in the Pan-European Biological and Landscape Diversity Strategy (Council of Europe, 1996), have all been significant developments in this direction. The adoption (2000) and put into force (2004) of the European Landscape Covention[2], represents the most recent reflection of this interest, putting the interaction between people and nature at the core of the idea of landscape: “an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors” (Council of Europe, 2000).

Seeing landscape as "nature plus people" has a particular resonance in agricultural landscapes, where human influences are so obvious (Meeus et al. 1990) and hence their significance in the attainment of sustainable development (Council of Europe, 2006). From here stems the interest in gaining knowledge about landscape structure, character and value (e.g. Sporron, 1995, Aalen, 1997, Wascher, 1997; Countryside Commission, 1998). Further, the recognition that landscape values are facing clear risks derived from agricultural changes has driven important efforts in developing indicators as a tool to track the current state and trends in agricultural landscapes (OECD, 2003), as a prerequisite for their protection, management and planning.

2.3 CAP, intensification and biodiversity and landscape

The relationships between farming and biodiversity have long been studied in Europe, both from a species perspective (see reviews in O'Connor & Shrubb, 1986, and Pain & Pienkowsky, 1997) and from the policy viewpoint (Baldock & Conder, 1985; Bernáldez, 1991; Valladares, 1993). Despite this attention and the increasing impact of environmental issues on the successive reforms of the European Common Agricultural Policy (e.g. Ritson & Harvey, 1997), the conservation status of most typical farmland birds in Europe continues to worsen (Donald et al., 2001, EEA,

[2] Council of Europe Treaty Series no. 176.
See http://www.coe.int/t/dg4/cultureheritage/Conventions/Landscape/default_en.asp

2005) despite the spatial importance of those agricultural systems usually characterized as extensive (EEA, 2004).

It is now commonplace to regard the Common Agricultural Policy (CAP) as the main instrument behind the dual processes of intensification and abandonment undergone by the majority of those European agricultural systems of high nature conservation value. Intensification of agriculture associated with the CAP has entailed profound changes in the functioning of European agroecosystems over the last 50 years. Traditionally, these were conformed by complex mosaics of extensive land uses, with remnants of natural habitats interspersed among crop fields and semi-natural grasslands (Potter 1997). But since its foundation, the policy designed to ensure food supply to the European nations after World War II not only has largely surpassed its original objective, but the intensification process triggered to reach it is causing a loss of biodiversity considered to be equivalent to that which might finally be caused by climatic change (Krebs et al. 1999).

The intensification of European agriculture has been linked to two main trends (Potter 1997): 1) Increase in the application of certain production factors per unit area to increase yields, like agro-chemical (fertilizing, herbicides and pesticides) and machinery (greater intensity and frequency of ploughing and tillage); 2) Specialization of land use and crops to maximize the economic return derived from application of production factors, both through a reduction in the number of cropped and husbandry varieties and an spatial differentiation between crops and grassland and increase in plot size. Consequently, yields have multiplied in the last decades, accompanied by a marked increase in farm and plot size, and, in parallel, by the redeployment and abandonment of farming in the less productive plots or areas.

The relationship between farming and biodiversity is not only a negative one, since also land abandonment might impact negatively on the associated habitats and species. That there is a positive relationship between extensive farming and biodiversity was shown by several studies showing biodiversity declines as a consequence of declines in extensive farming systems (e.g. Dunford and Feehan, 2001; Heath et al., 2000; Sirami et al., 2008; Peco et al., 2005, 2006; Bignal & McCracken, 1996 & 2000; MacDonald et al., 2000; Diemont, 1996; Schaminée and Meertens, 1992; Miles, 1981). Furthermore, recent research (e.g. Bunzel-Druke et al., 2002; Dirkx, 2002; Vera, 2000) shows that the native vegetation has been adapted to grazing over a very long period of time. Given the historic perspective that large herbivores are a natural component of the ecosystem and that most present day open habitats have been created and maintained by grazing, it seems logical that these habitats and all of their functional components must be perpetuated together with grazing animals. The importance of these grazed habitats is further underlined by the large number of species of different biota that rely on these (e.g. Anger *et al*., 2002; Bignal and McCracken, 1996; Miguel, 1999; Nagy, 2002).)

As a result, both intensification and abandonment processes have brought about a degradation of habitat quality and a decrease in the diversity and total biomass of the resources used by herbivores (except some pest species on crops) and predators and, ultimately, to rather similar rates of local species loss for very different taxonomic groups across European agricultural landscapes (Robinson & Sutherland 2002). Decline at the community level have affected species of plants, of insects (Wilson et al 1999), and better known, of birds. Nowadays agricultural habitats harbour the greatest proportion of species of birds with unfavourable conservation status in Europe (Tucker & Heath 1994; Donald et al. 2001).

Most of the available information on the relationship between agricultural intensification and biodiversity is about birds and vegetation. Especially the former, with its position at high levels of the trophic cascade and its varied food requirements, post it as a good general indicator of the health of the agro-ecosystem.

According to the review by Sanderson et al. (2005), there is a weight of convincing evidence that agricultural intensification is the direct cause for many farmland bird declines in Europe. The studies behind this evidence fall in three main types: a) correlative evidence linking changes in bird populations with temporal or spatial variations in agriculture; b) ecological evidence linking some element of agricultural change with bird ecology, distribution or density (habitat selection and habitat availability, food availability, etc.); c) evidence of long-term demographic changes attributable to agricultural change. Further, in a handful of cases, it has proved possible to establish causal links between the effects of agricultural change and the demography of declining species. This has been the case of declining Grey partridge populations in the UK, found to be causally linked with increased chick mortality resulting from reductions in their most important invertebrate prey caused by increased agrochemical use (Potts, 1997).

Also relationships between grazing and vegetation have been well documented. Grazing, as long as it is causing low to medium disturbance levels, determines the relative abundance of plant species in a habitat, thus influencing the competitive abilities of plant species relative to each other, preventing one species to become dominant over the rest. The range of species present and structures in the vegetation is therefore maintained at a higher level (see e.g. Palmer and Hester, 2000; Harris and Jones, 1998; Mitchell & Hartley, 2001; Alonso et al., 2001; Stevenson and Thompson, 1993; López-Mariño et al., 2004; Reiné et al., 2000).

It is clear that large knowledge gaps remain with respect to response of other taxonomic groups, such as invertebrates, earth biodiversity and mammals to changes in farming (Pain & Dixon 1997; Sutherland 2004). Nevertheless, negative relations with intensive farming have been demonstrated for invertebrates (e.g. Weibull et al. 2000; Östman et al. 2001 a; Sunderland & Samu, 2000), mammals (e.g. Harris & Woollard, 1990) and soil ecology (e.g. Kladivko, 2001).

Besides population density and species diversity within the different trophic levels, also the diversity of trophic levels within the agro-ecosystem has been affected by agricultural intensification (Benton et al. 2002). In particular, the loss of structural complexity of the agricultural landscape has been negatively related to the potential for biological control on pests in the own system, through a reduction in the diversity of natural enemies available to attack the pest species (Östman et al. 2001 a, b; Thies & Tscharntke 1999; Tscharntke et al. 2005). In this sense, intensification not only has affected biodiversity itself, but also to the ecosystem services provided by certain groups of that diversity (Tilman et al., 2002). However, also in this aspect doubts remain, essentially as to the type and extent of the changes experienced by trophic web structures as a result of intensification and the consequences that these changes might have on the pest's biological control (Schmidt et al. 2003)

The conclusions of the studies that analyze the effects of agricultural intensification/abandonment on biodiversity have been recently summarized in the process of habitat heterogeneity reduction at multiple spatial and temporal scales (Benton et al. 2003). This would be the universal consequence, from within individual fields to whole landscapes, of multivariate agricultural intensification. Therefore, it has been argued that future research should develop cross-cutting policy frameworks and management solutions that recreate that heterogeneity as the key to restoring and sustaining biodiversity in temperate agricultural systems.

As to the relationships between the intensification/abandonment process and the amenity, aesthetic and cultural values of the agricultural landscapes, available information is much less conclusive, although there is a general opinion that these farming trends have also deleterious effects on landscape values (e.g. Rossler, 1995; Potter, 1997; Vos & Meekes, 1999; Wascher, 1997, 2005; Meeus et al., 1990; Delbaere & Nieto Serradilla, 2004). In fact many countries have legislation explicitly recognising the importance of preserving the recreational, cultural, heritage, aesthetic and other amenity values embodied in agricultural landscapes (OECD, 2001).

A wide range of activities from the agricultural sector can affect landscapes, and because both appear to be intrinsically linked with each other it is not easy to identify which are actual pressures from agriculture in landscapes. The definition of what is actually acting as a pressure on today's landscapes has been said to be very much a discussion on threshold values and ecological carrying capacities on the one hand, and human perceptions and preferences on the other (Wascher, 2000).

As to the former point, it is generally agreed that there does seem to have been an overall trend towards increasing homogenisation of landscape structures in OECD countries over the past 50 years. Closely related to the structural changes of agricultural production (Parris, 2003), intensification, specialisation and concentration have been interdependent processes causing landscape simplification, including removal of landscape man-made elements (e.g. Adams et al., 1994; Agger & Brandt, 1988; Barr et al., 1993; Ihse, 1995). However, marginalization and abandonment of farms on the agricultural pockets and fringes that are unable to compete in an open market have also taken place (e.g. Baudry, 1991; Baldock et al., 1996), with variable although generally negative consequences for the diverse European agricultural landscapes (Potter, 1997).

As to the second point, main difficulties arise when assessing the scenic quality of landscapes and the recognition that landscape valuation is, to a greater or lesser degree, dependant up on subjective judgements and preferences. These issues have received attention since long, and various methodologies have been proposed to deal with both the way landscape is conceptualised and represented. This methodological diversity has been resumed in two main approaches (Lothian, 1999; Daniel, 2001): expert/object based approaches, where quality is an intrinsic physical attribute of landscape; and public perception based approaches, where landscape quality is "in the eye of the beholder".

Expert/object based approaches, mainly developed in the context of public land management practice, represent landscape in terms of the physical arrangements of features; that is, the structure and pattern of a land cover mosaic and its relationships with physical and biotic elements (i.e. terrain, geology, soils and vegetation), and cultural factors (i.e. people's use and management of the land). Landscapes are surveyed and classified and their quality evaluated based on assumptions which may or may not be made explicit. Landscape quality is taken as inherent to landscape properties. Examples of this approach are Linton, 1968; Iverson, 1975; Ramos et al., 1976; Countryside Commission, 1987; Crawford, 1994; Wascher, 2005.

Landscape indicators of quality proposed under the expert/object approach threatened by intensification/abandonment include statistical measures of landscape structure, such as patch and edge densities (e.g. European Commission, 1999); elaborated landscape attributes, such as diversity, coherence, and openness (Wascher, 2000); and cultural features on the agricultural landscape resulting from human activity, such as hedges, terraces or vernacular architecture (Meeus, 1995). Although generally understood to be positively related to landscape quality, their interpretation has proved a rather challenging objective. A first difficulty stems from the assumption it makes that quality is an inherent characteristic of the landscape, meaning that this is assessed using a subjective approach, lacking replicability (Lothian 1999). The credibility of the method relies on the reputed expertise of the individuals who carried it out, being it ideal that the team includes more members that only specialists, in order to ensure that the criteria used to measure landscape quality reflect community preferences. Further difficulties are related to the interpretation of changes in the indicators and the identification of 'thresholds for landscape quality', which are partly linked to subjective judgements that can considerably vary across regions and nations (Delbaere & Nieto Serradilla, 2004).

On its side, the perception based approaches, have been developed and used mostly in applied environmental perception and landscape assessment research, and represent landscape depending fundamentally upon understanding the perceptions of people; that is the biophysical features of the landscape are treated as stimuli that evoke in the observer aesthetically relevant physiological responses. Examples of this approach include Appleton, 1975; Kaplan, 1977; Zube 1973; Bernáldez & Parra, 1979; Herzog & Smith, 1988; Strumse, 1994.

However, the outcomes of the public perception based approaches to assess landscape quality are neither fully consistent. In general, public landscape preferences vary in terms of the balance between natural ("naturalness") and human influences ("managed" features) in the landscape, but not firm conclusions concerning the variables that underlie these differences have been attained (e.g. urbanity, familiarity, age, race, income, etc; Kaplan & Kaplan, 1989). Differences between expert judgments and preferences of the general public have been handled in extended research, concluding that important differences in opinion exist between these groups.

Research on public preferences as regard to agricultural landscapes in particular has shown differences depending on the uses that the different interviewed groups apply to the territory. For instance, in an study on the agricultural-livestock landscapes of the central Iberian Peninsula Gómez-Limón & de Lucio (1999) found different groups of users (livestock farmers, managers and recreationists) having different landscape preferences. The livestock farmers tend to prefer open landscapes, in comparison to the recreationists and managers who preferred landscapes with denser vegetation. Similarly, in a study on perception of rural landscapes in Flanders, Rogge et al. (2007) found striking differences between farmers, landscape experts and the general public. Farmers considered the openness and maintenance of the landscape to have an important influence on its overall attractiveness, while the green, enclosed landscapes were less preferred. For the experts, vegetation and openness influenced landscape in a positive way, while intensive agricultural crops had a significant negative influence. For the country-dwellers, appearance of vegetation, openness and maintenance of the landscape were decisive predictors for attractiveness.

Landscape preferences tend to converge when single groups of individuals are investigated. In a study in the Mid-West United States, Nassauer (1989) found several themes used by residents to value agricultural landscapes as attractive or unattractive. Among them figured *neatness*, which was associated with good farming, straight rows of crops, the absence of trees, the absence of weeds, and mown roadsides. Also reflecting on the farmer, figured *stewardship*, which was associated with strip-cropping, broad-base terraces, perennial covers and complex field or cropping patterns. Other studies in the USA (Nassauer & Westmacott, 1987) the Netherlands (Van den Berg et al., 1998) and Wales (Scott, 2002) have also shown that the maintenance and well managed settings of a landscape is also an important predictor for preference.

Investigating preferences for traditional and agrarian landscape scenes from Western Norway among students, Strumse (1994, 1996) found an almost unanimous consensus with respect to (a) the high preferences for traditional human-influenced settings and nature scenes including green grassy fields, and (b) the relative dislike for dominating human influence and many of the effects of modern farming practices. He concludes that support for the preservation of traditional agrarian landscapes could be expected as well as for the reintroduction of aesthetically valuable traditional landscape elements in modern agrarian landscapes.

Another predictor is the variety of the landscape. In a preference study in southern Spain with a varied group of respondents, Arriaza et al. (2004) found that the greater the homogeneity of the agricultural landscape, the lower the perceived visual beauty, due mainly to the lack of colour contrast. Further, these authors found higher preference for multi-crop land allocation and the use of green natural cover between olive trees.

There are also studies investigating preferences as to the resulting landscapes from management measures. For instance, in a study with a mixed group of respondents on impacts of buffer strips management on the visual acceptability of Finnish agricultural landscapes, Tahvanainen et al., (2002) showed that the interviewees preferred managed buffer strips to unmanaged strips, non-farmers preferring strips 5m wide over those 3m wide. Both the farmers and non-farmers, however, perceived the major impact of buffer strips as improving the quality of water courses. Further, investigating perception and aesthetic assessment by locals and tourists of spontaneous reafforestation in abandoned agricultural lands, Hunziker (1995) found that spontaneous reafforestation resulting in vast homogeneous forest patches reached lowest preference. Similar results were reached by Tahvanainen et al. (1996).

2.4 Cross Compliance standards and landscape and biodiversity

The previous sections have reviewed the main evidences and concerns about the impacts of intensification and/or abandonment of agriculture on biodiversity and landscape values associated to European farming systems. In the present section, the general principles of Cross Compliance in relation to landscape and biodiversity will be discussed in order to complete the state- of-play used for the further assessment approaches discussed in rest of report.

Outstanding, the main conclusion of our review is that the relationship between farming and associated biodiversity and landscapes is a complex one. The decisions taken by individual farmers, often driven by agricultural policies, can have both positive and negative impacts, since agriculture can both sustain and deteriorate the species, communities and landscapes which we now appreciate for their conservation value.

Baldock et al. (1989) in their elaboration and overview of this relationship identified the principle concerns regarding agriculture's impacts on the environment as follows:

- Water pollution including eutrophication from farm nutrients and wastes, pesticide contamination, and soil sediment plus groundwater salinisation in some regions.
- Unsustainable levels of water extraction for agriculture.
- Air pollution from ammonia which is an important source of acidification, with impacts on soils, forests, water, biodiversity and buildings.
- Agricultural emissions of greenhouse gases especially nitrous oxides and methane although emissions are projected to decline over the next twenty years.
- A number of concerns about soil quality, including reduced organic content and fertility, compaction, heavy metal and agrochemical contamination and acidification. Probably the most important and best documented impact is soil erosion, at unsustainable rates.
- Continuing declines in biodiversity. This is a very widespread phenomenon throughout the EU and its causes are not always fully understood. However, the loss or degradation of valuable semi-natural and fragile or particularly important ecosystems are still major causes for concern.
- Threats to high nature value farming systems and the difficulties of maintaining appropriate forms of agriculture in many marginal areas in the face of farm enlargement or intensification on the one hand, and decline and abandonment on the other.
- Increasing scale and homogeneity in landscapes is cited as a more general trend, as is a significant decline in labour input for undertaking sensitive land management.
- Concern about both the environmental and human health impacts of specific technologies, most notably with respect to pesticides and, more recently, GMOs.

It is has been noted that it is on mostly extensively managed, not intensively managed areas, where land abandonment and associated impacts more frequently occurs, while the contrary is the case for intensification related impacts (Moravec & Zemekis, 2007). Nevertheless, the fact that the real environmental impacts corresponding to these concerns are regionally variable, even inside any given region, has been shown by previous publications (e.g. Whitby, 1996; Buller at al., 2000) and project outputs (e.g. Schramek et al., 2006; Dimopoulus et al., 2007).

Therefore cross-compliance is likely to impact, to a greater or lesser degree, on most of the negative processes related to agriculture which are affecting biodiversity or landscape (Swales, 2007).

In the following, the different components of cross-compliance which are to be tackled in prototype 1 are reviewed in terms of the likely effects that their implementation might cause on biodiversity and landscape values.

Nitrate Directive: Nitrogen inputs.

High fertiliser inputs in agriculture and large concentrations of livestock rearing lead to leaching of nitrogen and cause eutrophication of surface water and soils affecting wildlife flora and fauna (e.g. shift in species). The consequences are however not always harmful. Depending on initial conditions and the degree of pollution, productivity may increase to the benefit of certain bird species (Newton, 1998). Evidence of negative effects on biodiversity comes for example from Van Wingerden et al. (1992), who found that grasshopper density and diversity decreased with increasing fertilisation levels. Another study by Siepel (1990) shows a shift from larger to smaller sized invertebrate species with increasing fertilisation levels, which may be a major cause of the decrease of insectivorous vertebrates in his highly fertilised samples. Nutrient inputs are obviously designed to favour crop growth and hence certain 'weed' species may be suppressed by dense crops. Similar effects may also occur due to vigorous growth of relatively few weed species which can exploit such conditions, leading to loss of plant species diversity which may in turn affect invertebrate abundance and diversity (Kleijn & van der Voort, 1997; Wilson & Tilman, 1993). Dense growth of crops can also impede access to the crop and ground by foraging birds and chick preventing them to get enough shelter against cold and wet weather (Shrubb & Lack, 1991). Increased fertilization has also been related to the loss of structural heterogeneity of crop sward (Benton et al., 2003). Nutrition of crops, normally in combination with plant protection, increases uniformity of establishment and subsequent growth, and reduces species and structural diversity of vegetation by killing and shading out of non-crop species in favour of dense, homogeneous crop swards. High nitrogen inputs may therefore also affect the landscape quality in terms of a reduced heterogeneity in species and structural diversity of vegetation.

GEACs for the prevention of soil erosion, soil organic matter and soil structure

A minimum soil cover and land management, including the retention of terraces, has proved to prevent soil erosion, particularly in areas already vulnerable (Lasanta et al., 2000; Boellstorff & Benito, 2005). It is clear that a loss of the topsoil as a result of erosion will inevitably go together with a loss of soil biodiversity (Kladivko, 2001), but also with the loss of vegetation and all other species connected to them in the higher trophic levels. In particular, appropriate management of arable stubble favoured vegetation cover and food abundance for the benefit of wintering granivorous birds (Moorcroft et al., 2002; Moreira et al., 2005; Whittingham et al., 2006).

Further, the maintenance of grassy covers, traditional man-made elements and a diversity of colours and textures in the landscape, have all been identified as components for the higher appreciation of agricultural landscapes (Arriaza et al., 2004), reflecting neatness and stewardship (Nassauer & Westmacott, 1987; Van den Berg et al., 1998; Nassauer, 1989; Scott, 2002) which are also important predictors for preference.

Minimum levels of maintenance.

Abandonment of agricultural land is an important cause for farmland biodiversity decline (EEA, 1999; Baldock, et al. 1996; Preiss et al., 1997; MacDonald et al., 2000; Suárez-Seoane et al., 2002; Sirami et al., 2008). Abandonment of farming would have many drastic effects. When the mechanisms that create farmland-habitats are lost, these habitats will also be lost. The rotational arable cropping and permanent grasslands will soon be overgrown with by fast growing weed-species, and later on by shrubs. Without the grazing pressure of domesticated ruminants on the rough grazing its vegetation will radically change. The loss of farmland habitats will inevitably lead to a loss of species depending on them and will certainly include many bird, butterfly and plant species of European nature conservation concern that depend upon (extensively) farmed land. Ostermann (1998) estimated that abandonment might put the preservation of 16 Annex I grassland habitats in danger. It is also estimated that approximately 16% of the habitats in Natura 2000 areas depend on a continuation of extensive farming[3].

In relation to landscape quality it is also clear that agricultural landscapes which are well maintained are generally appreciated more by the public (Nassauer, 1989; Nassauer & Westmacott, 1987; Van den Berg et al., 1998; Scott, 2002).

1) Minimum livestock stocking density and appropriate regimes

Appropriate grazing regimes on biodiversity are very beneficial to biodiversity as many studies have shown already. Stocking density is closely related to grazing pressure, which is an important controlling factor for the vegetation, and therefore also for the birds that use it as a habitat. Low stocking densities create a diverse habitat, with suitable ecological niches for many species. The range of species present and structures in the vegetation is therefore maintained at a higher level (see e.g. Palmer and Hester, 2000; Harris and Jones, 1998; Mitchell & Hartley, 2001; Alonso et al., 2001; Stevenson & Thompson, 1993; Peco et al., 2005; López-Mariño et al., 2000; Reiné et al., 2004). For farmland birds the diversity at the landscape level is very important too, and this is strongly influenced by the grassland management practices. Appropriate grassland management provides more open types of vegetation without letting these develop fully to their climax stage which results in suitable habitats for birds to winter and roost (Angelstamm, 1992; Söderström & Pärt, 2000). Another factor is that low stocking rates in the breeding season reduce the chance of

[3] Reporting of Member States in the framework of the Habitats Directive (92/42/EEC); status of July 2006.

egg- and chick trampling for ground breeding birds (Vickery et al., 1992). A low livestock stocking rate in winter leaves more food available for geese.

In terms of aesthetic attractiveness of the landscape it can generally be assumed that landscapes that are grazed and therefore have a wider diversity in openness and closedness, and structure and diversity of the vegetation, are more appreciated than monotonous woodland or arable landscapes (e.g. Hunziker, 1995; Tahvanainen et al., 1996; Rogge et al., 2007).

2) Protection of permanent pasture

Among the major land use types, permanent grassland is generally considered the most important from a landscape and nature conservation perspective (Ostermann, 1998; Bignal and McCracken, 2000; Beaufoy et al., 1994). Extensively managed permanent grassland provides habitats for many specialised plant and animal species (Díaz et al., 1997; Part & Soderstrom, 1999; Vickery, 2001; Brak et al, 2004; Beaufoy et al., 1994; Peco et al., 2006). For example, 92% of all target butterfly species in Europe depend on extensive grasslands.

As well it can generally be assumed that landscapes that are grazed and therefore have a wider diversity in openness and closedness are more appreciated than monotonous woodland or arable landscapes (e.g. Hunziker, 1995; Tahvanainen et al., 1996; Rogge et al., 2007).

3) Retention of landscape features

The presence of landscape features, such a hedges, un-sprayed field margins, tree-lines etc, is extremely important for biodiversity of different biota. Much research has already proven this and it is therefore clear that the retention of landscape features under the GEAC standards may potentially have positive effects on farmland biodiversity. Firstly, because it can be assumed that the larger the size the higher the nature value of that feature (De Blust and Hermy, 1997, p. 42-47). This is well illustrated by results of birds inventories in hedges and tree lines where numbers of different bird species increase with increasing length. Secondly, because the presence, density and size of species influences on many life functions of a species but overall it can be said that the more features there are the better the connectivity of a landscape for different species. Several studies (Chiverton & Sotherton, 1991; Sotherton *et al.*, 1985; Chiverton, 1994; Helenius, 1994) showed that unsprayed field margins supported higher densities of weeds and arthropods. Furthermore, partridge and pheasant chick survival was higher on farms where unsprayed margins were implemented (Sotherton *et al.*, 1985; Chiverton, 1994). Higher numbers of individuals and species of butterfly are found in unsprayed field margins as compared to sprayed ones (de Snoo, van der Poll & Bertels, 1998; Dover, 1997; Sotherton *et al.*, 1985) and foraging activity by butterflies is higher in unsprayed margins. These effects have been also proved to be valid on birds (de Snoo et al., 1994).

Landscapes with a high density of traditional man-made features also obtain a higher aesthetic appreciation (Strumse, 1994, 1996; Tahvanainen et al., 2002; Arriaza et al., 2004).

4) Diversity in land use

Farmland consisting of a mosaic of different arable, grass and semi-natural habitats generally provides a much larger biodiversity resource than large-scale monoculture habitats of intensive farming. A varied habitat-mosaic generally offers the greatest biodiversity benefit (Angelstamm, 1992). In the case of farmland birds because it offers a combination of breeding- foraging- and roosting-habitats, to which these species have slowly become adapted over time (Benton et al., 2003). Söderström and Pärt (2000) found that landscape composition could be an important factor if birds use different habitats for foraging and breeding. Bignal et al. (1988) came to the conclusion that EU Birds Directive Annex I species use different land-types in different times of the year. So whereas they may not be associated with agriculture in summer, they may be found on inbye fields in winter (e.g.: golden plover). It is therefore important, especially when studying birds, because of their high mobility and large range, to put the high nature value of a particular habitat in a landscape perspective.

Diversity of colours and textures in the landscape, favoured by a high diversity in land use, has been identified a component for the higher appreciation of agricultural landscapes (Nassauer, 1989; Arriaza et al., 2004).

5) Effects of increased irrigation

In dry and especially in arid regions the introduction of widespread irrigation has had a large effect on the populations of key species. This is especially true in the ‘pseudo steppes’ of Spain, Italy and France. Populations of steppe birds are decreasing rapidly, both in Europe as a whole and in Spain (see Goriup et al., 1991; Collar, 1996; Suarez et al., 1997). Using habitat suitability modelling, Brotons et al. (2004) estimated significant decreases in distribution after irrigation for seven of the nine species examined, i.e. the Little bustard, the Montagu’s harrier, the Roller and the Calandra lark, the species predicted to be more severely affected by predicted decreases in dry land area exceeding 50%. Investigating the case of the Little bustard in the Crau (Southern France), Wolf et al., (2001) found that both historical data and present habitat use suggest that little bustard population trends in the Crau are driven by the development of extensive agriculture, which may provide little bustards with resources unavailable or scarce in natural steppe. Moreover they found that irrigation schemes in these regions are currently reducing the availability of extensive agricultural habitats. The same negative effects of irrigation have been reported for many grassland species (Suárez, Naveso & De Juana, 1997; Lane, Alonso & Martín, 2001) including little bustards (De Juana & Martínez 1996).

3 Approach to estimate the effectiveness of standards for biodiversity/landscape quality

3.1 Introduction

The effectiveness of standards for biodiversity and landscape will be estimated through an analysis on the basis of expert knowledge.

As mentioned before, this expert qualitative estimate of the standards effectiveness will concentrate on the standards under the selected SMRs and GAECs as specified in D2.3, Section 1.2.2 (Tabel 1.1), and will focus on those that target the preservation of landscapes and biodiversity (Birds and Habitats Directives, and GAECs targeted on habitat/landscape preservation, including e.g. measures against soil erosion). For prototype 2 the inclusion of other Directives and GAECs, such as Groundwater, Nitrates and Plant Protection Products might be considered.

3.2 Standards to be included in the effectiveness assessment

The following standards (represented by their short names) will be taken into account in the first prototype:

Table 1: Standards taken into account in the effectiveness assessment (first prototype)

Measure	**Indication of the impact on biodiversity/landscape**
All standards under the **Wild Birds Directive**	o Birds, their eggs, nests and habitats will be preserved, specifically: (a) species in danger of extinction; (b) species vulnerable to specific changes in their habitat; (c) species considered rare because of small populations or restricted local distribution; (d) other species requiring particular attention for reasons of the specific nature of their habitat. o Landscapes with a rich nature (many of historic interest), that otherwise are likely to become extinct, will be preserved within the Special Protection Areas.
All standards under the **Habitats Directive**	o Natural habitats and of wild flora and fauna will be protected within Special Protection Areas (overlap with Birds Directive) and Special Areas of Conservation. o Landscapes with a rich nature (many of historic interest), that otherwise are likely to become extinct, will be preserved within the protected areas.
Relevant standards under the GAEC Soil erosion:	
Minimum coverage	o Abandonment of land will be avoided; especially the extensively managed vegetation cover are of ecological value and contribute to biodiversity. o The landscape will look well-maintained, which is appreciated by many people.
Minimum land management	o Land erosion will be prevented, facilitating the establishment of green covers which are of ecological value and contribute to biodiversity. o The landscape will look well-maintained, which is appreciated by many people.
Retain terraces	o Traditional terrace landscapes, many of historic and ecological interest, that otherwise are likely to become extinct, will be preserved.
Relevant standards under the GAEC Soil organic matter:	
Standards for crop rotation	o Crop diseases will be avoided, which might also be ecologically important
Appropriate stubble management	o Drastically burned landscapes (not appreciated by most people) and fire risk to

	nearby areas will be avoided.
Relevant standards under the GAEC Minimum level of maintenance:	
Minimum livestock stocking density and appropriate regimes	o Traditional grasslands will be preserved, which are of ecological value and contribute to biodiversity. o The landscape will look well-maintained, which is appreciated by many people.
Protection of permanent grassland	o Traditional landscapes with permanent pastures and rough grazing will be preserved; many of these are of ecological and historic interest. o Many of these are important habitats for meadow birds and raptor species. o These landscapes are also appreciated by many people for their beauty.
Retention of landscape features	o The character of traditionally enclosed landscapes will be preserved, many of which are important habitats for a large variety of animals. o Traditional landscape features that otherwise are likely to become extinct, will be preserved; many of these are of ecological, historic or archaeological interest.
Avoiding the encroachment of unwanted vegetation	o The character of traditionally open landscapes will be preserved, many of which are important habitats for meadow birds. o The landscape will look well-maintained, which is appreciated by many people.
Maintenance of olive groves	o Traditional olive groves (many of historic interest) will be preserved. o Traditional olive groves, by most people considered to be attractive will be preserved.

Since member states have a large freedom in choosing the way they implement their GEACs it is clear that this has lead to a large variation in the way they are implemented and which measures have been implemented (See also Jongeneel et al, 2007). The potential effects of GEACs on biodiversity and landscape quality are therefore expected to also very strongly between regions.

The assessment of the effectiveness of the standards mentioned above will be performed in the following steps:

1. Qualitative assessment of the potential effectiveness of standards.
2. Grouping of the standards according to their impact/effectiveness.
3. Computing the effectiveness and aggregate the results.

3.3 Qualitative assessment of the potential effectiveness

First the intrinsic potential effectiveness of standards will be specified in the following qualitative way:

+++: Standards targeting explicitly species or landscape features.
++: Standards targeting elements related to habitat/landscape quality
+: Targeting to preserve habitats/landscape in a general or subtle way
0: No expected impact (not a probable relationship).
-: Negative impact expected on biodiversity (not a probable relationship).
?: No clear relationship.

This qualitative analysis of the standards will be done based on the short names contained in the CIFAS database further elaborated and adapted in CCAT. These short names synthesise the detailed standards implemented in the surveyed

MS/Regions on the basis of their similarities (see CIFAS report Schramek et al., 2006 for further details). Included in the database there is information on the particular MS/Regions to which each short name is relevant, as well as information on the number of implemented standards in each particular MS/Region which are resumed by the short name.

The detail of the available information per standard will also determine whether each of them can be related to particular fields of biodiversity as its potential effectiveness, e.g. invertebrate species, bird species, mammal species or plant species. Since it is expected that a further refinement of the translation of the standards to the short names will still be in process until the end of 2008 it is not possible to do the effectiveness assessment at the level of biota. Instead for prototype 1, we will only assess the potential effects on "biodiversity/landscape" as a total. As for prototype 2 the feasibility of the disaggregated analysis to biota will be considered.

3.4 Grouping of standards

For an efficient assessment in the CATT tool we will group the standards according to the type of impact and potential effectiveness level. For the time being we assume that all standards we take into account will have a positive impact, so we will make a distinction between: +++, ++, +. When we apply these in a simple way to the relevant Directives and GAECs we might get the following groups:

For **the Wild Birds directive** (BD) and **Habitat directive** (HD):

- Groups BD3 and HD3: standards targeting explicitly the protection of bird, animal and plant species (e.g. prohibited practices-death, hunt, catch, possession), potential effectiveness level +++.
- Groups BD2 and HD2: standards targeting elements related to habitat quality for biodiversity (e.g. prohibited farming practices like the removal of hedges, potential effectiveness level ++.
- Group BD1 and HD1: Mostly indirectly affecting habitat quality for biodiversity (e.g. prohibited practices as applying chemically treaded seeds), potential effectiveness level +.

For the GAECS mentioned above (G) we will probably only distinguish 2 groups:

- Group G2: standards targeting protection of specified structural elements (e.g. retain terraces, protection of permanent grassland, retention of landscape features, maintenance of olive groves), potential effectiveness level ++.
- Group G1: standards mentioning management practices that help to preserve the landscape in a general or subtle way (e.g. practices like minimum coverage, avoiding encroachment of unwanted vegetation, minimum livestock stocking density, minimum land management, minimum level of maintenance), potential effectiveness level +.

In prototype 2 we might consider to distinguish more groups than these 5, if we consider this useful in relation to the availability of more specific data at a later stage.

3.5 Computing the effectiveness

We will use the **regional share of UAA** (or the share of a specific land use to which the standard is targeted, e.g. olive groves) to weight the potential effectiveness per NUTS2. The logic is that the greater the UAA, where the standards are to be implemented, the higher their potential effect on biodiversity will be. Both, share of UAA and absolute hectares of UAA at NUTS2 level will be used as weighting factors, reflecting respectively the magnitude and the extent of the potential effects. Since certain CC and SMRs standards are targeting particular land uses and crops, such as cereals, permanent grasslands, permanent crops or olive groves, the possibility to particularize in these cases the weighting exercise according to the regional share or hectares of these land uses will be investigated and possibly partly applied in prototype 1 and fully in prototype 2.

To come from a *potential* effectiveness to an estimate of the *expected* effectiveness the **level of compliance** is introduced in the analysis. This is done by using the data delivered by WP3 on the land use share per NUTS 2 region estimated to be compliant with different standards under 3 different scenarios of compliance. The scenarios considered will be the baseline situation in 2005, 75 % and 100 % compliance. With the CCAT integrated assessment tool, the end user will however obtain the possibility to change the compliance scenarios according to their own data and/or wishes for one or more regions (see also D2.5). If it turns out to be problematic to estimate shares of compliance for certain standards per land use category we will work with the regional average compliance levels for all land uses.

Since there are many different standards that in potential might have different levels of compliance in each NUTS2 region, it will be very cumbersome to assign in the CCAT tool to each individual standard (and each region and land use type) a different compliance level. Therefore, we propose to compute and assign an average level of compliance for each of the standards groups mentioned above to each NUTS2 region or Member State (or the whole EU).

For prototype 1 we propose to compute the effectiveness of the standards groups as follows:

Per standards group (BD3/2/1/, HD3/2/1, G2/1) per MS/NUTS2:

<u>Effectiveness = %UAA potentially affected x the average compliance level % of that standards group</u>

So the average compliance level per MS/NUTS2 per standards group must be computed first. If there is no sufficient detailed information to distinguish the level of compliance between standards groups and/or NUTS2 regions, then we will have to use more aggregated levels of compliance (e.g. for the Directives as a whole and per MS).

Aggregating the results

In the CCAT tool the computed effectiveness might be visualised (in tables, graphs or maps) for each standards group individually. This is only useful when enough detailed information is available on standards specific compliance levels, and/or on standards specific application in terms of share of UAA per NUTS2 region. Otherwise aggregated results will be more appropriate.

It’s now considered to aggregate the results per MS/NUTS2 as follows:

For the **Birds Directive**: effBD = (3 x effBD3 + 2 x effBD2 + effBD1)/6

For the **Habitat Directive**: effHD = (3 x effHD3 + 2 x effHD2 + effHD1)/6

And we could aggregate these two to make one result map/table for the Biodiversity Directives as follows: (effBD + effHD)/2

For the **GAECs**: effG = (2 x effG2 + effG1)/3.

For the interpretation of the aggregated results it is clear that the higher the score of the aggregated results, the more effective the CC standards targeting biodiversity and landscape quality are assumed to be.

4 Approach to the land use based assessments

4.1 Introduction

In prototype 1 we will perform the following assessments of impacts induced by predicted land use changes as a consequence of Cross Compliance:

- change in share of intensive/extensive land use
- change in density and share of intensive/extensive livestock
- change in land use diversity (evenness)

These impacts of compliance level scenarios can be positive but also negative because land use changes can also be triggered by financial aspects which may induce land use changes with an adverse effect on biodiversity and landscape as discussed in Chapter 2.

The three mentioned indicators will be calculated at NUTS 2 level using the output of CAPRI/MITERRA models. They will be used as pressure indicators on biodiversity and landscape. From chapter 2 it became clear that these changes in pressures can be interpreted as positive for both biodiversity and landscape values is there is a shift to more extensive land uses, more extensive livestock and towards a higher land use diversity.

Contrary to the case of the potential effectiveness approach, compliance levels are already internalised in the calculation of the land change by the CAPRI/MITERRA models.

4.2 Change in share of intensive/extensive land uses

To assess the impacts the present share of intensive/extensive land uses (the situation for the baseline year (2005) needs to be compared with (modelled) future patterns after e.g. an expected level of increase in Cross Compliance implementation. Then it can also be derived where and in what degree an increase in extensive land use can be expected, which is likely to lead to an improvement in biodiversity and landscape quality.

The input data for this assessment comes from the CAPRI database, which specifies 35 different land use categories (for the base line situation which is 2005) and the CAPRI model output for the future situations. CAPRI assesses the response of farmers in relation to CC standards and translates this response in a change in land use per NUTS 2. CAPRI works with the same land use classes as in FSS which includes 34 different crops and permanent grassland.

These land use classes will be classified in intensive and extensive categories, taking ecological principles into account. Certain crops are intensive per definition, in the sense that they are always managed under high levels of chemical inputs, including fertilisers, herbicides, plant protection products and irrigation (see the Table 4.1 below).

Table 4.1 EUROSTAT definitions of land uses and grouping into intensive and extensive crops

Crops	**Intensive**	**To be assessed**
Soft Wheat		X
Durum Wheat		X
Barley		X
Rye		X
Oats		X
Maize	X	
Other Cereals		X
Fallow Land		X
Rice		X
Sunflower		X
Soya	X	
Texture Crops	X	
Pulses		X
Other Crops		X
Potatoes	X	
Sugar Beet	X	
Root Crops	X	
Rape		X

Tobacco	X	
Other Industrial	X	
Tomatoes	X	
Other Vegetable	X	
Flowers	X	
Other Fodder		X
Permanent grassland		X
Nursery	X	
Fruits		X
Citrus	X	
Olive		X
Vine		X

Some others nevertheless can be managed under intensive or extensive practices (e.g. wheat, rye, rice, etc.). Therefore, a preliminary assessment needs to be made to determine whether a particular crop in a region belongs to the intensive or the extensive modality. Information on other input levels can be derived from the pre-model CAPRI input data. These include estimates on input levels for different crops both in terms of artificial fertilisers, agro-chemicals and irrigation since such information is needed to make realistic estimates of the production costs per crop and per region (NUTS 2). The information on fertiliser input levels per crop type per region are delivered by MITERRA to CAPRI (as is explained in D2.3 Chapter 2 and 3).

For prototype 1 the estimation of the share of intensive and extensive crops, before implementation of CC (before 2005) and at different compliance levels after implementation, can only be done at the whole Nuts 2 level. This is because CAPRI also delivers the assessments of the changes in land use at the level of the whole region. In prototype 2 a more sophisticated approach to estimating intensive and extensive land uses in which estimates will be made of the share of separate crops and land uses by involving different types of farming systems in the estimation characterised by intensity within the different Nuts 2 regions.

In order to come to a division of intensive and extensive crops, for prototype 1 first and EU wide assessment is made, based on the pre-model CAPRI input data and the MITERRA-CAPRI estimates on input levels in 2005 (baseline-situation)specified for:

1. Artificial fertilisers and total N-gift per hectare per crop (CAPRI-pre-model and MITERRA)
2. Agro-chemicals
3. Irrigation

With these data an average EU-wide input level for the 3 categories of input per crop is calculated. Crops per NUTS 2 with an average region-wide input level (in the three input categories together) below this EU average are classified in the extensive category. If the average region-wide input level is above the EU average they are classified as intensive crops. The exact break even points between intensive and extensive will of course be established with care. The analysis will show how the eventual EU-average should be calculated and what the distance to this average should be to become classified in the intensive or extensive class.

Equally, the division in intensive and extensive land uses will be made for the predicted CAPRI-MITERRA land use change and input level results (input levels might also change with CC implementation) under compliance scenarios (75 % and 100 % compliance. In this way, the changes in intensive/extensive land use shares can be calculated.

The impacts of land use change will be derived from the % UAA of intensive and extensive land uses:

- If in a MS/NUTS2 region the share of extensive land use will increase (and intensive land use decreases), then the impact is positive both on biodiversity and landscape quality;
- If in an MS/NUTS2 region the share of extensive land use will decrease (and intensive land use increases), then the impact is negative both on biodiversity and landscape quality;

The impacts will be expressed in the % of the positively changed area - the negatively changed area per MS/NUTS2.
For the first prototype only positive and negative impacts for biodiversity/landscape as a whole will be assessed in relation to changes in intensity of landscape. Specification in relation to different biota, e.g. birds, mammals, invertebrates, plants, will not be made either. For prototype 2 we might go into more detail if we think this is necessary and feasible.

4.3 Change in density and share of intensive/extensive livestock

For assessing the effects on the livestock density indicator the input comes from the present livestock patterns and the by the CAPRI model predicted changes in livestock mix and numbers under different compliance scenarios.

Like with the land use intensity indicator, the calculation of the share of intensive livestock and the its change under different compliance scenarios can only be done at the level of the whole NUTS 2 region in prototype 1. For prototype 2 this will also be estimated in a more spatially differentiated way however, taking account of the large distribution of farms over different intensity types.

The indicator LU/ha UAA will be directly used for certain livestock types (LU) considered always intensively managed, such as pigs and poultry. Therefore figures of LU of pigs and poultry per ha UAA at NUTS2 will be added directly to the "intensive pool".

But other types of livestock, such as dairy cattle, beef, sheep and goat, can be managed either in an intensive or extensive way and therefore an estimation of their intensity requires a further analysis. There are several indicators that can be used for estimating this intensity such as stocking density (LU/ha fodder or LU/ha of total UAA), or level of concentrate feeding per animal.

However, both indicators are difficult to translate directly into intensity figures. High stocking rates could be a sign of high intensity, but it could well be that these figures are distorted because communal grazing lands have not been included in the statistics and therefore in the calculation of the stocking density (no legal access rights have been signed for the use of communal lands and therefore they are not included in the farm statistics). High concentrate feeding per animal however does indicate towards introduction of external nutrients into the system. If this also goes together with a high stocking density per hectare of UAA it indicates towards a high intensity livestock system with a high nitrogen excess. The system and the livestock can be regarded as intensive. However on a regional level the livestock can still go into the extensive pool if there is little livestock in the region as a ratio of total UAA. Since for prototype 1, data to calculate the share of intensive and extensive livestock are only available as regional averages. The calculations for estimating the intensity of the different livestock types need to be based on simple assumptions looking at the average Nuts 2 situation. Only in prototype 2 we will use more detailed information taking account of variations in livestock numbers, densities and input levels at the level of farm types within Nuts 2 regions. This will enable us to make a better estimate of the relative share and spatial distribution of intensive and extensive livestock within Nuts 2 regions.

In Table 4.2 an overview is given of the variables available to classify the livestock types in intensity classes. From this table it becomes clear that the classification should preferably build on a combination of proxy indicators and not just 1. It should also be mentioned that the thresholds suggested in the Table are now indicative but can only be made definite after analysis of the data per NUTS 2 and over the whole EU. However, the threshold of 2 LU/ha seems to be a good threshold as this maximum is also taken as a proxy for estimating the areas where there is a risk of nitrogen emission above the maximum of 170 kg nitrogen/ha as specified in the Nitrogen Directive.

Table 4.2: Overview of indicators to be used for classifying the land dependent livestock classes in high, medium and low intensity livestock classes

	Main indicators					Only for dairy category
	LU/ha fodder on livestock farms	LU/ha UAA on livestock farms	LU/ha UAA all farms farms	Concentrate feeding/LU grazing livestock	Likely presence of communal grazing land	Milk yield/cow/year
Intensive	>=4	>= 2 LU	>=1.5 LU	high	low	> EU average
Medium intensive	>=4	>=2 LU	<1.5	high	low	<= EU average
Medium intensive	>=4	>=2 LU	<2 LU	high	high	<=EU average
Low intensive	<4	<2	<2	low	low/high	< EU average

Once the division in intensive and extensive livestock types has been made for the 2005 situation and for the predicted CAPRI-MITERRA livestock change under new scenarios with improved levels of compliance, the changes in intensive/extensive livestock type shares can be calculated.

The impacts of changes in livestock composition will be derived from the % LU in intensive, medium intensive and extensive categories:

- If in a MS/NUTS2 region the share of extensive livestock will increase (and intensive livestock share decreases), the impact is positive both on biodiversity and landscape quality;
- If in an MS/NUTS2 region the share of extensive livestock will decrease (and intensive livestock increases), the impact is negative both on biodiversity and landscape quality;

The impacts will be expressed in the % of the positively changed (+) or negatively changed (-) livestock per NUTS2.

4.4 Change in land use diversity

This assessment will comprise of the following steps:

1. The share of CAPRI agricultural land uses will be computed at NUTS2 level for the baseline situation and the scenarios.
2. These will be aggregated per NUTS2 into the 3 classes: arable crops, permanent crops, grasslands/fallow land.
3. The diversity will be calculated of these 3 classes for the baseline (2005 implementation levels) and additional compliance level scenarios, using the evenness part of the Shannon's Diversity Index.
4. The evenness of the compliance scenarios will be compared with the baselines evenness.
5. From this it can be derived where and in what degree an increase in land use diversity can be expected, assuming this will lead to higher landscape diversity, and a higher biodiversity.

Since CAPRI only predicts changes in agricultural land, this indicator relates exclusively to the agricultural components of the landscape, independently of the quantitative and spatial contribution that other landscape components (e.g. forest, shrubs, unproductive, water, marshes, etc.) might have to the entire picture.

Grouping of the CAPRI land use types

For this assessment the land use classes used in the CAPRI model will be classified according to similarity of structure and appearance. We are considering the following three classes on the basis of EUROSTAT definitions (see Table 4.3):

- arable crops,
- permanent crops,
- grasslands/fallow land.

Table 4.3 EUROSTAT definitions of land uses and grouping into arable crops, permanent crops and grassland/fallow land

Crops	**Arable**	**Permanent**	**Grassland e.o.**
Soft Wheat	X		
Durum Wheat	X		
Barley	X		
Rye	X		
Oats	X		
Maize	X		
Other Cereal	X		
Fallow Land			X
Rice	X		
Sunflower	X		
Soya	X		
Texture Crops	X		
Pulses	X		
Other Crops	X		
Potatoes	X		
Sugar Beet	X		
Root Crops	X		
Rape	X		
Tobacco	X		
Other Industrial	X		
Tomatoes	X		
Other Vegetable	X		
Flowers	X		
Other Fodder	X		
Permanent grassland			X
Nursery	X		
Fruits		X	
Citrus		X	
Olive		X	
Vine		X	

Shannon's Diversity Index (SHDI)

In this paragraph the Shannon's Diversity Index (SHDI) is explained, using the following source: Landscape metrics *From land cover to landscape diversity in the European Union*. This publication is the result of a close collaboration between three Services of the European Commission - DG AGRI, EUROSTAT and the Joint Research Centre (Ispra) - and the European Environmental Agency.

http://ec.europa.eu/agriculture/publi/landscape

The Shannon Diversity Index quantifies the diversity of the countryside based on two components: the number of different patch types and the proportional area distribution among patch types. Commonly the two components are named richness and evenness. Richness refers to the number of patch types (compositional component) and evenness to the area distribution of classes (structural component).

The Shannon Index is calculated by adding for each patch type present the proportion of area covered, multiplied by that proportion expressed in natural logarithm, according to the formula:

$$SHDI = -\sum_{i=1}^{m} (Pi * \ln Pi)$$

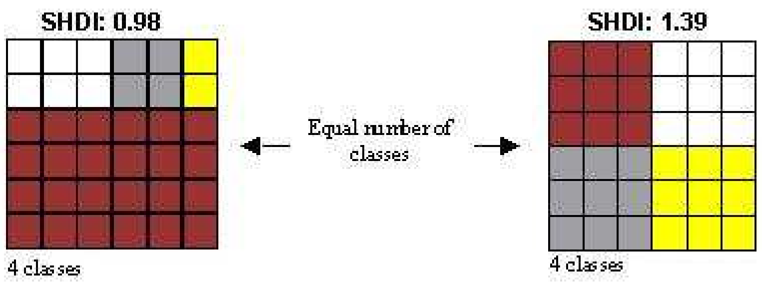
m = number of patch types

P_i = proportion of area covered by patch type (land cover class) i

Shannon Diversity Index increases as the number of different patch types (=classes) increases and/or the proportional distribution of the area among patch types becomes more equitable. For a given number of classes, the maximum value of the Shannon Index is reached when all classes have the same area. The following examples try to illustrate the influence of richness and evenness on the index.

In figure 4.1 the effect of evenness is shown: two different reference units, both composed of four classes, i.e. with an equal richness, are presented. The share of the surfaces occupied by the four classes varies. The effect of this variation of evenness is reflected by the SHDI: the more equal the share of the classes, the higher the Shannon Index.

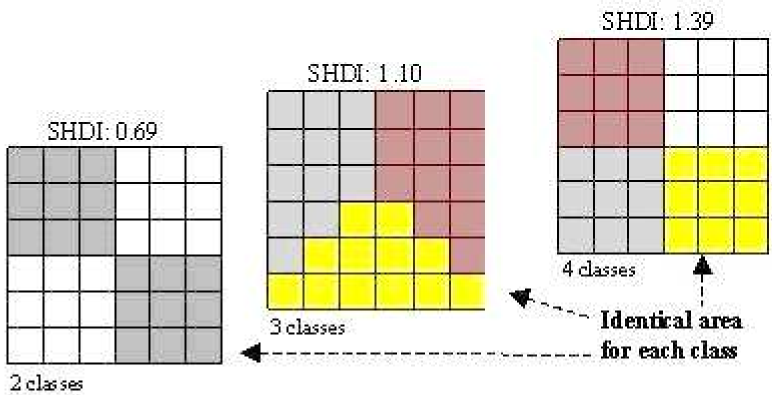
Figure 4.1: Influence of area proportion (evenness) of different classes on the Shannon Index



In the following example the evenness is constant i.e. the proportional percentage of the area covered by each class is constant, but the number of classes is rising (increasing richness).

As a result of the increasing number of classes the Shannon Index is increasing (figure 4.2). The Shannon Index can be used as a relative index enabling the comparison of different "landscape" units or enables their comparison at different times. However, due to the fact that the index is a combination of richness and evenness, the interpretation is somewhat difficult.

Figure 4.2: Influence of number of classes (richness) on the Shannon Index



In prototype 1 we will apply the Shannon index on NUTS2 level with the three land use classes mentioned earlier. Since we work with so few classes in large areas we assume that the number of classes will hardly have any effect on the results. Therefore we propose to use only the evenness part of the Shannon Index in the first prototype.

In prototype 2 we might have spatially more detailed information on land use, and we might then consider increasing the number of land use classes and using the complete Shannon Index.

5 Approach to assess impacts on habitat quality derived from environmental indicators

The following environmental indicators will be used for this impact assessment:

- Chemical soil quality: Gross balances for carbon, nitrogen and phosphorous.
- Ground and surface water quality: Nitrate leaching to ground water and runoff to surface water from agriculture.
- Air: Ammonia emission (NH3) to air

In Chapter 2 from the literature review the relationships between biodiversity and emissions of nutrients to soil and water and ammonia to air have already been discussed extensively and confirm that the lower the emissions the better for biodiversity overall. It will therefore be assumed that if MITERRA predicts a lowering of emissions in the additional compliance scenarios as compared to the baseline, the better it is for biodiversity. For the interpretation of changes in the environmental qualities we will therefore generally assume that an increase in environmental quality, will lead to an improvement of biodiversity. How large this relative improvement will need to be before it is really interpreted as an improvement of quality will still need to be defined, but can only be done after extensive analysis of the modelled MITERRA output.

Contrary to the case of the potential effectiveness approach, compliance levels are already internalised in the calculation of environmental indicators in the CAPRI/MITERRA models whose output is used here.

The way these indicators will be computed has been outlined in D2.3, Chapter 3.

We consider showing in the CCAT tool also an overlay of the results of the effectiveness of the standards assessment with the (change in) values of the environmental indicators. This overlay will show where biodiversity/landscape targeted measures might be more effective because they coincide with improvements in the environment and habitat quality.

6 Further work and data needs

Data needs for prototype 1 assessments

Overall it is clear that for prototype 1 there will be 3 types of assessments:

a) An expert qualitative estimate of the effectiveness of standards for biodiversity and landscape;
b) Assessments of impacts induced by predicted land use changes as a consequence of Cross Compliance;
c) Impact assessments on habitat quality derived from environmental indicators.

Data requirements for these assessments are given in Table 6.1.

Table 6.1: Overview data needs for specification of the selected indicators for land use, biodiversity and landscape in prototype 1

Assessment	Indicator	Data needs for assessment	Data sources
An expert qualitative estimate of the effectiveness of standards for biodiversity and landscape	Effectiveness of CC standards for biodiversity and landscape	a. Short name descriptions of all SMR and GAEC standards per region included in PT1 for whole EU for as far as available b. UAA and land use per farm type c. Estimation of Compliance levels per standard per farm type per region in 2005	a. Existing data sources and additional data collection in CCAT b. COCO and FSS c. Existing data sources, additional data collection CCAT and own best estimates
Assessments of impacts induced by predicted land use changes as a consequence of Cross Compliance	Changes in intensive and extensive crop share	a. Present land use shares (35 land use classes) b. Future land use changes c. Average input levels per land use per region and changes in input levels	a. COCO/FADN data (available) b. CAPRI-MITERRA model output c. COCO/FADN data and modelled output CAPRI-MITERRA
	Changes in intensive and extensive livestock share	a. Present numbers and composition of livestock population per region b. Future changes in numbers and composition of livestock population per	a. COCO/FADN data (available) b. CAPRI-MITERRA model output c. COCO/FADN data and modelled output CAPRI-MITERRA

		region c. Average stocking density, milk yield levels, input levels per region and changes stocking density, yield and input levels	
	Changes in land use diversity (evenness)	a. Present land use shares (35 land use classes) b. Future land use changes c. Data on nationally protected sites and tourist attendance	a. COCO/FADN data (available) b. CAPRI-MITERRA model output c. SENSOR project
d) Impact assessments on habitat quality derived from environmental indicators.	Change in habitat quality caused by CC standard's effects on environment	a. Emissions of ammonia (kg NH_3-N/ha/yr b. Nitrates in water, including leaching in kg N/ha/yr and concentrations in mg NO_3/l (water quality). c. Gross Balances for Carbon Nitrogen and Phosphorous	a. and b. and c. CAPRI-MITERRA model output

As for the assessment of the effectiveness of standards for biodiversity and landscape, there is a need to obtain clear information on the way SMRs and GEACs have been implemented. Such information should be already partly be available from other projects and partly still needs to be collected by WP 3. WP3 will provide short names on the implementation of SMRs and GEACs which is basically a summary of the full text of SMRs in the member states and other official national documents on which and how GEACs are implemented. They provide a brief characterisation of the standards in relation to those factors that are of importance for assessing their potential impacts, including those on biodiversity and landscape.

At this stage it is however clear that short names for SMRs in the new Member States are however not available yet, as these countries are still in the process of developing them (since they do not need to be implemented until 2011). This is however not the case for the GEACs, which means that in prototype 1 practically all GEAC standards will be involved in the assessment both for old and new MS.

Beside information of the SMR and GEAC standards, we also require data on the regional share of UAA (or the share of a specific land use to which the standard is targeted, e.g. olive groves) to weight the potential effectiveness per NUTS2. The logic is that the greater the UAA, where the standards are to be implemented, the higher their potential effect on biodiversity will be. Both, share of UAA and absolute hectares of UAA at NUTS2 level will be used as weighting factors, reflecting respectively the magnitude and the extent of the potential effects. Since certain CC and SMRs standards are targeting particular land uses and crops, such as cereals, permanent grasslands, permanent crops or olive groves, the possibility to particularize in these cases the weighting exercise according to the regional share of these land uses will be needed. This will require a good relative division of all agricultural land uses per NUTS 2.

To come from a *potential* effectiveness to an estimate of the *expected* effectiveness data are also required on the real level of compliance of the different standards. This requires estimates on the land use share per NUTS 2 region estimated to be compliant with different standards under 3 different scenarios of compliance (the baseline situation in 2005, 75 % and 100 % compliance). However, if it turns out to be problematic to estimate these shares of compliance for certain standards per land use category we will work with the regional average compliance levels for all land uses.

The data for the *assessment of the changes in share of intensive/extensive land use* come from the CAPRI database (COCO), which specifies 35 different land use categories (for the base line situation which is 2005) and the CAPRI model output for the future situations. CAPRI works with the same land use classes as in FSS which includes 34 different crops and permanent grassland. Information on certain input levels will be derived from the pre-model CAPRI input data which include estimates on input levels for different crops both in terms of artificial fertilisers, agro-chemicals and irrigation. It is expected that there are no data gaps occurring for the calculation of this indicator in prototype 1. For prototype 2 it is expected to further improve these indicators to a more spatially detailed level, instead of working with NUTS 2 averages, and this requires input of down-scaled data on land use and farm management data coming from SEAMLESS and DYNASPAT projects and from post-model disaggregation approaches to be applied in CCAT in prototype 2.

For assessing the effects on the *livestock density indicator* the input comes from the present livestock patterns and the - by the CAPRI model predicted - changes in livestock mix and numbers. Additionally for the assessment for certain livestock types (LU) such as dairy, beef, sheep and goat, which can be managed either in an intensive or extensive way, an estimation of their intensity will be made based on stocking density, use of concentrate feeding per animal and milk yield per cow. Again the main data source for this assessment is the CAPRI database, COCO, FADN and FSS data and modelled output of CAPRI on changes in livestock numbers and livestock composition.

Also for this indicator for prototype 2 it is expected to further improve these indicators to a more spatially detailed level, instead of working with NUTS 2 averages. It will require input of down-scaled data primary data on livestock numbers and types and farm management data coming from SEAMLESS and from post-model disaggregation in CCAT of CAPRI modelled output.

For the assessment of changes in *land use diversity (evenness)* in prototype 1 the land use classes used in the CAPRI model according to the EUROSTAT definitions will be classified according to similarity of structure and appearance. The diversity will be calculated by using the evenness part of the Shannon's Diversity Index. For the assessment of the landscape diversity data on land uses will come from the CAPRI database (COCO), which specifies 35 different land use categories and the CAPRI model output for the future situations. Beside two additional indicators will be used in prototype 1, depending on the availability of the required input data: Nationally protected sites/landscapes/ World heritage sites and Tourist attendance, non-residential/ Tourist attendance residential. The required data can most likely be obtained from the EU SENSOR project.

The area and share of semi-natural (extensive) habitats (e.g. fallow, permanent grassland, hedgerows, and other linear elements) have also originally been planned to be used as an indicator for landscape diversity. However as described for the biodiversity indicators, this indicator can not be used mainly because in the CAPRI model it is not yet feasible to distinguish between improved grassland and semi-natural grassland. Therefore, it will have to be postponed to prototype 2. Furthermore, like with the two other indicators on intensity, in prototype 2, spatially more detailed data will become available through pre- and post-model disaggregation approaches enabling the specification of the changes in land use and livestock intensity and the evenness indicator on a higher spatial detail.

The selected environmental indicators reflecting a change in habitat quality to be used in prototype 1 are Emissions of ammonia in kg NH_3-N/ha/yr (air quality) and Nitrates in water, including leaching in kg N/ha/yr and concentrations in mg NO_3/l (water quality). They will be derived from the output produced by the environmental models. For the first prototype the output of the environmental model MITERRA (in the form of environmental indicator values) will be used as input for a qualitative assessment of effects on farmland biodiversity within regions.

Further work for prototype 2

Regarding the land use based assessments following extensions will be made in prototype 2: For the impacts of land use change as a consequence of Cross Compliance in the first prototype only positive and negative impacts for biodiversity/landscape as a whole will be assessed, without specifying for e.g. birds/mammals/ invertebrates/ plants. In a later stage we might go into more detail if we think this is necessary and feasible. For the assessment of changes in land use diversity due to Cross Compliance standards in prototype 2 we might have spatially more detailed information on land use than on NUTS2 level, and we might then consider to increase the number of land use classes and use the complete Shannon Index (only the evenness part of the Shannon Index will be used in the first prototype).

In a later stage, for **prototype 2**, it is further envisaged that:

2) The environmental models will be applied to environmental regions which are smaller then NUTS2 regions and which are characterised by a more homogeneous environment. Model calculations will then deliver a better picture of the CC effects taking account of the larger variation in combinations of farming practices with much localised bio-physical environmental factors.

3) More detailed combinations between the qualitative assessment of pressures on different impact fields of biodiversity/landscape and the present state of biodiversity will be made. It will be investigated whether it is possible if state data are available to make a prediction of changes in certain species groups using either qualitative or quantitative relationships between farming practices and species numbers.

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