UNIVERSIDADE TÉCNICA DE LISBOA

INSTITUTO SUPERIOR TÉCNICO

Sustainable Land Uses and Carbon Sequestration:

The Case of Sown Biodiverse Permanent Pastures Rich in Legumes

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(Mestre)

Dissertação para obtenção do Grau de Doutor em Engenharia do Ambiente

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Setembro de 2010

Resumo

A presente dissertação em Engenharia do Ambiente consiste no estudo do sistema de pastagens permanentes semeadas biodiversas ricas em leguminosas (PPSBRL) como opção sustentável para mitigação e adaptação a alterações climáticas. Utilizando várias ferramentas de avaliação de sustentabilidade, determina-se o potencial de sequestro de carbono em PPSBRL e em pastagens naturais, que são a principal alternativa em termos de uso do solo. O aumento do nível de matéria orgânica no solo é a variável chave para o sequestro de carbono e também para os restantes serviços ambientais providenciados por estas pastagens. Por último, apresentam-se as formas de valorizar economicamente os serviços ambientais quantificados anteriormente, nomeadamente através da remuneração do sequestro de carbono pelo Fundo Português de Carbono (FPC), ou da valorização pelos consumidores de carne.

Concluiu-se que as PPSBRL como uso do solo têm vantagens ambientais e económicas generalizadas. Para além de sequestrarem carbono a uma taxa de 5 t $CO_2.ha^{-1}.a^{-1}$, são pelo menos tão positivas para a biodiversidade selvagem como as pastagens naturais, reduzem a erosão do solo e controlam o ciclo da água. Em termos económicos, o apoio do FPC será crucial para aumentar a área deste uso sustentável do solo.

PALAVRAS-CHAVE:

- Avaliação de sustentabilidade;
- Serviços de ecossistema;
- Pastagens;
- Matéria orgânica do solo;
- Carbono;
- Avaliação económica.

Abstract

The present PhD. thesis in Environmental Engineering consisted on the study of the sown biodiverse permanent pastures rich in legumes (SBPPRL) system as a sustainable option for mitigation and adaptation to climate change. Using several sustainability assessment tools, we begin by quantifying carbon sequestration in SBPPRL and the alternative land use of natural grasslands. Increased soil organic matter is the key variable for carbon sequestration and also for other environmental services in these pastures. We end by presenting ways to economically value those environmental services. We namely study the possibility of payments for carbon sequestration by the Portuguese Carbon Fund (PCF), and direct valuation by meat consumers.

We concluded that SBPPRL are a win-win land use, since they have widespread environmental and economic advantages. Besides sequestering atmospheric carbon at an average rate of 5 t $CO_2.ha^{-1}.yr^{-1}$, they are at least as good as natural grasslands for wild biodiversity, reduce soil erosion and control the water cycle. Economically, support from the PCF will be crucial to increase the area of this sustainable land use.

KEYWORDS:

- Sustainability assessment;
- Ecosystem services;
- Pastures;
- Soil organic matter;
- Carbon;
- Economic valuation.

Acknowledgments

The past four years have been an amazing journey. I met so many people that my list of telephone numbers increased exponentially. Since most of these people, directly or indirectly, personal or work wise, helped and influenced me, I believe that it is only fair to refer to them here. So please, bear with the long list that follows for a while.

First and foremost, I would like to thank my advisor, Tiago Domingos. I thank him for complying with the task of directing and correcting my work, patiently and always with a smile. Working with him is a tremendous learning experience. Whenever a difficulty arises, we know we can rely on his judgement – he always finds a way around it. His commitment, optimism and amazing problem-solving abilities are overwhelming. I would have never been able to complete this PhD. thesis under guidance from anybody else.

Sometimes we are lucky to create special bonds with people we spend our whole days with in the workplace. I want to particularly thank, from the people I met at Secção de Ambiente e Energia (SAE, where I have been working for the past four and a half years), my dear friends Ana Simões and Tatiana Valada. During my first years, I learned a lot from working with Ana, who I got used to looking up to. She has the strongest presence I have ever seen in anyone, and she is a reference in all aspects of my daily life. Tatiana became the best friend you can have in the workplace (or anywhere, for that matter). She is so caring, kind and dedicated, always looking to help. Both of them made me enjoy being in IST. They actually made me look forward to Mondays and getting back to work. Their presence lights up the room and makes work fun. In fact, I learned a lot just from who they are. They supported me through the hard times, and laughed with me through the good times. They showed me the best part in being human, and are both lessons in good will for me. I was also fortunate to find friendship and support in Nuno Rodrigues and André Serrenho, two of the greatest and funniest guys I ever knew. I have the luck of working directly with Nuno and his laid-back style has been essential to balance me in my constant stress. And for the existence of this thesis I can thank no one more than Alexandra Marques, who made me overcome my biggest obstacle: starting to write and finding my motivation. Even if we don't work together again soon, I will never forget our winter weekend afternoons working in IST.

I also want to thank several other important people in my life. Clara Fiúza, my best friend for quite some time now, who is always ready to support and guide me when I feel lost, and shares my darkest humor; Lígia Reyes, who has driven me away from work in so many afternoons and nights, who gave new goals and dreams, but always kept me focused on who I am; Maria João Faustino, traveler partner, advisor, counselor and confessor, who, after all the ups and downs that life has put us through, understands me like no one; Paulo Martins, my oldest friend, since we have grown up together and he has been in all the stages and changes that brought me to this day; and finally, my parents, without whose support I would not have even been halfway of where I am today, both academically and as a person. Though I may not always have been the best of friends (or of sons) to you, I deeply thank you. You are all an essential part of me. You provided me with the support and understanding I needed when I was away from work, so that when I was working I could feel balanced and comfortable enough to be productive.

I also wish to thank four special friends I made during my adventure in the Masters course in Economics at Instituto Superior de Economia e Gestão (ISEG). Gui Pedro Mendonça, José Jardim, Nuno Costa and Renata Mesquita helped me to understand Economics. Before our long philosophical and political discussions, Economics was like a foreign language to someone like me, an Environmental Engineer.

I must also name the Masters students in Environmental Engineering of which I was or am the co-advisor: Ana Catarina Henriques, Carlos Lopes, Francisco Amaral, Gonçalo Abrunhosa, Miguel Rodrigues and Natasha Lemos. During the course of their dissertations, I learned probably as much as they did.

I would also like to thank all my colleagues in SAE that I haven't referred before, which was a second home to me in the last years (well, many times it was practically the first): Alexandra Nogal, Ana Gonçalves, Ana Rosa Trancoso, António Lorena, Carlos Teixeira, Cátia Rosas, Cristina Marta-Pedroso, Gonçalo Domingues, Gonçalo Marques, Joana Abreu, João Fernandes, João Rodrigues, Jorge Palma, Josefina Tomé, Marco Reis, Maria Guadalupe Saião, Nuno Sarmento, Oriana Rodrigues, Rui Mota, Sónia Barbosa and Tânia Sousa. I thank them for useful comments and recommendations, but, above all else, I thank them for a fun and healthy working environment.

I would also like to thank the co-authors of the papers resulting from this dissertation, namely João Dias, A.P.S.V. Costa, Ramiro Oliveira, Lídia Farroupas, Fátima Calouro, Ana Barradas, João Paulo Carneiro, Paulo Canaveira, Teresa Avelar, Gottlieb Basch, Carlos Carmona Belo, David Crespo, Vítor Góis Ferreira and Casimiro Martins. I also acknowledge useful comments by Nuno Calado, Pedro Chambel Leitão, Alexandra Lopes and Carlos Aguiar. I particularly thank Helena Martins who revised the final version of the thesis from cover to cover.

I thank most data I used to projects "AGRO 87 – Biodiverse Permanent Pastures Rich in Legumes", developed between 2001 and 2004, and "AGRO 71 – Recovery and Improvement With Pastures of Degraded Soil in Alentejo", from 1997 to 2004.

Part of this work was supported by project Extensity – Environmental and Sustainability Management Systems in Extensive Agriculture, funded by the Life Program of the European Commission (LIFE03 ENV/P/505) and by Project PTDC/AGR – AAM/69637/2006, funded by Fundação para a Ciência e Tecnologia. This work was funded by Fundação para a Ciência e Tecnologia through grant SFRH/BD/25399/2005.

In a final note, I want to acknowledge the groundbreaking role of David Crespo in Portuguese agriculture. His idea inspired this work.

Ricardo Teixeira

August 2009

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Glossary

AFOLU	Agriculture, Forestry and Other Land Uses
ALE	Alentejo
APA	Portuguese Environmental Agency (*Agência Portuguesa do Ambiente*)
BD	Bulk Density
BI	Beira Interior
CDM	Clean Development Mechanism
C	Carbon (atomic)
CO_2	Carbon dioxide (molecule)
CO_2e	Carbon dioxide equivalent (GHG converted to CO_2e using GWP)
DALY	Disability Adjusted Life Years
DDG	Dry distilled grain
DM	Dry Matter
EC	European Commission
EI95	Ecoindicator 95
EI99	Ecoindicator 99
EMAS	Eco-Management and Audit Scheme
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FNG	Fertilized Natural Grasslands
FSC	Forest Stewardship Council
GHG	Greenhouse Gases
GSN	Guaranteed Sustainability Norm
GWP	Global Warming Potential
IACA	Portuguese Association of Producers of Commercial Feeds for Animals (*Associação Portuguesa dos Industriais de Alimentos Compostos Para Animais*)
IC	Impact Category
INE	National Institute of Statistics (*Instituto Nacional de Estatística)*
IOA	Input-Output Analysis
JI	Joint Implementation
KP	Kyoto Protocol
LCA	Life Cycle Assessment
LPN	Liga para a Protecção da Natureza

LU	Livestock Unit
LULUCF	Land Use, Land Use Change and Forestry
MA	Millennium Ecosystem Assessment
NEP	Net Ecosystem Production
MBD	Mineral Bulk Density
NG	Natural Grasslands
NPP	Net Primary Production
NPV	Net Present Value
NV	Normalization Value
ptMA	Portuguese Millennium Ecosystem Assessment
PCF	Portuguese Carbon Fund
PNAC	National Programme for Climate Change (*Programa Nacional para as Alterações Climáticas)*
PPP	Purchasing Power Parity
QF	Quinta da França
RC	Replacement Cost
RO	Ribatejo e Oeste
SBPPRL	Sown Biodiverse Permanent Pastures Rich in Legumes
SIP	Sown Irrigated biodiverse permanent Pastures
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
USLE	Universal Soil Loss Equation
VSL	Value of a Statistical Life
VSLY	Value of a Statistical Life Year
WF	Weighing Factor
WTP	Willingness To Pay
YLD	Years Lived Disabled
YLL	Years of Life Lost

List of units

Symbol	Designation	Definition
Prefixes, suffixes and symbols		
c	*centi*	$= 10^{-2}$
%	*percentage*	$= 10^{-2}$
k	*kilo*	$= 10^{3}$
M	*mega*	$= 10^{6}$
ppm	*parts per million*	$= 10^{-6}$
< x >	*quantity of x per unit of mass*	(used in Chapter 3)
{ x }	*quantity of x per unit of area*	(used in Chapter 3)
Mass units		
g	*gram*	
t	*ton*[1]	1 t = 1 Mg
Time units		
s	*second*	
day	*day*	1 day = 86 400 s
yr	*year*	1 yr = 31 536 ks
Length units		
m	*meter*	
Energy units		
J	*Joule*	
cal	*Calorie*	1 cal = 4.184 J
Other units		
Pt	*Ecoindicator points*	

[1] The unit "ton" can also be written as "tonne". We adopted the first designation in this thesis.

Foreword

In the fall of 2005, I became a member of the Project Extensity team. There was a presentation for the European Commission which took place in Herdade dos Esquerdos, Vaiamonte, where the system of Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL) was developed. During that day, we were guided in the farm by Eng. David Crespo. He was the person responsible for the development of the system.

The most striking fact about this pasture system that I knew nothing about at the time was that the fields were still green, even though it was already Fall. Non-SBPPRL pastures from neighbours were already showing signs of decreased production. How could it be that two fields side by side could exhibit such different endurance?

At one point, Eng. David Crespo stated the following:

“In this land, due to SBPPRL, I have been responsible for the sequestration of enough carbon to compensate my emissions, my children’s emissions, and my grandchildren’s emissions too.”

This thesis is aimed at proving him right or wrong.

1. Introduction

This thesis' main focus is to determine the carbon sequestration potential of one particular pasture system, that of Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL), and its other environmental co-benefits. We study whether SBPPRL are a possible response to the decline in ecosystem services provided by pastures in Portugal.

The present Introductory Chapter is used to define the scope and objective of this thesis. We begin by stating the Portuguese obligation in Kyoto Protocol and by justifying the relevance of studying carbon sequestration in pastures. Since carbon sequestration is an ecosystem service, we state the relation between ecosystem services and agricultural sustainability, in order to show the importance of pastures in the subject. We define the system of SBPPRL in terms of its composition and a priori *effects. Since livestock is one of the key parameters in pasture systems, we then turn to the definition of scenarios for livestock production that we will use later on in this thesis. We end this Chapter by presenting an overlook of the next Chapters of the thesis.*

1.1 The Kyoto Protocol and policy instruments

1.1.1 Setting international standards

The urgent need to respond to the alleged threats of climate change led to the United Nations Framework Convention on Climate Change (UNFCCC, 1998). The framework of this Convention paved the way for the appearance of a mandatory agreement between most countries, stipulating for each a maximum emissions scenario (Harvey, 2004). This agreement was named after the Japanese city in which it was signed, and became known as the Kyoto Protocol (KP). The KP set quantitative targets for each country or assembly of countries, in order to reach a worldwide reduction in net GHG emissions of 5% by 2010, in relation to 1990. The finishing level is obtained by averaging emissions in the years between 2008 and 2012. If any country cannot keep its designated upper bound, there are four mechanisms that it can recur to:

- The carbon-trading scheme, in which over-compliers sell their exceeding credits to under-compliers using a market price;
- Clean Development Mechanisms (CDM), in which developed signatory countries execute projects leading to a GHG emissions reduction in developing countries, and the credits revert to their favor;
- Joint Implementation (JI), in which developed signatory countries execute projects in other signatory countries, with the same effect as CDM;
- Investments in funds managed by independent third parties, or other alternative instruments.

Unlike many important polluters, like most notably the United States of America, Portugal signed the KP. We now turn to the national consequences of the agreement.

1.1.2 The Portuguese target

In the KP, Portugal agreed not to increase its GHG emissions, in relation to 1990, by more than 27%. According to the Portuguese Environmental Agency (APA, 2006a), Portugal is one of the European Union (EU) countries with the lowest *per capita* GHG emissions. However, in the period from 1990 to 2003, its emissions increased 37%, an increase over the KP limit (APA, 2006b). The Portuguese deficit will likely be about 3.73 Mt $CO_2e.yr^{-1}$.[2]

Faced with the risk of under-compliance of the KP, Portugal created an instrument of analysis, which was the National Programme for Climate Change (PNAC, 2006). The function of PNAC is to assess the current situation in terms of the compliance of the KP, build scenarios for future trends, and try to determine additional measures necessary to meet the national goal.

PNAC defined additional measures in almost all economic sectors, and gave birth to an emissions limit for polluting industries within the country (APA, 2006a). Amongst those additional measures, Portugal decided to elect some optional mechanisms, which countries are not required to account for in their inventories. The next section is about such mechanisms.

1.1.3 Optional mechanisms of the Kyoto Protocol

There are strict stipulations in the KP as to how a country's emissions inventory is made, namely regarding what to account. However, there are some items that remain as an option for each signatory country. These options relate to the agro-forestry sector, and are the so-called Land Use, Land Use Change and Forestry (LULUCF) activities, now renamed Agriculture, Forestry and Other Land Uses (AFOLU), under the framework of Article 3.4 of the KP. While most sectors are net polluters, where all that can be done is to minimize CO_2 emissions, and energy biocrops and many sub-products may also be used to produce renewable energy which substitutes fossil fuels, AFOLU acticities are responsible for CO_2 sequestration in soils and living biomass. Therefore, AFOLU activities (IPCC, 2006) do not promote a decrease in emissions, but rather the sequestration of CO_2 (which is not permanent[3]).

Portugal plays a leading role in its account in the KP, since it decided to elect, in the framework of these voluntary AFOLU activities under Article 3.4 of the KP, the activities: "Grassland Management", "Cropland Management" and "Forest Management". The rationale for this choice will be addressed latter.

However, even using such additional measures as these, PNAC (2006) still pointed to an excess in emissions. This requires Portugal to search for new possibilities to compensate the high emissions. To such effect, a Carbon Fund was established by the Portuguese Government in 2006.

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[2] The unit "CO_2e" refers to carbon dioxide equivalent, which is obtained by converting all greenhouse gases into the CO_2 equivalent using their specific global warming potential. This is shown in Section 3.2.

[3] The KP does not regard the issue of permanence, and therefore this mechanism, in the present period, is equivalent to emissions reduction.

1.1.4 The Portuguese Carbon Fund

The Portuguese Carbon Fund (PCF) is an operational instrument which intends to finance several actions with positive returns regarding a decrease in GHG emissions. These actions must be additional to those considered by PNAC, since they mean to fill the current gap of emissions. The fund was started in 2006, with an initial sum of 6 000 000 €.

The fund may be used to acquire credits using one of the four resorts considered in the KP, or, alternatively, it may be used to finance national projects. Even though the political priorities are yet to be defined, it is possible to assume that national projects are the most interesting option. First, the use of any KP scheme would mean that Portugal would be indirectly investing in forestation or energy efficient projects elsewhere in the world. Second, while CDM and JI have many practical implementation difficulties, carbon-trading is subjected to market uncertainties and price fluctuations that make it unreliable as a long-term policy. The emissions trading scheme was, however, designed to guarantee that reductions occur where it is cheaper to generate credits (Wagner and Wegmayr, 2006). The international price thus sets a standard for the price of national projects.

National projects may respect to permanent emissions reduction, or to carbon sequestration. If national projects were used, then the investments would occur in national territory, and since projects may be screened by the PCF itself (instead of indirectly acquired as a cabon credit), there is a possibility of strong complementarities between GHG reduction and other environmental and policy objectives.

1.1.5 Is agro-forestry important in the global carbon balance?

The rise in GHG concentrations in the atmosphere is mostly due to fossil fuel consumption. However, the agricultural and forestry sector has an important role to play in the global carbon cycle. The classical examples are intensive agriculture (due to fertilizer use, for example) and deforestation. But a significant fraction of the responsibility for emissions in this sector is due to livestock and land use changes.

Greenhouse gases and livestock production are deeply related, as shown by the Food and Agriculture Organization of the United Nations (FAO) (Steinfeld *et al.*, 2006). In a recent report, FAO states that the world's livestock sector is responsible for 18% of greenhouse gases emissions (measured in CO_2 equivalent), which is a higher share than transport. Reduction strategies have targeted intensive systems, focusing on nitrogen runoff and emissions related to fuel and fertilizer inputs, whereas efforts to lower methane emissions have focused on the more extensive systems, where lower productivity implies higher methane emissions per unit of product (Subak, 1999). And it is not just livestock that is a problem. The whole chain of meat production has high GHG emissions. Tukker *et al*. (2006) conclude that food and beverages are one of the three components of European Union consumption with the largest environmental impacts, with meat consumption being the most important item within this group. This fact gave rise to a specific study just for the meat sector (Weidema *et al.*, 2008). Emissions from livestock are deeply related to meat consumption (Teixeira and Dias, 2008).

It is also believed that land use and land management practices are responsible for 12 to 42% of total GHG emissions (Watson *et al.*, 2000). It is also estimated that about 80%

of terrestrial carbon pools are stored in soils (Watson *et al.*, 2000). In the European Union (27 countries), total carbon stocks are estimated around 75 billion tons of C[4], 50% of which is located in Sweden, Finland and the United Kingdom due to the large area of peatlands in these countries (Schils *et al.*, 2008).

Changes in carbon stocks occur due to land uses or land use changes. There are steady-state differences between soil carbon levels in different land uses. Soils typically accumulate carbon at a high rate in grasslands and forests, which are net sinks (0-10^{11} t.yr^{-1} C), and to a lesser degree in croplands (1-4 x 10^{10} t.yr^{-1} C). Therefore, carbon losses are expected to occur following conversion to cropland, and increases when croplands are converted to grasslands or forests (Schils *et al.*, 2008). Even though forests are usually considered to accumulate more soil carbon than grasslands, some studies indicate otherwise (Ganuza and Almendros, 2003). Nonetheless, for a while forests store carbon in trees, and not only in soils. If timber is then used in long-lasting goods (e.g. furniture, houses), C may be stored indefinitely. The conversion between land uses itself, when resorting to tillage, implies a spike emission of carbon to the athmosphere, due to increased aeration of deep layers of soil, an corresponding mineralization.

The current balance of the agro-forestry sector in Portugal is negative (Pereira *et al.*, 2009b). According to APA (2006b), the Portuguese agro-forestry sector contributes to 10% of the country's total GHG emissions. This sector has increased its emissions in 7% since 1990, thus contributing to the national deficit. Considering particular GHG, agriculture is responsible for 65% of national N_2O emissions (APA, 2006b), associated with nitrogen fertilizer use and manure management (EEA, 2006). Agriculture is also responsible for 35% of national CH_4 emissions (APA, 2006b), mainly due to animal production (EEA, 2006).

Therefore, grasslands seem to be an important type of land use. They not only store large amount of C, but are also typically grazed by livestock, and therefore are related to two of the key issues respecting to GHG balances. Besides, in Portugal, many grasslands occur in low-density oak forest areas, and therefore they are an important link to sustainable landscape management, with implications far beyond carbon sequestration.

Recognizing this fact, FAO prepared a document (FAO, 2009) based on the results of a workshop held at Rome from 15 to 17 April 2009. The meeting featured several experts and members of the Grassland Carbon Working Group. In the document, FAO advocates the enhancement of carbon sequestration in grasslands as a low cost mitigation option with important environmental and economic co-benefits. The main goal of this document was to spur the discussion on the inclusion of land use and land use changes at COP15 in the Copenhagen summit and afterwards.

The calling made by FAO included a call of attention to the possibility of carbon sequestration in grasslands. The potential has been established for a long time, but questions regarding methodologies used and feasibility potential still remain. According to FAO, original sequestration enhancement possibilities are welcome. In those solutions, the dynamics of CO_2 and soil carbon in the atmosphere-soil-plant system should be connected with beneficial impacts in soil, water and biodiversity. Carbon sequestration via AFOLU activities would be especially interesting if it has other complementarity co-benefits.

[4] Throughout the present thesis, we use the terms "CO_2", "carbon dioxide" and "carbon" as synonyms. When we are referring to atomic carbon, we use only the letter "C".

Portugal may have one of those original and cost-effective solutions with widespread benefits (Domingos *et al.*, 2009). In order to understand how, we now turn to the history of pastures in Portugal.

1.2 Main factors influencing ecosystem services and impacts in pastures

1.2.1 The history of pastures in Portugal

Since the beginning of the domestication of livestock by humans, fields of herbaceous plants, named in this sense as “pastures” or “grasslands”[5], have been used to feed animals. According to Diamond (1999), domesticated animals must have a flexible diet, consisting mainly of food such as grasses and forages, which are not part of human diet. This makes them less expensive to be kept in captivity than, for instance, carnivores.

Traditional pastures feature only spontaneous species, and were thus grazed by small herds requiring large areas. Figure 1 shows how pastures have become very important in Portuguese agriculture. We can see the decline in agricultural land, part of which was converted to forests and an even larger part that became “abandoned” land. Agricultural abandonment in the last decades in Portugal paved the way for extensive animal husbandry systems to take the place of croplands. When a previous agricultural land is abandoned, a primary mechanism of natural succession takes place. It is thereafter invaded by native herbaceous species first, and then by shrubs. Some of these plant species are palatable but certain types of livestock. As a consequence, many abandoned lands carried on as usable in their most productive seasons of the year as “natural” pastures. Traditional animal production is less labor-intensive and less input-demanding, and so it was one convenient solution to the economic situation in rural areas (Pereira *et al.*, 2009a).

It should be noticed that part of the loss in agricultural area also translates the fact that less area was required for the same level of production, due to the introduction of production factors such as improved seeds, pesticides, mechanical means and chemical fertilizers as substitutes of organic fertilizers. These new technologies led to productivity increases (measured in production per hectare) in areas where the agricultural activity was intensified.

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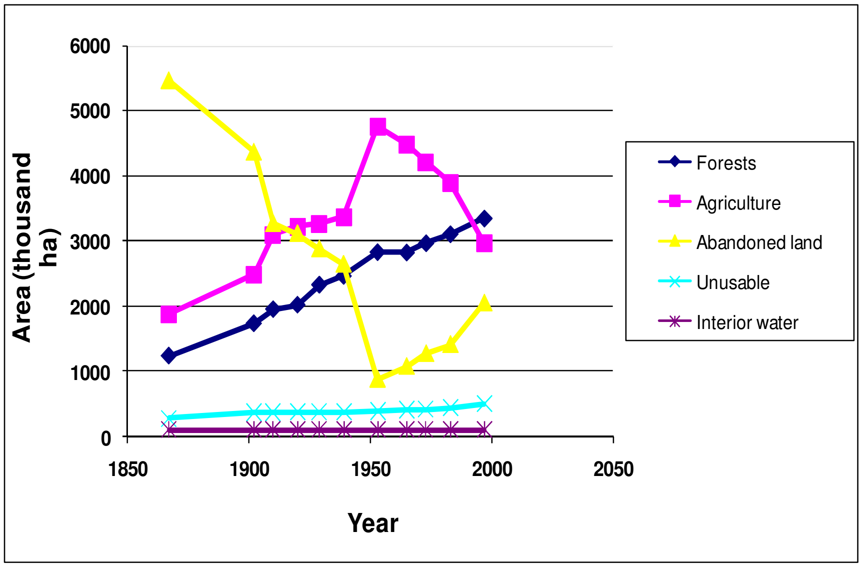
[5] The term “grassland” is usually referred to signify the plant system apart from other components, and pasture to express the whole grazing system (including animals). Since there is no significant gain in meaning from separating the two, and to keep language simple, in our work, we consider the terms “grassland” and “pasture” as synonyms.

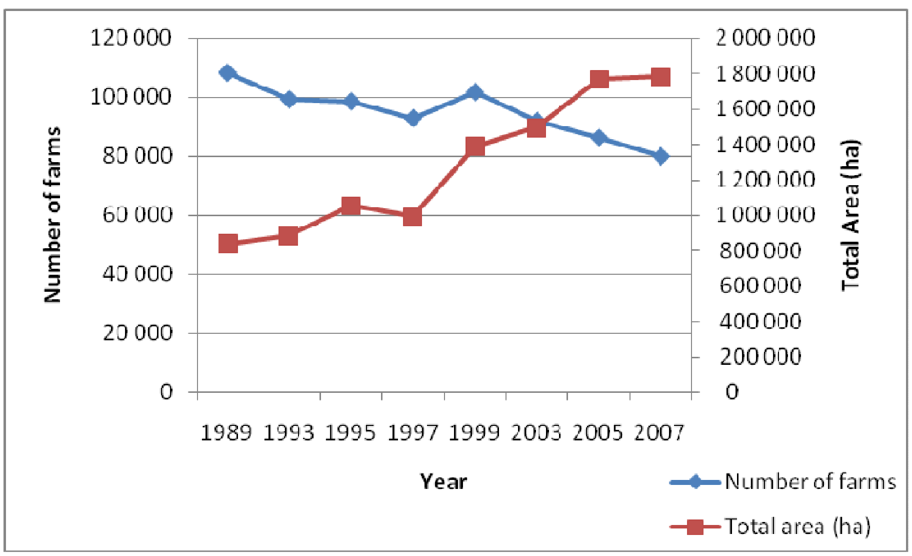
Figure 1 – Relative land use changes* in Portugal from 1850 to 2000.



*** The sum of all uses in each year is equal to the total area of the country. "Unusable area" is composed mainly of urban areas. Source: Pereira *et al.* (2004b).**

But Figure 2 shows that the area of pastures did increase. Even though the trend in the past years has been a decrease in the number of farms with pastures, the total area of pastures has doubled since 1989. Spontaneous herbaceous pastures are, therefore, key to understand the recent changes in Portuguese rural landscapes.

Figure 2 – Number of farms and total area of pastures per year between 1989 and 2007.



Source: INE (http://www.ine.pt, 2009).

To understand the whole chain of effects caused by the land use changes in Portugal, and also possible responses, there is one framework that can be used, and which we present next.

1.2.2 Main ecosystem services and impacts from pastures in Portugal

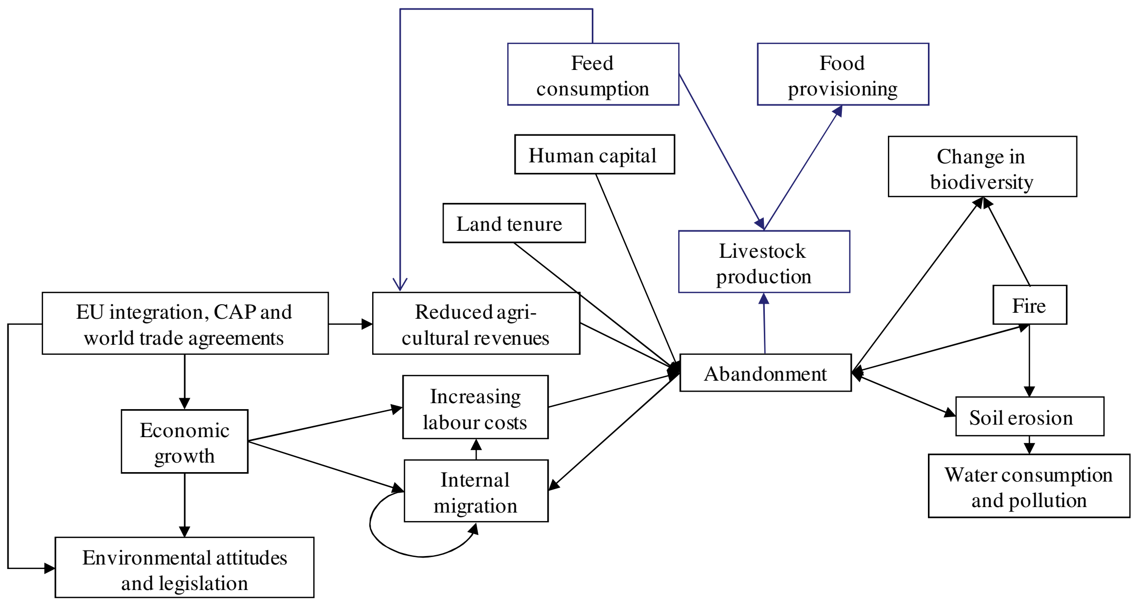
The Millennium ecosystem Assessment (MA) was launched by former United Nations (UN) Secretary-General Kofi Annan (MA, 2005). It aimed at a general worldwide assessment of the state of ecosystem services. Focusing on the consequences of ecosystem change for human well-being, the MA set the scientific basis for conservation, protection and valuation actions to be taken. Under the MA famework, several national, regional and local assessments were authorized. Portugal was one of the sub-global assessments. The Portuguese Millennium ecosystem Assessment (ptMA) had several case studies at three scales: (1) the national scale(whole country), (2) the basin scale (the Mondego and the Mira basins), and the local scale (Sistelo, Quinta da França, Ribeira Abaixo and Castro Verde).

At the national scale, the ptMA team chose the following services to be assessed (Pereira *et al.*, 2004b):

- Biodiversity;
- Provisioning services: water, food, and fiber;
- Regulating services: climate regulation, soil protection and runoff regulation;
- Cultural services: recreation.

The first step of the assessment was the determination of the main drivers of change in ecosystem services. The MA conceptual framework defines driver as any natural or human-induced factor that directly or indirectly causes a change in an ecosystem (MA, 2005). The ptMA determined the main drivers of change for Portugal through expert judgment, extensive literature review and workshops with the research team (Pereira *et al.*, 2004b). Since the increase in pastures has been a key change in land use in Portugal, we can adapt the ptMA findings to pastures, and include other specific environmental impacts, thus obtaining Figure 3.

Figure 3 – Main drivers of change in Portuguese Ecosystems regarding abandonment, according to the ptMA.



Adapted from: Pereira *et al.* (2004b). Original ptMA boxes are shown in black, and the boxes included by the present work are shown in blue.

A brief explanation of Figure 3 may be adapted from Pereira *et al.* (2004b)[6] as follows. The rise in the attractiveness of the industrial and service sectors in Portugal led to higher labor costs in agriculture (both in hired labor or opportunity costs for farmers). But the entry into the European Common Market and global trade agreements dropped agricultural prices, prices of which subsidies were hardly a sheer compensation. Agricultural firms then had to adapt. Some decided to abandon the agricultural activity, leaving fields unmanaged, or using them for extensive livestock production[7].

Many of these abandoned lands were used for extensive animal husbandry. But these "natural pastures" have significant problems. They can feed a small number of animals, which do not graze the totality of plants. Having resulted mainly from abandoned land, natural pastures are easily invaded by shrubs, which are primary stages of ecological succession. This process of succession may stop at very uninteresting degraded stages, and only active mangement can make them evolve further to climax. These shrubs have always had a tendency to appear, but were traditionally controlled by selective harvesting and use for animal bedding. Rising labor costs made this procedures, done by human hand, costly. Besides, less people managing such traditionally humanized landscapes as those in Portugal decreases discontinuities from compartmentalization, creating continuous areas of woody species left untouched and free to invade the pasture. This highly increases fire risk and severty (ISA, 2005).

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[6] For a full explanation, se Pereira *et al.* (2004b), page 22 and next.

[7] Options for farmers were the intensification of their land use, trading labor for mechanization and production factors, or to convert their production to more extensive (in the sense of less labor-demanding) land uses (such as afforestation, pastures or simple abandonment instead of crops). Intensification was chosen if land owners had enough financial resources, soil quality was high and water was available. Most farmers simply switched land use or purely abandoned the activity, as shown before in Figure 1.

For that reason, the necessity to control shrubs quickly became obvious. Even though animals control shrubs by stomping, degraded spontaneous pastures could only sustain low stocking rates, and therefore they are inefficient as a control method. Since it is too dangerous to use fire as a shrub control mechanism, mechanical action must be taken to shred the shrubs, decreasing fire risk and providing renewed conditions for nutritious plants to grow. But mechanical actions also have their flipside. They are usually intrusive and destructive for soil structure, leading to erosion phenomena and also carbon loss[8]. So, in the last 50 years, in an increasing fraction of Portuguese agricultural area, we had either high forest fire risk, which takes place without shrub control, or high soil erosion and carbon loss, which has been a natural consequence of damaging mechanical means to control shrubs. Fire and soil loss have dramatic consequences in terms of soil water holding capability and biodiversity promotion.

These pastures, however, provide the important provisioning service of food production (namely meat and meat products). But, even if livestock production is less labour-intensive and capital-intensive than alternative activities, it is also true that the degraded conditions of natural pastures do only support feeble stocking rates, with high inter and intra-annual variability in provision. Therefore, livestock will always require supplementation with forages or commercial feeds, which are expensive and further contribute to the decrease in agricultural revenue.

It is, then, clear that the use of abandoned land as pastures plays an important (negative) role in the (in)sustainability of the identified trend for land use changes by the ptMA. The previous framework clearly links environmental impacts with economic liabilities. It could be so that a solution for the promotion of ecosystem services also improves the economic viability of livestock production. In the past couple of decades, several strategies were devised and adopted to improve pasture performance. We now turn to those improvement strategies.

1.2.3 A response for the improvement of ecosystem services and impacts in Portuguese pastures

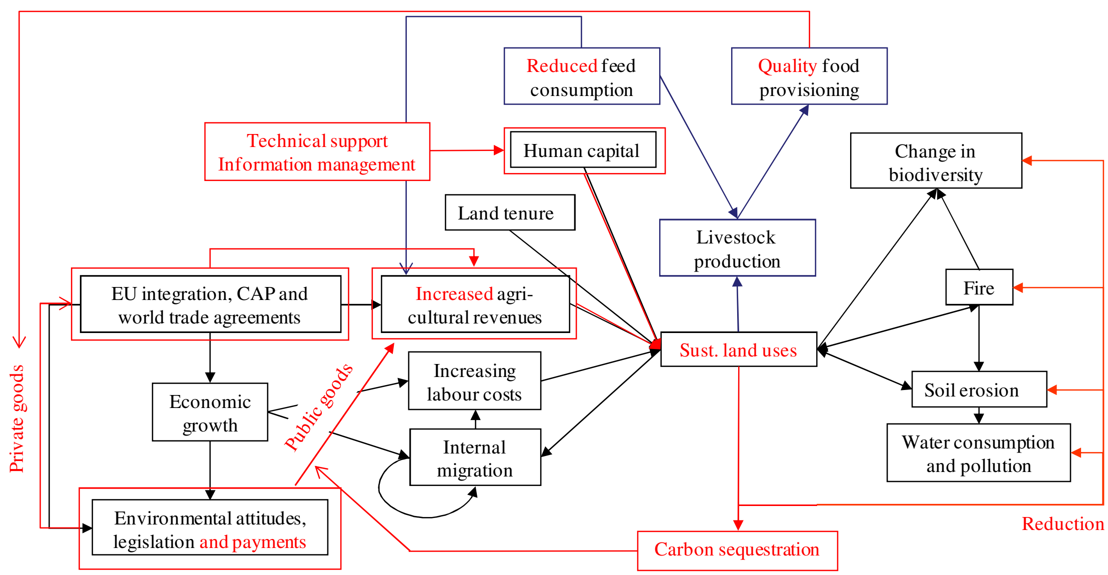
In the ptMA, some proposals for responses were formulated. Project Extensity was referred in the ptMA as one example of a coherent set of responses (Pereira *et al.*, 2004b). Project Extensity – Environmental and Sustainability Management Systems in Extensive Agriculture, was funded by the Life Program of the European Commission (LIFE03 ENV/P/505). The project took place between November 2003 and February 2008, reaching 86 farmers and over 70 000 ha spread throughout the country. In this farms, a sustainability management system was implemented. The system comprehended several levels, which went from the production of simplified Sustainability Reports to the implementation of the Eco-Management and Audit Scheme (EMAS) of the European Union. To obtain data, an LCA approach was privileged, and an information system was developed to collect it more efficiently. Once the system was implemented, various monitoring and analysis work projects were carried out, namely monitoring of biodiversity and the food chain. At the same time, consumers were scouted for their preferences through surveys. The reason why Extensity's responses are

[8] If the type and wight of the machinery is selected according to slope and soil type and condition, and if biomass is left and integrated into the soil, there are no evidences of such erosion. However, such is not traditionally the case in Portuguese agriculture, where tillage is still the most commonly used method of shrub control (Nuno Calado, personal communication).

perfectly adapted to pasture improvements is that the target farmers of the Project were those practicing extensive agriculture.

Responses from Project Extensity are shown in Figure 4, which may be read as follows. Technical support and knowledge for farmers increases human capital. This capital may lead them to search for sustainable land uses. Sustainable pastures must be more productive, in order to reduce feed consumption, and provide quality food (which is then also sold at a higher price). They must also reduce the environmental impacts of natural grasslands and, if possible, provide environmental services. One of those services should be carbon sequestration. Those land uses will only be sought if they increase farmers' revenue. This increase in revenue may be triggered by new opportunities from environmental regulations and payments for private goods and public goods. If carbon sequestration is one of those public goods, more chances of revenue exist, since carbon is the only environmental commodity for which there is a market currently established.

Figure 4 – Possible responses to improve ecosystem services in Portuguese Ecosystems regarding abandonment, according to the ptMA.



Adapted from: Pereira *et al.* (2004b). Original ptMA boxes are shown in black, and the boxes included by the present work are shown in blue. Responses are shown in red.

Figure 4 confirms that changing structural conditions, namely regarding the economic (increased revenue) and social (information and knowledge) aspects of rural zones, will positively influence the adoption of sustainable land uses. One important conclusion to be drawn from this analysis is that socio-economic and environmental sustainability go hand in hand in Portuguese agriculture. Rather than taking the economic performance and the environmental performance as substitutes, it is possible to design policies which are complementary in their goals. The rational use of public policies and private entrepreneurship may lead to practices which are environmentally positive.

This response, set by Project Extensity, is, then, an important course of action. But it remains to be determined whether there is a particular type of pasture that fits to this response, or weather it is only a theoretical possibility. The aim of this work will be to enquire if there is at least one type of pasture that provides the service of carbon sequestration will all other co-benefits refered here. In order to do so, we need to know first what kinds of pastures exist in Portugal today.

1.3 Which types of pasture are there in Portugal today?

1.3.1 Data sources

The main source of information for the characterization of Portuguese agriculture is the General Agricultural Survey done by the National Institute of Statistics (INE, the Portuguese acronym for *Instituto Nacional de Estatística*) every ten years. Since the last such Survey was done in 1999 (and was repeated in 2009, but results are yet unpublished), it is difficult to extrapolate the present situation of Portuguese agriculture from its findings.

However, a second important source of information is Project Extensity, which was directly aimed at farms with mostly extensive production systems, and pastures are, almost by definition, extensive or semi-extensive systems (as we mentioned in section 1.2.3). Therefore, the database collected from the Extensity farmers is both an update and a fine tune of the Survey done by INE.

1.3.2 Types of pasture

After inspecting data from both sources, it is possible to notice that pasture type varies with the following characteristics[9]:

- If the species are spontaneous or sown;
- If they are biodiverse[10] or not;
- If there is fertilization or not;
- If they are rainfed or irrigated;
- If they are under trees or in open areas.

However, some combinations of these options are not commonly found. To look at the representability of each type, the Extensity farmers were inquired as to what kind of pasture they have. Results are in Domingos *et al.* (2008), and can be summed up as shown in Table 1[11]. The same results may also be seen graphically in Figure 5. Table 1 shows that rainfed pastures clearly dominate. There are three significant groups of rainfed pastures[12]:

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[9] Some comments must also be done about the history of the field. However, those are now ignored for simplicity.

[10] The criterion for biodiversity is highly debatable. For rainfed pastures, 6 different species and varieties are considered to be the cut-off point for official uses (Rurual Development Programme, National Inventory Report). Still, some natural pastures comply with this requirement. The main originality of the SBPPRL system studied in this thesis is the fact that plant biodiversity is achieved by sowing.

[11] Note that in our analysis we only referred to permanent pastures. Temporary pastures are functionally similar to crops, since they require yearly sowing.

[12] In Project Extensity, two very large agricultural firms biased results significantly – Companhia das Lezírias and Grupo Sousa Cunhal. Together, they have more than half of the total area of pastures in Extensity. If we do not consider them, then sown biodiverse pastures are the majority.

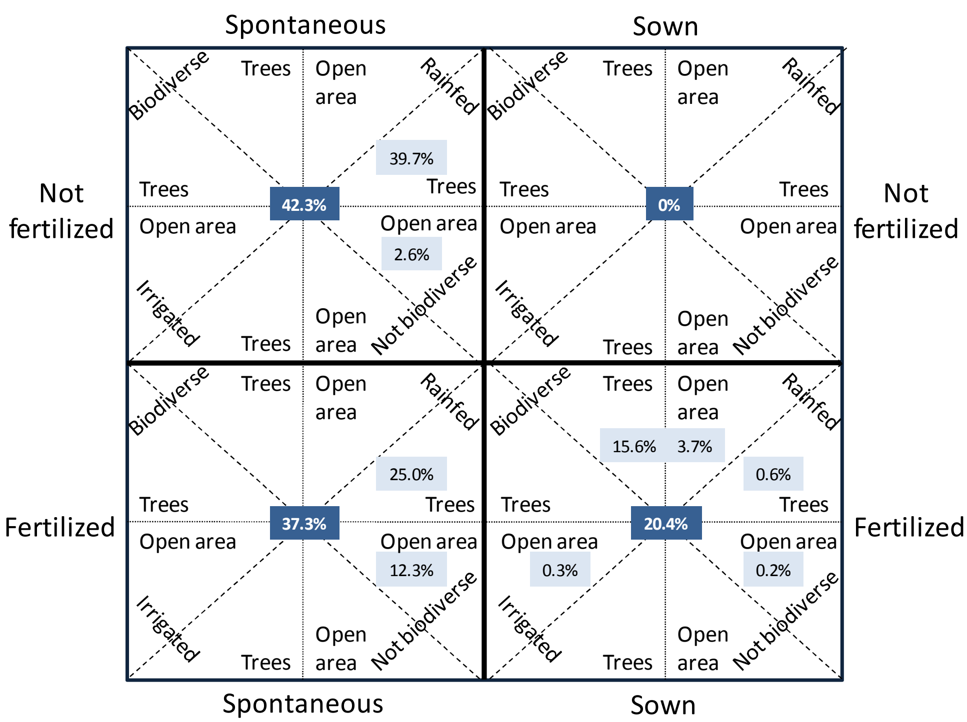
- Spontaneous unfertilized pastures;
- Spontaneous fertilized pastures;
- Sown biodiverse pastures.

Table 1 – Percentage of area for each type of pasture in the universe of farmers in Project Extensity.

Spontaneous or sown?	Fertilized?	Biodiverse[13]?	Rainfed or irrigated?	Under trees?	Area (%)
Spontaneous	Yes	No	Rainfed	Yes	25.0%
Spontaneous	Yes	No	Rainfed	No	12.3%
Spontaneous	No	No	Rainfed	Yes	39.7%
Spontaneous	No	No	Rainfed	No	2.6%
Sown	Yes	Yes	Rainfed	Yes	15.6%
Sown	Yes	Yes	Rainfed	No	3.7%
Sown	Yes	Yes	Irrigated	No	0.3%
Sown	Yes	No	Rainfed	Yes	0.6%
Sown	Yes	No	Rainfed	No	0.2%
All other options					0.0%
Total					*** 100.0%**

*** Total area of pastures was 17.950 ha.**

Figure 5 – Schematic representation of the different options for pasture types, and respective percentage of area of farmers in Project Extensity.



[13] In the case of Project Extensity, the arbitrary cutoff point for biodiversity was if more or less than 6 different species were sown (if it is a sown pasture) and are present today (for any type).

If Project Extensity has a representative sample of the fraction of Portuguese farmers with pastures, then we could assume that these three systems are also the most important in the rest of the country. But is this a fair assumption?

Considering results from the INE's Survey, there are two major differences. The first difference is that in 1999 only 55% of pastures in the country were under tree cover. For Project Extensity's areas, the figure rises to about 80%. This is a consequence of the fact that the cork and holm *Montado* zones are over represented in Extensity. *Montado* is an agro-forest system where usually pastures and oak trees coexist[14].

The second difference is that about 20% of the Extensity areas are sown biodiverse pastures. Moreover, they correspond to one specific system, the sown biodiverse permanent pastures rich in legumes (SBPPRL), which is the subject of this work and will be thoroughly analysed in chapters to come. However, this system is not present in 20% of the country's pasture land. Data from INE shows that there are about 1.8 million hectares of pastures in Portugal (Figure 2 above). But if we look at the number of hectares of these pastures which were purchased to Fertiprado (shown in Section 1.4.4), the main firm in the pasture seed business in Portugal, we see that only about 90 000 ha were sown, or about 7%. Therefore, the fraction of the area of sown biodiverse pastures is overestimated in the Extensity sample[15]. This is not a disadvantage, however, since, as will become evident in the next section, we will particularly focus on this type of grassland system from this Chapter on.

Regarding other systems, naturally (not sown) biodiverse pastures are underestimated among Extensity's farmers. This is because those pastures are mainly present in the Northern regions of Portugal. Those areas are also outside of the geographic scope of the present thesis, which is Central and South Portugal (since it is the region where SBPPRL are mostly present, and also the regions for which we have data). Sown pastures which are not biodiverse are not significant in area, since they correspond to failed installations of sown biodiverse pastures or to common mixes of one grass and one legume.

Therefore, we conclude from this analysis that it is possible to define three types of pastures which are the most representative of Portuguese pasture areas, all of which are rainfed. For purposes of linguistic and conceptual simplicity, we now define each of them as:

- Natural pasture (or grassland[16]) – a natural pasture is a spontaneous unfertilized rainfed pasture, which may or may not be under tree cover;
- Fertilized natural pasture (or grassland) – a fertilized natural pasture is a spontaneous fertilized rainfed pasture, which may or may not be under tree cover;

[14] According to Pereira *et al.* (2004), "the *Montado* system is an evergreen oak woodland, where the predominant tree species are the cork-oak (*Quercus suber* L.) and holm-oak (*Quercus rotundifolia* Lam.). Montado is an agroforestry system, where the main activities are cork, livestock and cereal crop production. Montado is an economically and ecologically important system and is characteristic of Portugal and Spain (where it is called *dehesa*)".

[15] Due to the previous footnote, if we do not consider the bias from the two largest firms in Extensity, this difference would be even larger.

[16] Recall that we are using the terms "grassland" and "pasture" as synonyms.

- SBPPRL – a SBPPRL is a specific case of a sown fertilized rainfed pasture, consisting on a high number of plant species, amongst which there are many legumes.

We must notice that, when comparing these three systems, we are comparing three degrees of intensification. Natural pastures do not require any particular input or maintenance operation, apart from periodic shrub control operations. Their fertilization is an intermediate grade, and SBPPRL are the most "intensive" system, since they require seeds and other inputs, as we will discuss later on[17].

Keep in mind that we stated before that pastures are important, both as a driver (in the case of natural pastures, which are the reference situation in the largest fraction of the area) and as a response (in the case of optimization options such as fertilization or sowing), to the conditions and trends affecting ecosystem services. While natural pastures are straightforwardly defined, since they consist on any area with spontaneous herbaceous plants which is used for grazing, SBPPRL are an innovation which requires a clear definition.

1.4 Sown biodiverse pastures (SBPPRL)

1.4.1 Why biodiversity?

The generalized positive effects of plant biodiversity are now well established in the literature. Biodiversity in grassland composition increases plant productivity and the stability of nutrient retention in soils (Schläpfer and Schmid, 1999), by promoting the storage of both carbon and nitrogen in soils (de Deyn *et al.*, 2009).

Some authors, such as White *et al.* (2004), have argued that high species diversity in all plots of a given managed area is a positive factor in grassland areas. However, other authors, such as Duru *et al.* (2005) have argued that, even though the role of local biodiversity is unquestionable, it is also possible to maintain in the same managed area a mosaic of parcels, each one with low functional diversity, but highly diverse as a whole. Grazing intensities should also vary throughout the managed area, in order to allow different types of herbaceous plants to grow (McIntyre *et al.*, 2003).

One thing is for sure: biodiversity, either local or across a given range, is positive both to plant productivity and to ecosystem functioning (Schläpfer and Schmid, 1999; Spehn *et al.*, 2005). Sowing many species is important in the Portuguese heterogeneous soils. But it is functional diversity, more than number of species, that has the most effect in ecosystem processes (Díaz and Cabido, 2001; Wardle *et al.*, 2000). It is because biodiverse pastures have a high percentage of nitrogen-fixing legumes that they work like a well-oiled machine – legumes capture nitrogen, grasses use it to grow, sequestering carbon from the atmosphere.

1.4.2 Development of the SBPPRL system

As we've mentioned before, there are several types of pasture. Today, the sum of improved and spontaneous pastures makes up for about 70% of the world agricultural area (Suttie *et al.*, 2005). In Portugal, there are now around 1.8 million hectares of

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[17] Since SBPPRL are more "intensive" and also more productive, there is some area which is vacated for the same national animal stocking rate.

grasslands (INE, http://www.ine.pt, 2010), divided mainly between the three above mentioned grassland systems: Natural Grasslands (NG), Fertilized Natural Grasslands (FNG), and Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL). These three types of pasture correspond to three different degrees of intensification.

Data shown before supports the assessment that NG is by far the most used grassland system in Portugal. These pastures are relatively poor in terms of feedstock and have several associated environmental impacts. They consist of either fallow stages from long cereal rotations, or spontaneous vegetation in previous croplands which have since been converted to areas for livestock feed. NG typically have no specific management, except for occasional operations to control shrub growth. The most widely used operation is tillage.

The only difference between NG and FNG is that the latter are fertilized. The species and varieties of spontaneous grasses and legumes are the same, but fertilization increases productivity. Therefore, advocates of FNG claim that fertilization is a compromise between productivity and natural values. Therefore, in FNG, frequent shrub control methods are also required. Since the most significant area of pastures in Portugal is located in cork and holm oak agro-forestry systems (*Montado*), all types of natural pasture give room for some natural regeneration of trees. Natural regeneration is the result of inefficient control of shrubs, which are primary stages of the succession mechanism of *montado*. Furthermore, advocates of FNG claim that methods of shrub control other than tillage will benefit the soil nutrient recycling system. Shrubs have deeper roots than grasses and legumes, and therefore access nutrients in deeper layers of soil to grow. Control operations will then shred their aboveground biomass. This biomass remains on the ground and is incorporated in the first layer of soil, which is then used by pasture plants to grow.

Starting from the reference situation of natural (degraded) pastures, some other farmers and agricultural scientists believe that fertilization alone does not provide the best results in terms of plant productivity and animal feed quality. Advocates of sown pastures believe that the introduction of specific species or varieties, either absent or in lesser percentage in spontaneous grasslands (as, for example, some varieties of legumes) will establish a functioning ecosystem with complementary ecological niches and improve production.

This line of thought led to the development in Portugal in the 70s of an alternative, using the concept of "biodiversity engineering" as a defence against the very diverse Mediterranean natural conditions, which are SBPPRL. They have been installed throughout some regions of the country in the past decade, as well as in some minor areas in Spain and Italy.

The SBPPRL system consist of diverse mixes of up to twenty different species or varieties of seeds, and are rich in legumes. Commonly SBPPRL are more productive than natural grasslands, and are also richer in number of species (Carneiro *et al.*, 2005). There are less gaps in plant cover throughout the plots, since species variability ensures that the species most suited for each spatial conditions will thrive. Even though there is a well documented experience with the use of sown pastures (FAO/CIHEAM, 2008), this specific system only exists in Portugal and, to a lesser degree, Spain and Italy. There are many studies on the role of biodiversity in productivity, but SBPPRL remain the only widespread large-scale application of what may be called "biodiversity engineering".

Since SBPPRL are a relatively new system which, to our knowledge, has not yet been fully studied, we decided to dedicate this chapter to its description. The next sections will be used to explain their environmental and economic effects.

1.4.3 Qualitative description of SBPPRL

SBPPRL have higher productivity than natural grasslands, and are also richer in number of species. Their high productivity is due to the fact that biodiversity allows the most adapted plants to prosper in each zone. The seed mix is designed specifically for each location after soil sampling and analysis. Species in the mix are adapted to soil physical and chemical characteristics, as well as to local climate conditions, and therefore there is no single representative mix. However, some very common sown species in SBPPRL mixes are *Trifolium subterraneum, Trifolium michelianum, Ornithopus spp., Biserrula pelecinus,* annual *Medicago spp.,* and grass species of the genera *Lolium, Dactylis and Phalaris*. The mixes of sown species are often enriched with seeds from spontaneous plants such as *Plantago spp*., *Vulpia spp.* and *Bromus spp* (Carneiro *et al.*, 2005; David Crespo, personal communication). Legumes are inoculated with bacteria of the genus *Rhizobium*. These bacteria induce nitrogen-fixing nodules in the roots of legumes. The fixated atmospheric nitrogen is then used by grasses. Therefore, the overall system is self-sufficient in terms of nitrogen. Legumes cover more than 50% of first-year SBPPRL (Carneiro *et al.*, 2005). As pasture settlement progresses, legumes increase and eventually dominate. Percentage of legumes in the plant cover of a mature SBPPRL (more than 5 years) is around 25 to 30%.

Increased productivity in SBPPRL allows a sustainable increase in animal carrying capacity. Animals graze the plants, which have an annual life cycle. High plant productivity implies increased atmospheric carbon capture through photosynthesis. Part of the biomass produced is stored in soils due to the high density of yearly-renewed roots. Storage is in the form of non-labile Soil Organic Carbon (SOC), which is part of the Soil Organic Matter (SOM) pools. SOM pools are also increased by leaves' senescence and decomposition, and by animals returning undigested fibre to the soil.

Increasing SOM improves soil nutrient availability and water holding capacity, thus increasing plant productivity[18] and reducing surface runoff of water, which in turn decreases sediment loss and soil erosion (EEA, 2004). Decreasing water runoff and soil erosion have positive effects even outside the plot. Sediments, nutrients, organic matter and pesticides carried in water contribute to silting, eutrophication and contamination of surface waters. These effects are known, but their true costs are still hard to estimate[19]. Nitrogen fixation by legumes eliminates the need for nitrogen fertilizers, whose production is highly energy demanding, and therefore responsible for high greenhouse gas emissions. Finally, both increased stocking rate and reduced fertilizer use increase the economic viability of the farms. This is particularly important because socio-political and economic conditions are barriers to the successful implementation and

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[18] Water holding capacity allows the establishment of a soil water reserve which is then used throughout the months with higher water deficit. This effect attenuates inbalances in productivity during the year..

[19] Note that at the regions where SBPPRL are mostly sown (South and Central Portugal), surface runnof only takes place at extreme rainfall events and in high slopes. Therefore, the positive environmental effects of decreasing runnof seldom take place strictly due to SOM. It must also be noticed, however, that the effect of increasing water holding capacity always takes place in soils with high SOM. This effect is agronomically very important, as it increases production.

management of pasture systems which provide environmental services (Neely *et al.*, 2009).

It should be noticed that grasslands and agricultural soils do not store carbon indefinitely. As SOM content increases, so does the organic matter mineralization rate, while the soil's storage rate decreases. As storage and mineralization rate become equal, eventually a steady-state is reached. In that steady-state, there is no carbon sequestration, but the system maintains all the other advantages. Therefore, SBPPRL could be relevant for carbon sequestration in the short term, but it's other environmental and economic benefits that can prove their value in the long term.

Increasing SOM, nutrient availability and water in soils provides both mitigation and adaptation to climate change. SOM accumulation through an increase of SOC is the mechanism through which carbon is sequestered in grassland soils. This is particularly important for Portugal, since, as we refered before, it was one of the few countries to elect the "Grassland Management" voluntary activity, in the framework of the Land Use, Land Use Change and Forestry (LULUCF) activities, now named Agriculture, Forestry and Other Land Uses (AFOLU), under Article 3.4 of the Kyoto Protocol. This choice was made mainly because of the implementation of the SBPPRL system in Portugal. However, there is currently no study published on the potential of the SBPPRL system to increase SOM. According to IPCC guidelines, Tier 1 approaches imply that one sequestration factor is attributed, regardless of related field data. The present work should be a first step: we study how much, on average, SBPPRL increase SOM, in relation to the baseline, which are natural grasslands.

1.4.4 Implementation of SBPPRL in Portugal

Fertiprado started selling those mixes of SBPPRL seeds in 1996, and have since then become the most representative seller in Portugal. Therefore, the data series for Fertiprado sales provides an approximate depiction of the evolution of the installation of SBPPRL in Portugal. Therefore, they can also be used to extrapolate future installations of pastures.

In order to do so, we observed that sales have followed an approximately logistic increase. After an approximately constant increase during the first years, there was an accelerated increase in sales after the year 2000. That increase has been lowering from 2005 on. To model this logistic curve, we chose a Verhulst-Pearl function for population growth. This is a non-linear function using two parameters, r and K, the first one being the growth rate in the initial stage (yearly installation) and the last one the carrying capacity, corresponding to the asymptotic maximum (maximum area of pastures installed). In the case, population growth is used as an analogy for area growth due to spreading of the word between farmers, in the absence of any extra support, as for example the Terraprima-Portuguese Carbon Fund (PCF) Project.

The logictic equation we used is:

$$\frac{dP}{dt} = rP\left(1 - \frac{P}{K}\right), \tag{1.1}$$

where P is the area of SBPPRL installed in the year t. This differential equation has an exact solution, which is:

$$P(t) = \frac{K \cdot P_0}{\left(K - P_0\right) e^{-rt} + P_0}, \quad (1.2)$$

where P_0 is the SBPPRL area installed in the first year[20] (1996), and *t* is the number of years. This function is not linear in *t*, and therefore must be calibrated using non-linear estimation. We used actual Fertiprado sales for the calibration, and applied the numerical algorithm in software SPSS 16 to perform successive iterations on the parameters. The final values established for the parameters are those that minimize the mean quadratic error in the estimation of the input data (Fertiprado sales). Calculations were made for the whole country, and also for each of the three most representative regions of SBPPRL installation: Beira Interior, Ribatejo e Oeste, and Alentejo.

Parameters obtained, as shown in Table 2, indicate that the maximum area obtainable (parameter *K*) in the sum of the three regions would be 109 535 ha. Around 80% of this area is in the Alentejo region.

Table 2 – Estimated parameters for the logistic model of SBPPRL area installed in Portugal.

Estimated parameters	Alentejo	Ribatejo e Oeste	Beira Interior	Total
r (yr^{-1})	0.447	0.354	0.372	0.429
K (ha)	86 739	11 091	12 415	109 535
P_0 (ha)	1,783	359	325	2,443

r, K, P_0 – parameters in the logistic function.

Table 3 shows observerd areas and areas calculated using the calibrated logistic model for the total of the three regions.

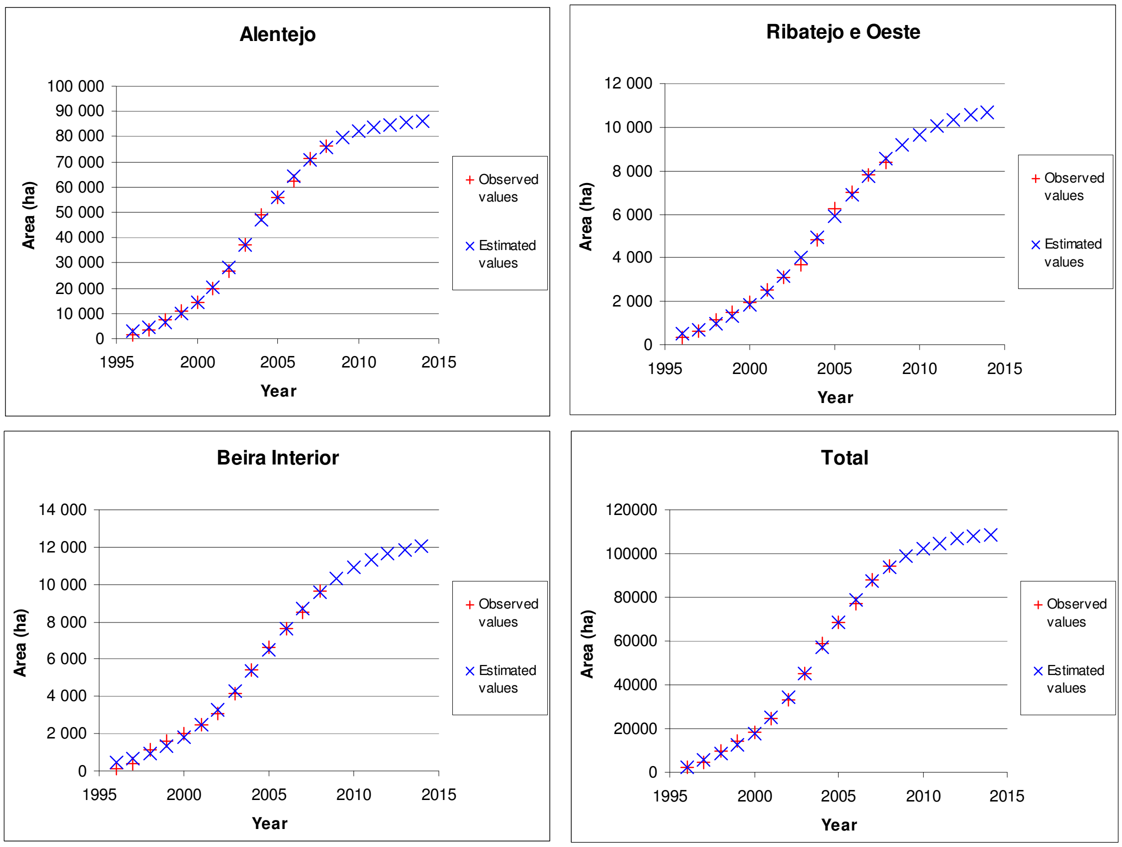
Table 3 – Cumulative area of SBPPRL (observed and calculated using the logistic model), according to sales from Fertiprado.

Year	Observed area (ha)	Calculated area (ha)
1996	2 071	2 443
1997	4 459	5 593
1998	9 439	8 361
1999	14 042	12 336
2000	18 505	17 866
2001	24 564	25 233
2002	32 907	34 494
2003	45 081	45 326
2004	59 120	56 977
2005	68 786	68 431
2006	77 045	78 738
2007	87 751	87 301
2008	*94 260	93 954
2009	-	98 859
2010	-	102 339
2011	-	104 739

[20] Note that, for estimation purposes, P_0 is a parameter and not an input, in order to avoid anchoring the model to the initial point.

Figure 6 represents values in Table 3, and also for each of the regions. Visual inspection clearly shows that the logistic pattern is a good fit, particularly for the sum of the three areas. It is important to stress that observed values can only be considered until 2008, since 2009 marked the beginning of the Terraprima-PCF Project (we will explain what we mean by this Project in due time, namely in Chapter 4).

Figure 6 – Observed and estimated area using a logistic model, including forescast.



Upper right corner: Alentejo region; lower right corner: Beira Interior region; upper left corner: Ribatejo e Oeste region; lower left corner: total.

1.4.5 Interpretation of the reasons for the pace of SBPPRL implementation

If SBPPRL are indeed a good option for farmers, with many economic and environmental advantages, then it is somewhat puzzling at first sight that the rate of their implementation has been decreasing.

There are many possible explanations for the fact that some farmers stop installing pastures and other farmers never install them at all. Many of them have to do with social reasons, namely cultural barriers. There seems to be an intuitive idea amongst farmers that annual pastures are more productive than permanent pastures, even though annual pastures are more expensive and do not show better results. This idea is persuasive because farmers are economic agents sensitive to social representations of their activity, and believe that showing soil management implies actively performing actions such as sowing every year (João Paulo Crespo, personal communication). One dramatic example of this are most parts of the Trás-os-Montes region, where farmers are socially well-regarded as good soil managers if they keep grasslands completely safe from infestants all year round. They achieve this by recurring to very frequent tillage, sometimes as often as two or three times a year (António Martelo, personal communication). Due to the slope of most plots in the region, this accelerates soil loss and, in the medium run, greatly decreases productivity. Keeping a permanent pasture also implies good management, but in a less visible way, and thus the social perception of the activity of the farmer is harder.

We must also note that SBPPRL is a relatively new and innovative system, which implies some knowledge transfer not only to break social barriers, but also to guarantee that farmers manage the pastures correctly. Financially, the agricultural sector faces risks that the farmer does not control (price fluctuation of inputs and of products sold, climate, among others), even for activities that the farmer dominates. Uncertainty is even higher for innovations, and so farmers are more risk adverse.

For example, the year 2008 was exceptional, in that the price of almost all production factors (fuels and fertilizers, mainly) raised significantly (Pedro Silveira, personal communication), while the sale price for livestock, meat and meat products seeply decreased. Even though it could be argued that SBPPRL can be a defense against such uncertainty (since they can be more productive, and better soils are more reliable against uncertain climate conditions, they are always better than the alternative of natural pastures), there is an important block to their installation. SBPPRL require a significant initial investment for the installation, which must be made in the first year, and 25% of which can be supported by public funds at most. The first year is a particular year in management as well, since grazing must be limited in order to allow plants to complete their cycle undisturbed, and establish a good seed bank. This seed bank ensures the renewal of the pasture each year, and therefore is crucial for its maintenance. Some farmers cannot support this first-year loss.

Besides, one must consider that there is a whole range of farm sizes and natural conditions. For some farmers, the expected revenue is higher (farms with naturally good initial soil and climate conditions), and the costs are lower (for example, for very large areas some fixed costs are diluted). These farmers, who had the largest benefit from sowing SBPPRL, were the first ones to install them, and they installed them where it was less expensive to do so. We may depict the most likely sequence of adoption of the system as follows:

- First, only well informed farmers with large areas and capital to invest installed SBPPRL. First-movers installed them in places where the costs with machinery are lower, namely in valleys and other clear areas.
- Then, after all clear areas were filled, farmers persuaded by the results obtained with this system started sowing pastures in agro-forestry areas within their farms. It is unknown in what fraction of the farm farmers stop sowing pastures, but since they sow them essentially for livestock feed, it is plausible to assume that they sow as many pastures as needed to end concentrated feeds consumption. We will explore this assumption in further detain in Chapter 3.
- Other farmers became aware of the existence of the system and of its results, and the area expanded at a high rate afterwards.
- As all farmers who expect to receive a positive income from sowing SBPPRL install pastures, the rate of installation starts decreasing, up until the point when

almost all other farmers with grasslands can expect a negative revenue from SBPPRL.

If this sequence of events is correct, than the existence of a specific support such as, for example, payments from the PCF would make the system worthwile for farmers who, otherwise, would not have a positive income from SBPPRL. Those would be the farmers that would then be interested to install pastures, and they would be the implementation scenario for such a project. But to know if the PCF could support such a system, we must first establish why SBPPRL promote cabon sequestration.

1.5 Comparison of natural pastures and SBPPRL

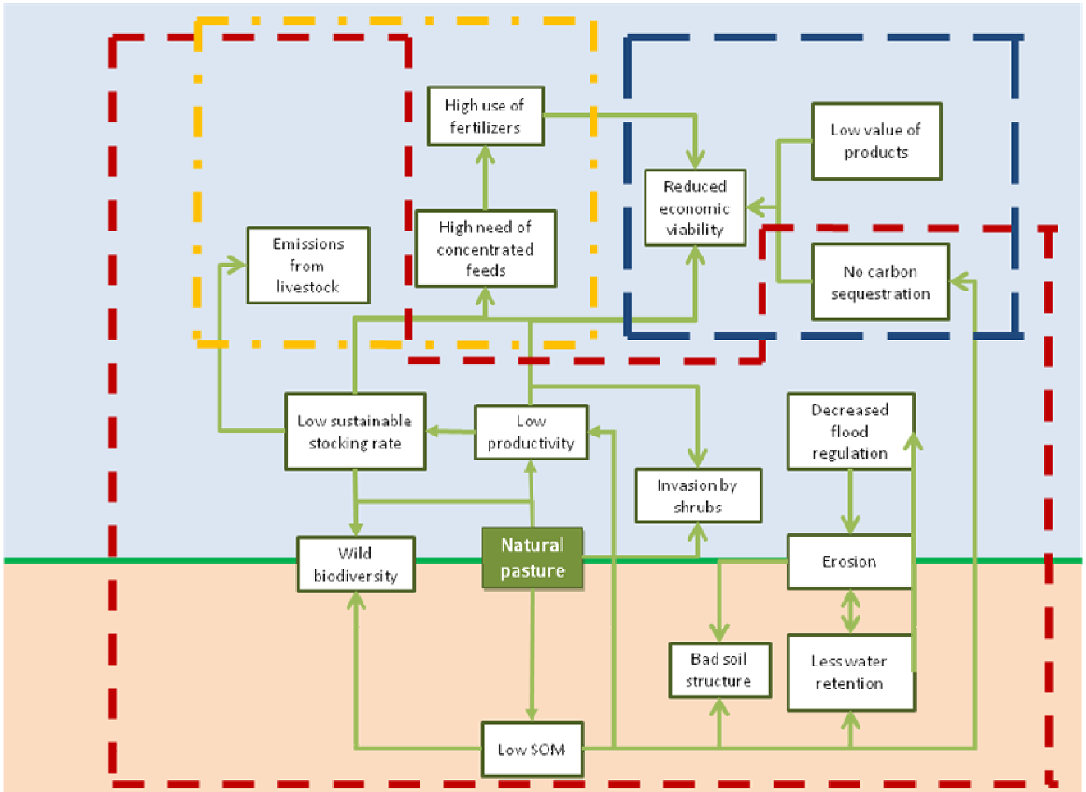
In this Section we will qualitatively compare natural pastures and SBPPRL in terms of the three sustainability sub-systems we defined. We will not mention fertilized natural pastures, since they are an intermediate situation between those two.

The basic scheme of socio-economic and environmental effects in livestock production in natural pastures is shown in Figure 7. Figure 7 should be read as follows.

1. Natural pastures are less productive[21], both above and belowground. Low belowground productivity is translated by low SOM.
2. Low SOM and low soil cover by pasture plants imply a bad soil structure, leading to more erosion, less water retention and consequent decreased flood regulation. More superficial runoff leads to increase erosion as well.
3. Since natural pastures are less productive, they feed fewer animals, and so are typically exploited with low sustainable stocking rates. Low stocking rates mean that animals do not control shrubs, which increase in the field. Besides, low productivity means that shrubs increase, since they find small competition from herbaceous plants (grasses and legumes).
4. Livestock in pastures emits methane, which is a GHG.
5. Since pastures are not productive enough, the only way to balance the animals' diet is to recur to concentrated feeds. These feeds are composed mainly of cereals and oilseeds, which required fertilizers to be produced.
6. Lower SOM has a negative effect on soil fauna, but low stocking rates and more woody vegetation may have a positive effect on biodiversity.
7. Constantly low SOM means that no carbon sequestration occurs.

[21] Note that all qualifiers (e.g. low, high, more, less) are attributed to each pasture type (natural, SBPPRL) in relation to the other, except when explicitly referred.

Figure 7 – Causal scheme of effects of livestock production in natural pastures.



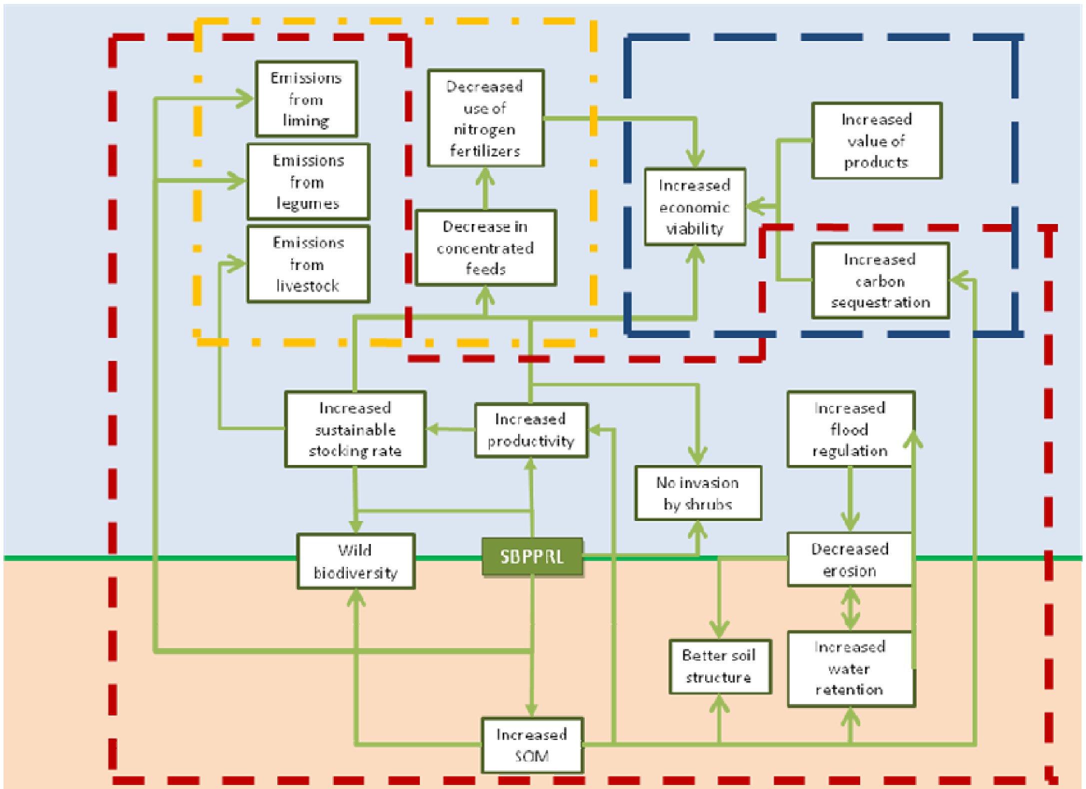
Red dashed line – Direct environmental effects. Yellow dashed line – Indirect environmental effects. Blue dashed line – Socio-economic effects.

Regarding SBPPRL, the scheme in Figure 7 becomes similar to the one in Figure 8. Again, we can make nine statements similar, one by one, to ones made for natural pastures. The analysis of each link at a time is as follows.

1. SBPPRL are more productive, both above and belowground. High belowground productivity is translated by high SOM.
2. High SOM and increased soil cover by pasture plants imply an improved soil structure, leading to less erosion, more water retention and consequent increased flood regulation. Less superficial runoff leads to a decrease in erosion as well.
3. Since SBPPRL are more productive, they feed more animals, and so are typically exploited with high sustainable stocking rates. High stocking rates mean that animals control shrubs either by stomping or by using them in their feed, since shrubs are rich in fiber, to complement for excess protein from the consumption of legumes. Besides, higher productivity of grasses and legumes leaves fewer resources available for shrubs.
4. Increased stocking rates mean that livestock in pastures emits more methane, which is a GHG. There are also emissions due to biological fixation of nitrogen by legumes, and emissions from liming, a process required to increase the pH of soils (if they are too acid).
5. Since SBPPRL are more productive enough, there is less need to recur to concentrated feeds. These feeds are composed mainly of cereals and oilseeds, which required fertilizers to be produced, and which are then avoided.

6. Higher SOM has a positive effect on soil fauna, but higher stocking rates and less woody vegetation have a negative effect on biodiversity.
7. High SOM increases mean that carbon sequestration occurs.

Figure 8 – Causal scheme of effects of livestock production in SBPPRL.



SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes. Red dashed line – Direct environmental effects. Yellow dashed line – Indirect environmental effects. Blue dashed line – Socio-economic effects.

Differences between the two methods are summarized in Table 4.

Table 4 - Differences between baseline and proposed scenarios.

Baseline Scenario	Proposed Scenario
Degraded natural grasslands / former cropland areas:	**Sown biodiverse permanent pastures rich in legumes:**
• Net carbon emissions (from animals)	• Carbon sequestration by agricultural soils and improved soil fertility
• Low stocking rate	• Increased stocking rate
• Shrub invasion and fire	• Shrub control and reduced fire risk
• Low inputs and machinery	• Increase in production factors' consumption
• Synthetic nitrogen fertilization	• Nitrogen fixation by legumes
• Erosion and low water cycle regulation	• Benefits in soil and water cycle regulation

This *a priori* descriptive comparison constitutes the basic hypothesis we tested in this thesis. In the next Chapters, we will justify each of the seven statements made before.

When speaking of pastures, we must always keep in mind their final goal, which is to feed livestock, and, as evidenced in Figure 7 and Figure 8, animals are a crucial part of both the system's environmental and economic effects. So, the final part of the present Introduction regards the definition of scenarios for livestock production (type of animal and life span in pastures).

1.6 Pastures and meat production

Since the beginning of the domestication of livestock by humans, fields of herbaceous plants have been used to feed animals. According to Diamond (1999), one of the important reasons for any animal to be domesticated is to have a flexible diet, particularly if it consists on food such as grasses and forages, which is not consumed by humans. This makes them less expensive to be kept in captivity than, for instance, the domestication of carnivores, which would mean that other animals would have to be bred and fed to them.

Therefore, herbivores have been the natural choice for animals domesticated for food. They are, together with fish, the main source of animal protein in human diet. Grasses and most legumes are traditionally the choice to feed livestock, since they are not used in human diets.

Traditionally, these animals were, then, fed in pastures and from excess crops. Pastures could not support large stocking rates, and meat production was, a consequence, low. In many countries, including Portugal in the beginning of the 20th Century, meat was a luxury in many households.

With the growth of the concentrated feed industry, animals begun being placed in small areas called stables where they are fed only feeds (and, to a smaller degree, straw or forages. This new option for animal production is considered intensive production.

1.6.1 Intensive vs. extensive production

1.6.1.1. An issue of intensity

The intensity of animal or agricultural production is an issue connected with the relative *intensity* of production at a given place (tons per hectare for plant production, and heads per hectare for animal production). Thinking in terms of animal production only, when comparing both systems we are comparing how much area we need to produce each animal.

This concept of area, however, has both direct and indirect consequences. Direct consequences deal with the fact that in extensive production a relatively low number of animals graze relatively extensive areas. Meanwhile, in intensive production systems, a relatively high number of animals are stabled indoors. Therefore, switching from extensive to intensive production, there are areas that become vacant and may be occupied by other land uses. Whether the environmental balance of this substitution is positive or negative depends on the specific alternative land use.

However, significant indirect effects also arise, which deal with the use of commercial feeds. The relevance of commercial feeds derives from the fact that in both intensive (always) and extensive (during the least productive seasons) production systems of meat it is necessary to provide feeding to the animals elsewhere than the pasture.

Since the 1950's, consumption of meat products has increased steadily. It is considered that 1 kg of beef requires 7 kg of high-protein feedstuffs (Brown *et al.,* 1999), 1 kg of pork requires 4 kg of grain (CIWF, 1999) and 1kg of poultry requires 2 kg of feed (CIWF, 1999). Therefore, a higher meat production corresponds to a higher ingredient demand. Today, 95% of the world's soybean production and a third of commercial fish catches are used for animal rather than human feed (Millstone and Lang, 2003). The area needed to produce feeds for the increasing number of animals is, then, high; 75% of all agricultural land in the United States is used to produce ingredients in livestock feed (Millstone and Lang, 2003). This contributes significantly to the impacts of crop production, aggravated by the impact of the animals themselves, namely regarding soil loss and desertification – 85% of all topsoil loss in the United States is attributable to livestock ranching (Millstone and Lang, 2003). Therefore, an increasing number of studies have tried to assess sustainability in agriculture (Lewandowski *et al.*, 1999), particularly researching the environmental impact of feeds (Cederberg and Mattsson, 2000; van der Werf *et al.,* 2005).They invariably conclude that concentrated feed production and transportation is the main souce of environmental impact in the meat production chain.

In Portugal, the intensification trend was also followed. The animal feed industry is the third most important in the agricultural sector, representing 10.5% of total business volume in 2002 (IACA, 2004). The total production of feeds in the same year has been estimated in almost 3.5 million tons (IACA, 2004).

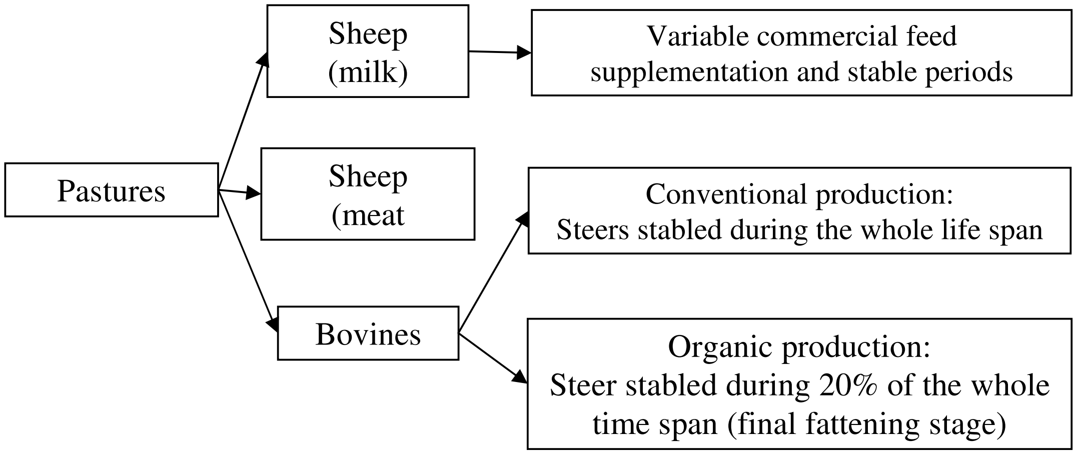
However, there is not always a competition between intensive and extensive production. Given the current structure of Portuguese livestock production, in some cases the comparison does not make sense.

1.6.1.2. Which animals go were?

To understand in which cases the intensive and extensive approaches are rivals, again we turn to results from the survey to Project Extensity's farmers in Domingos *et al.* (2008). There are three types of animals which are produced resorting to pastures (Figure 9). Sheep fed for meat production are always kept in pastures, whereas sheep for milk production are subjected to a mixed system consisting of variable time in pastures and stables, depending on the stage of lactation. Bovines, usually used for meat productions (amongst Extensity farmers), are either stabled during their whole time span[22] (purely intensive production), or only during the final period of fattening (extensive production). Note that, while stabled, animals are fed with concentrated feeds and straw or forages. We will come back to this issue when we analyse animal feeds.

[22] Time span, in this context, is defined as the time between weaning and slaying.

Figure 9 – Different types of animals and production methods, according to the survey to Extensity farmers.



Livestock production gives origin to different products (e.g. meat, milk, leather). Since each one has a particular economic value, we tried to restrict our analysis to one, which was meat. This choice is linked to another one, which was the type of animal studied. Each animal gives origin to several meat types, all of which are important in a balanced diet. However, it would be impossible to make a thorough study of all of them, given the specific economic data involved in each one, and also the characterization effort required to fully depict the whole system (e.g. feeds used). Therefore, in our work, we had to keep to just one type of animal. We chose bovines, namely those involved in the meat production system – breeding cows and steers. But even making this choice, specific scenarios for fattening and weaning must be defined. This is the subject of the next section.

1.7 Definition of scenarios

So from everything we stated before, we can now define the scenarios we chose to study. We will study three types of grassland, from the less extensive one (natural pastures) to the most intensive one (SBPPRL). We also determined that the product we will focus on will be bovine meat.

We justify our choice of bovine meat as the main indicator because it is currently the main source of income to most animal producers in the country, at least between those with SBPPRL (Pedro Silveira, personal communication). This will be clearer when we characterize the economic costs of pastures and describe the revenue and public supports available. Furthermore, bovine meat is still one of the most consumed meat types by Portuguese households. And, last but not least, bovine meat is the product for which more information is available.

Since we chose bovine cattle, we must decide on which scenarios to use for animal feed and weaning. Those scenarios were devised from the experience of project Extensity.

In extensive production, cows typically spend their entire life cycle grazing in pastures, and may or may not be supplemented with concentrated feeds or forages. Each cow has one steer per year, which may be weaned immediately after birth or around 6 months after. When weaned at birth, steers follow a completely intensive production life cycle, since they are stabled and fed entirely on concentrated feeds and forage. When weaned at six months of age, they may be kept in pastures or they may be partially or totally stabled, and in each of the cases they may or may not be fed with concentrated feeds. Steers are then slayed at 12 to 24 months of age. Which percentage follows each path is unknown. However, most steers are transferred to intensive production after weaning. This is because steers that stay in pastures for more than 6 months give origin to meat which is darker in color, and therefore is disregarded by consumers. We will return to this is issue in the last Chapter.

In this thesis, we consistently used a scenario consisting on the following characteristics:

- Functional unit: one average area unit per time unit (hectare per year) of a pasture plot.
- Field data were obtained at the plot level (individual SBPPRL and NG, regardless of the rest of the farm). We restrain our analysis to the conditions in which data was obtained, which were that only cows grazed the plots, eating the same amount of concentrated feed in both SBPPRL and NG plots. As we will show, SBPPRL have higher stocking rates than NG, but this is due to more grazing days.
- Cows have one steer per year. Steers are removed from the pasture after weaning at six months of age.
- For a whole-farm assessment, we consider that farmers sow SBPPRL in their farms in order to eliminate the need for concentrated feeds as much as possible.

Despite the use of this scenario, we refer as often as possible alternative scenarios and their consequences. For example, for the carbon balance in Chapter 3, we also assumed changes in stocking rate due to transfers of steers from intensive to extensive production, as well as feed intake from cows, and respective decrease as a way to make use of the higher sustainable stocking rate of SBPPRL.

1.8 Is our study on the right side of the fence?

The system boundary for our study is the animal farm gate. Therefore, we do not study the whole life cycle of meat. This may seem strange at first, since meat is one of the products with higher environmental impacts. Project EIPRO – Environmental Impact of PROducts (Tukker *et al.*, 2006) studied which were the groups of products with the greatest impacts. They conclude that those groups are food and drink, housing and private transport.

For Portuguese meat production, a specific study was done by Simões *et al.* (2005). All steps in the life cycle of meat were considered, namely the production of animal feed (concentrated feeds and pastures), slaughtering, transportation, storage, private transport to shops and cooking. It was determined that about 90% of the total impact in two categories (namely greenhouse gas emissions and energy resource consumption) was due to the first part of this cycle, which occurs until the animal leaves the farm heading for the slaughterhouse.

Our choice of system boundary is, then, justified by this fact. We only go as far the animal farm gates, because up until that point is where most of the significant impact is. Besides, we are comparing meat produced by several different pasture systems. The systems are only different up until the animal leaves farm. Afterwards, we consider that they follow the same path to the plate[23].

1.9 Objectives and overview of the thesis

In this first Chapter, we defined the systems studied and the scenarios we use next. Considering the framework set by this analysis, our main objectives are the following:

- Determine the potential for SBPPRL to sequester carbon, showing that they are a carbon sink as a bovine production system;
- Determine whether SBPPRL provide other environmental services besides carbon sequestration;
- Connect research results with the PCF Project.

Our study is limited to one product, bovine meat, produced in three different systems of rainfed pastures (considering only marginal substitutions between them, so that no global large scale effects take place). Even though we are including indirect (life cycle) effects, the border of our analysis of the production chain is the selling of the animal to the slaughterhouse. From then on, all three options are similar. Throughout this work, and unless specifically noted, the functional unit of the analysis is 1 hectare of pasture. Throughout our study, we discuss as much as possible the effect of changes in assumptions in final results.

The remainder of the thesis is structured as follows.

We noted, while describing the system effects of SBPPRL in a causal relation scheme, that SOM is the key variable to understand their environmental and agronomic effects. It is, for example, by accumulating SOM that pastures retain carbon in the soil. Therefore, in the second Chapter we determine a model for SOM increases in SBPPRL and alternative natural grasslands, using field data. Because SOM is so important in the rest of this work, we focus in depth on the analysis and discussion on results in Chapter 2.

In the third Chapter, we begin by quantifying carbon sequestration in pastures. We compare that value with other results for other agricultural land uses. Then, we determine the carbon balance, including emission sources such as livestock, grasses and limestone, to study if the system as a whole is indeed a sink. Then, we turn to other environmental services, such as soil protection and biodiversity. We then broaden the scope of our analysis by calculating the life cycle impacts of the system. We end with a brief reference to irrigated SBPPRL, and the comparison with the possibility of biofuel production.

Then, having studied the environmental aspects of the system, in the fourth Chapter we turn to economic considerations. We study consumer valuation of quality products, and then focus on the possibility for payments for carbon sequestration in SBPPRL, which makes the connection from applied research to policy advisory. We study whether the PCF could have an advantage from paying carbon sequestered in pastures, and then proceed to describe the project which was actually submitted to and approved by the

[23] Note that this may not be completely accurate. Meat quality may be different depending on the way the animal was produced. And the quality of meat determines the stores where meat is sold, the way it is cooked and the quantity that is consumed. However, since we have no data on any of these factors, we had to neglect them.

PCF. To wrap up our work, we will propose a research plan to provide answers to open questions.

2. Soil organic matter dynamics in Portuguese pastures

In Chapter 2, the SBPPRL system is described and compared with natural pastures. The hypothesis for the present thesis is presented by formulating nine statements concerning the qualitative differences between the two options. We build and calibrate a model to study the short-term dynamics of soil organic matter (SOM), which is the key parameter to characterize the environmental and agronomic effects of pastures. SOM results obtained from the models show which grassland system provides higher SOM increases. We also study several methodological issues on the models used, like the sampling depth and assumptions regarding which factors influence the mineralization rate. Finally, we conclude on the average potential of SBPPRL to increase SOM concentration.

2.1 A primer on soil science

2.1.1 Soil structure and type

Soils are composed of a matrix, which is the solid component, consisting of organic and inorganic elements which form a porous system. The pores of this matrix are filled with water and gaseous substances, such as oxygen. Soils are characterized according to the characteristic of both the inorganic and the organic parts, as well as the percentage of each.

Regarding the inorganic part, soils are also characterized according to the rock that gave them origin, and also according to their texture. Texture is a measure of the size distribution of the solid particles. Some particles are relatively thick in diameter. But the smallest particles are those responsible for most chemical properties of soils (de Varennes, 2003). These small components are three:

- Sand – Particles with diameters between 2 and 0.02 mm;
- Lime – Particles with diameters between 0.02 and 0.002 mm;
- Clay – Particles with diameters smaller than 0.002 mm.

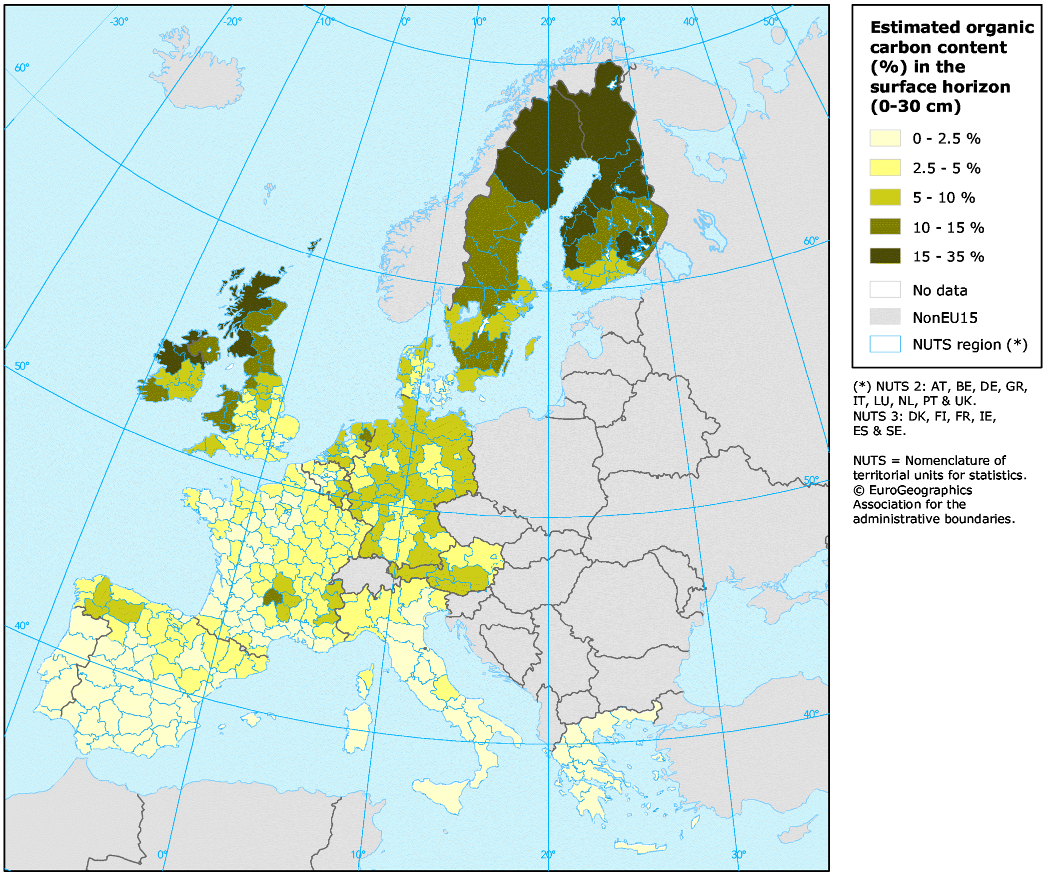
This is a particularly important classification for soils, since it translates how soils are aggregated, and hence also hydraulic characteristics such as field capacity, which is higher in clay soils (de Varennes, 2003).

Regarding the organic component of soils, note that this soil organic matter (henceforward called only SOM) is composed, as any type of organic matter, by elements such as nitrogen, carbon and hydrogen. Soils with a natural high SOM level are those where there is a slow decomposition of these organic elements in soils. This process, known as soil respiration or SOM mineralization, is executed by bacteria in aerobic conditions. In most conditions, mineralization is influenced by soil oxygen, soil moisture, pH, temperature, C:N ratio, type and abundance of clay minerals (Bot and Benites, 2005).

Therefore, low SOM mineralization rates occur in zones with less oxygen diffusion, such as damped areas, or cold regions (de Varennes, 2003). According to EEA (2004), 57.1% of Portugal's soils have low or very low SOM (between 0.5 and 2.0%), as shown in Figure 10. SOM in Europe is a characteristic of latitude, since it affects all

Mediterranean countries: Portugal, Spain, Italy and Greece, and even a very significant part of the French territory.

Figure 10 – European soil organic matter content (%) in the surface horizon (0-30cm).



Source: http://local.pt.eea.eu.int/.

The richness of the soil in terms of SOM is one of the most important parameters to determine the soil's quality. It is also the key to understand the ecosystem services provided by soils.

2.1.2 The key role of SOM

SOM is composed of living organisms (bacteria, fungi, plant roots and animals), dead animal and plant tissues in several stages of decomposition but still recognizable, and a complex mixture of decomposed, modified or reprocessed material called humus (which is usually 60 to 80% of all SOM) (Bot and Benites, 2005; de Varennes, 2003).

SOM quantity influences the quality[24] and resilience[25] of soils, since it determines the soil's physical and chemical proprieties. Humus is responsible in the most part for the aggregation of clay, and therefore is essential to keep soils stable. High SOM

[24] Quality of soils is defined, according to de Varennes (2003), as the capacity of soils to accept, retain and recycle water, nutrients and energy.

[25] Resilience of soils is defined, according to de Varennes (2003), as the capacity of soils to (almost) revert to the initial state after it has been disturbed (e.g. SOM returning to initial values after tillage).

concentration implies a high capacity for water and nutrient retention, and is therefore crucial in terms of plant productivity (Trumbore and Czimczik, 2008).. As more organic matter decomposes and enters the structure of the soil, more nutrients are being recycled and made available to plants[26] (de Varennes, 2003). We can now understand the environmental outcomes of these processes.

2.1.3 Environmental services provided by soils

The importance of soils in frequently underestimated. Soils are a resource which, despite all tries, is still hardly substitutable. Soils with high quality and resilience have mainly four functions, which represent environmental gains (de Varennes, 2003):

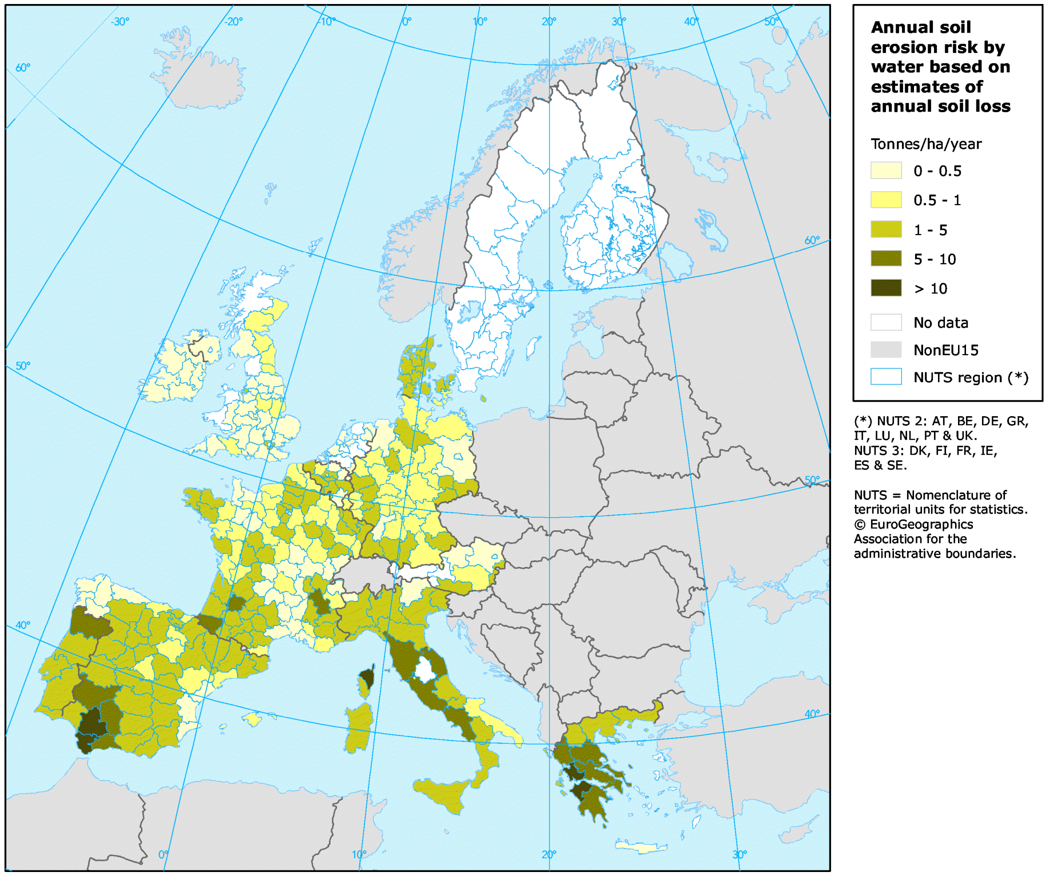
- To support plant growth, both mechanically and in terms of water and nutrient feed. Soils with high SOM levels are better for plants, which become more productive;
- To provide conditions for the recycling of both animal and plant residues and dead tissues, making their components available again in the ecosystem;
- To establish ecological niches for a fauna that goes from bacteria and fungi to small mammals, thus promoting biodiversity;
- To regulate the water cycle, both in terms of water flow quantity and quality, making water available for plants and at the same time regulating the outflow and thus preventing overflows in superficial waters due to high precipitation events.

Plant vegetation cycles are dependent of soils as a means to develop fixed roots with which they exchange heat and mass. Soils are relatively stable means, thus protecting the roots and seeds of high temperature variability. They also hold incoming water from precipitation and irrigation. This water is later provided to plants as they need it for use and transpiration. Soils also filter and withhold many substances which otherwise could be toxic to plants and downstream water masses (de Varennes, 2003).

In fact, diminishing water surface runoff and soil erosion have positive effects beyond the borders of the farm and soil type. Sediments, nutrients, organic matter and pesticides carried in water contribute to silting, eutrophication and contamination of superficial waters (EEA, 2004). These effects are known, but its true costs are still hard to pinpoint. Figure 11 shows the annual soil risk erosion by water in Europe, which is clearly higher in Mediterranean countries (including Portugal).

[26] Nitrogen, phosphorus and potassium are usually the major nutrients made available by soils with high SOM concentration, but there are other important ones, such as sulphur and carbon compounds.

Figure 11 – Annual soil erosion risk by water, based on estimates of annual soil loss.



Source: http://local.pt.eea.eu.int/.

But there is one effect which has not been mentioned yet. SOM is composed by the stoichiometric percentage of 58% of carbon. Since soil organic carbon (henceforth referred to as SOC) is mostly formed after decomposition of plant and animal residues, it is the result of atmospheric carbon (CO_2) sequestered by plants during photosynthesis[27]. At the same time, soil respiration, the process through which organic matter is decomposed, produces CO_2 (Trumbore and Czimczik, 2008).

2.2 Effects of pastures in the soil

As we have shown, SOM quantity influences the quality and resilience of soils, since humus is responsible in the most part for its aggregation (Bot and Benites, 2005). High SOM concentration contributes to many environmentally positive effects, as shown in the previous Chapter.

Pastures are one specific land use with widespread benefits to organic matter pools in soils. Pastures provide particularly high and stable SOC pools. Guo and Gifford (2002) indicate that SOC stocks decline after land use changes from pasture to cropland by 59%, and increase after land use changes from crop to pasture by 19%. Martens *et al.* (2004) show that for systems undisturbed for 130 years, pastures' soils had 25% more

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[27] Even in the case of the decomposition of animal material, somewhere in the trophic chain of the animal there was the primary consumption of plants, and therefore photosynthetic carbon.

carbon than cropped soils. Improvements such as fertilization and sowing in pastures increase SOM accumulation by increasing pasture productivity.

Therefore, pastures provide particularly high and stable carbon pools. But it is still necessary to determine exactly how SOM accumulates in pastures. Several authors have also tried to model SOM or soil organic carbon (SOC) in time. Six *et al.* (2002) found that an asymptotic curve depicts SOC variation with carbon input. Six *et al.* (2004) distinguish several SOM pools, based on their stability, and determine a saturation upper bound for all of them. Other, like West and Six (2007), have looked into soil carbon saturation. They argue that it is possible to increase SOC level by land management (e.g. no-tillage) and increased inputs (e.g. fertilizers), but only up to a given point. In time, SOC reaches an upper bound. This upper bound is that for which organic input into the soil and organic matter mineralization (soil respiration) are equal.

There is a significant quantity of studies that does point to temperature as the main parameter that controls soil respiration. Cao *et al.* (2004), for example, argue that grazing intensity also affects mineralization rate – low grazing intensity plots have higher soil respiration than high intensity ones. They claim that grazing increases the dependence of the mineralization rate on temperature. These results are supported by Raich and Tufekcioglu (2000), who have studied soil respiration rates for several land uses across several biomes. Their results show no significant differences between crop soil and either naked soils or grassland soils. Forest soils are the only ones with a slightly lower mineralization rate. What does seem to matter in terms of soil respiration is aboveground net primary productivity in grasslands, and meteorological factors such as temperature and precipitation. However, other studies such as Fitter *et al.* (1998) showed that soil respiration in grasslands depends on radiation fluxes, but is unrelated to temperature. They argue that temperature usually shows up as a significant parameter in short-term studies only.

What all studies seem to confirm is that pastures have a high potential for the accumulation of stable SOM. This accumulation is obviously limited by an upper bound. In pastures, if there are no land use conversions or other management activities, discounting for climate effects, SOC reaches a long term steady state equilibrium.

For Portugal, some early results hinted that in 10 years soils with Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL) increase SOM from 1 to 3% (Crespo, 2004). This value is confirmed by some empirical data. At Herdade dos Esquerdos, in Vaiamonte (Portalegre, Portugal), following a programme of SBPPRL installation, SOM concentration across the farm increased from between 0.7% and 1.2% in 1979 to between 1.45% and 4.40% in 2003 (Crespo *et al.*, 2004; Crespo, 2006a, 2006b). This SOM increase is predictably higher than that of any natural grassland under any form of management. However, we lack a systematic study that compares SOM increases for various types of pasture.

2.3 Determining SOM dynamics in pastures

For the rest of this section, we develop a model to determine the average trend of SOM concentration in NG, FNG and SBPPRL[28]. Our main objective is to determine a the average SOM accumulation potential in each grassland system. We hypothesise that the

[28] This Chapter is based on Teixeira *et al.* (2010c).

variation in SOM over time is the balance between SOM input and output in a plot. The implication of this model is that SOM asymptotically reaches a long-term equilibrium.

The dynamic parameters in this model are the SOM input and the mineralization rate. In order to estimate the values of these parameters, we calibrate the model statistically using field data. Data was collected from 2001 to 2005 in several locations in Portugal, during two demonstration projects. Project AGRO 87, "Sown biodiverse permanent pastures rich in legumes – a sustainable option for degraded land use" (Carneiro *et al.*, 2005) collected samples in six farms. At the same time, Project PAMAF 4073, which was continued as Project AGRO 71, "Recovery and improvement of Alentejo's degraded soils using grasslands" collected samples from two additional farms . We filled-in some missing data, since some samples were not collected.

We use two statistical methods for calibration: one where all parameters are specific for each grassland system, and one where there is only a specific SOM input. We then compare the dynamics of the three systems in 10 years. Finally, we validate the results obtained and draw some conclusions. In order to do so, we compare our results with other data and studies.

2.3.1 Characterization of the plots

Data was obtained from rainfed pastures in eight farms in Portugal from 2001 to 2005 (Table 5, location in Figure 12). Plot areas ranged from 5 to 15 ha. Each plot's soil and landscape type was approximately homogeneous, in terms of soil and previous use. These pastures were not isolated test sites. They were located in private land currently used by farmers for animal production. Prior to the beginning of the projects, plots were used in a system of long cereal/fallow rotations – one year of crop production for each five to seven years of fallow (which was used as a "natural pasture" featuring spontaneous herbaceous plants). In Farm #1 the NG plot was fertilized in 2002, and so the NG system was lost. In Farms #7 and #8 (Project Agro 71), FNG were not studied. Almost no samples were collected in 2002.

Table 5 – Soil and site characterization in the sites of Projects Agro 87 (farms 1 to 6) and Agro 71 (farms 7 and 8).

Farm No.	Farm	Location	Soil original material	Texture*
1	H. Cabeça Gorda	Vaiamonte	Gneiss	Loam
2	H. Mestre	São Vicente	Limestone	Loamy clay
3	H. Claros Montes	Pavia	Granite	Loamy sand
4	H. Refroias	Cercal	Schist	Loamy sand
5	H. Cinzeiro e Torre	Coruche	Sandstone	Sand
6	Quinta da França	Covilhã	Granite	Loamy sand
7	H. Monte da Achada	Castro Verde	Schist and Greywacke	Sandy loam
8	H. Corte Carrilho	Mértola	Schist	Loamy sand

*** The "feel" method was used. The textural class is ascertained by rubbing a sample of the soil in a moist to wet condition, between the thumb and fingers.**

Figure 12– Map of Portugal, with the indication of the sampling sites of Projects Agro 87 (farms 1 to 6) and Agro 71 (farms 7 and 8).

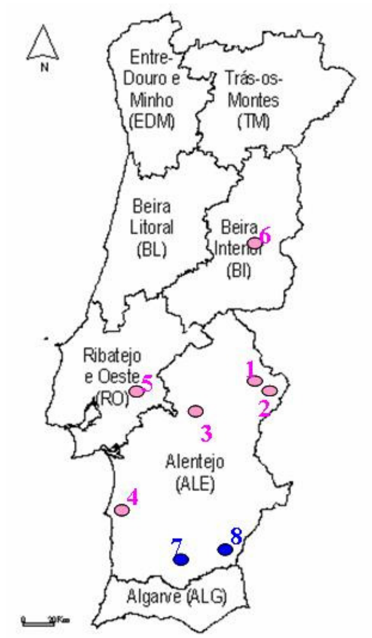


Table 6 shows the main meteorological characteristics of all test sites. Most sites are similar in their characteristics, except for farm #6, which is the only one in Central Portugal.

Table 6 – Main meteorological and texture characteristics of the sites of Projects Agro 87 (farms 1 to 6) and Agro 71 (farms 7 and 8).

Farm No.	Average daily temperature (ºC) [2]	Precipitation (yearly Oct-Sept, mm) [1]				
		2001-02	2002-03	2003-04	2004-05	2005-06
1	16.8	381	548	494	257	549
2	16.3	433	-	462	-	-
3	15.5	476	544	-	190	444
4	16.3	610	750	597	267	-
5	15.5	504	468	-	272	564
6	11.3	-	-	1195	624	-
7	15.5	475	458	478	207	546
8	16.8	451	477	480	-	-

Sources: [1] Actual data for the closest meteorological station in the SNIRH database (http://snirh.pt/). Empty cells are for years when a valid data series was not collected. [2] Portuguese Environmental Agency's Atlas of the Environment (APA, 2009); values are approximat mean figures for the period of 1930 to 1970.

Table 7 shows fertilization applied in SBPPRL and FNG in all sampling sites. Fertilization needs were determined according to the initial soil analysis of all fields. Both grassland systems were subjected to the same fertilization rates during all years of the project. The difference during the installation of SBPPRL is that, previous to sowing, plots are tilled in the upper layer of soil, and a phosphate and potassium fertilizer (superphosphate or 0:21:21) is used. Limestone is added in SBPPRL if soil pH is lower than 5.3 to lower acidity to optimum levels for legumes. Other micronutrients are added, like zinc sulphate copper sulphate or borax. Molybdate (a salt of molybdic acid) is added together with the seeds. SBPPRL were installed with 30 $kg.ha^{-1}$ of seeds.

Table 7 – Fertilization applied in SBPPRL and FNG in the sites of Projects Agro 87 (farms 1 to 6) and Agro 71 (farms 7 and 8).

Farm No.	Quantity applied ($kg.ha^{-1}$)				
	Limestone *	Superphosphate 18%		0:21:21	
		Sowing *	Maintenance	Sowing *	Maintenance
1	0	350	200	0	0
2	0	350	0	200	0
3	2000	0	350	0	200
4	2000	0	250	0	200
5	1000	0	0	450	200
6	2000	200	200	0	0
7	1000	0	100	300	200
8	0	1000	0	0	200

*** For SBPPRL only. SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.**

Table 8 shows the average yearly stocking rate in each grassland system (considering all farms). Farmers registered the number of animals put on each plot each day. That information was then averaged in a year (considering also the days when there were no animals in the pasture) (Carneiro *et al.*, 2005). Note also that the two types of natural pasture (NG and FNG) were obtained from the division in half of the same plot (part fertilized and part not) and were thus under the same livestock management. In 2004-05 figures are much lower because of a severe drought in Portugal. Nevertheless, Table 8 shows that SBPPRL support a higher stocking rate. Table 8 will be important in the Discussion to provide intuition for our results.

Table 8 – Average yearly stocking rate in SBPPRL and natural pastures (NG and FNG) in the sites of Projects Agro 87 (Carneiro *et al.*, 2005).

Year **	Average stocking rate ($LU.yr^{-1}$)	
	SBPPRL	Natural *
2001-2002	0.73	0.39
2002-2003	1.13	0.44
2003-2004	1.22	0.43
2004-2005	0.36	0.14

*** "Natural" refers to both NG and FNG, since both were managed with the same stocking rate.**
**** From October of one year to September of the next year.**

LU – Livestock Unit; NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; One LU is the equivalent of one adult cow (a steer corresponds to 0.6 LU and an ewe to 0.15 LU).

2.3.2 SOM data

SOM determination begins with collection of soil samples. One composite sample was collected in each plot. Each composite sample was obtained from the mix of a variable number os sub-samples collected throughout each homogeneous plot, in order to be representative of the average SOM in the plot. The samples were collected, and then analysed, by Laboratório Químico-Agrícola Rebelo da Silva (LQARS), which is the Portuguese Government's official soil laboratory.

In the preparation of the sample for analysis, any living organisms, as well as gross animal and plant material, are removed from the sample. Therefore, from the general composition of SOM indicated in section 2.1.2 (Bot and Benites, 2005), results in this section only refer to the 60-80% which are humus and some minor organic material.

Sample laboratorial analysis begins with the entire sample being spread on a tray and dried overnight, at 35-37ºC. The sample was crumbled mechanically and passed through a 2mm stainless steel sieve. The sieved material is the 'fine soil' subject to analysis. Samples were then analysed for several parameters, such as pH, nitrogen, phosphate and potassium levels, lime requirement and SOM concentration. It was the latter parameter that we used in this work.

From 2001 to 2004, results for SOM concentration were obtained by the wet oxidation method. This method consists in the digestion of organic carbon by sodium dichromate, followed by colorimetric determination on a molecular absorption spectrophotometer at 640 nm (Carter, 1993). From 2005 on, a dry combustion method was used (including for samples collected in 2005). It consists of the determination of total carbon by dry combustion, according to ISO Standard 10694, using a CNS elemental analyzer (Rodeghiero *et al.*, 2009). Organic carbon is determined indirectly after correction of the total carbon concentration for the carbonates present in the soil sample. Both methods measure SOC concentration, which is them multiplied by 1.724 (assuming that 58% of organic matter is carbon) to obtain the corresponding SOM concentration. Since two different methods were used, an internal (unpublished) study was conducted by LQARS to obtain an equivalence factor between results. This study guarantees that the results are consistent. Final results for SOM are presented in mass percentage (%) units, equal to grams of SOM per 100 grams of soil.

There are few results available for 2002, in a set obtained from 2001 to 2005. Since our regression models (whose equations we show in the next section) use pairs of points $(SOM_{i,t-1}, SOM_{i,t})$, two pairs are almost always missing: (2001, 2002) and (2002, 2003). If these values are filled-in, we end up doubling the number of observations. Furthermore, FNG are missing more pairs of points than SBPPRL and NG. Therefore, whenever there is a missing value between two others, we calculate the geometric average (the "growth rate" of SOM) of the two observations. For example, assuming that SOM_{t-1} is missing, while SOM_{t-2} and SOM_t are not, we calculate the missing value as:

$$SOM_{t-1} = SOM_{t-2} \cdot \left(\frac{SOM_t}{SOM_{t-2}} \right)^{1/2}. \tag{1.3}$$

2.3.3 SOM dynamic model

As the thorough review done by Pete Falloon and Pete Smith (2009) shows, other models in the literature intend to explain inter and intra annual variability in SOM. To explain such variability, they are required to use environmental variables, such as climate and soil type. Out of the 33 models reviewed and assessed by Falloon and Smith, only one (O'Brien, 1984) had a yearly time step and no meteorological and management variables. But the O'Brien model considered interactions with plants, and had a completely different objective than ours. However, in our case, it is the SOM trend we wish to estimate and not the interannual variation of SOM levels. Our objective is to calibrate a time series to capture the trend of SOM dynamics in the three grassland systems.

Therefore, we used a simple mass balance model for SOM dynamics, calibrated using field data. The model states that the mass percent balance of SOM is the difference between input and mineralization:

$$\frac{dSOM_t}{dt} = K - \alpha \cdot SOM_t , \tag{1.4}$$

where SOM is the SOM concentration (percentage points, equal to $g_{SOM}.100\ g_{soil}^{-1}$) at time t, K is the SOM input, and α is the organic matter mineralization rate.

We solve Equation (1.4) by integrating it between $t-\Delta t$ and t:

$$SOM_t = \frac{K}{\alpha}\left(1-e^{-\alpha\Delta t}\right) + e^{-\alpha\Delta t} SOM_{t-\Delta t} . \tag{1.5}$$

Therefore, the general solution for Equation (1.4) has an asymptotic exponential form. This means that SOM accumulation is limited by an upper bound. In pastures, if there are no land use conversions or other management activities, disregarding climate effects, SOM reaches a long-term equilibrium.

As the inspection of the SOM analysis results in Table 11 will show, farms with high initial SOM still increased their SOM concentration by a relatively high percentage, regardless of the pasture type. To capture this effect, we separate K in a fixed term $K^{'}$ (which is a function of grassland system and not of representative local conditions), and a variable part (which is a linear function of the initial SOM concentration), being the proportionality parameter a):

$$K = K' + a \cdot SOM_0 . \tag{1.6}$$

Equation (1.6) shows that we use initial SOM as a proxy for representative conditions of the location. This approach is justified by the fact that natural soil and climate conditions, as well as the history of the field, determine the initial SOM concentration. This slightly changes the model. Substituting Equation (1.6) in Equation (1.5), we obtain the general expression of the model:

$$SOM_t = \frac{K'}{\alpha}\left(1-e^{-\alpha\Delta t}\right) + \frac{a}{\alpha}\left(1-e^{-\alpha\Delta t}\right) SOM_0 + e^{-\alpha\Delta t} SOM_{t-\Delta t} . \tag{1.7}$$

There are now two alternative approaches to estimate the parameters in Equations (1.5) and (1.7):

1. In the "specific-data model", we consider that all parameters are a function of pasture type ($K_i^{'}, \alpha_i$ and a_i, where $i = \{SBPPRL, FNG, NG\}$). It is necessary to estimate one model per grassland system, obtaining three sets of three parameters each one specific of the grassland type.
2. In the "pooled-data model", we consider that only specific SOM input is a function of pasture type ($K_i^{'}$). It is possible to estimate one single model using all data for all types of pastures, obtaining one set of five parameters. In this

case, we are assuming that the mineralization rate is equal for the three grassland systems.

The difference in statistical procedure is summed up in Table 9. Both models are calibrated estimating a regression equation in which SOM_t is the dependent variable and SOM_{t-1} and SOM_0 are the independent variables. In the specific-data model, we use data from all locations but separately for each grassland system *i*. There are three different regressions, and the nine dynamic parameters are calculated from the nine regression constants. In the pooled-data model, only the constant term has a parameter that depends on the grassland system. We thus consider that the constant term is the sum of three dummy variables, one for each grassland system. The dummy $d_i = 1$ if observation regards grassland system *i* and $d_i = 0$ otherwise[29].

Table 9 – Presentation of the two different approaches to estimate the model for SOM dynamics.

Parameters	
"Specific-data model"	"Pooled-data model"
$K_i^{'} = f(i)$ $\alpha_i = f(i)$ $a_i = f(i)$ $i = \{SBPPRL, FNG, NG\}$	$K_i^{'} = f(i)$ α a $i = \{SBPPRL, FNG, NG\}$
Model	
$SOM_{i,t} = \frac{K_i^{'}}{\alpha_i}\left(1 - e^{-\alpha_i \Delta t}\right) +$ $+ \frac{a_i}{\alpha_i}\left(1 - e^{-\alpha_i \Delta t}\right) SOM_{i,0} +$ $+ e^{-\alpha_i \Delta t} SOM_{i,t-\Delta t}$	$SOM_{i,t} = \frac{K_i^{'}}{\alpha}\left(1 - e^{-\alpha \Delta t}\right) +$ $+ \frac{a}{\alpha}\left(1 - e^{-\alpha \Delta t}\right) SOM_{i,0} +$ $+ e^{-\alpha \Delta t} SOM_{i,t-1}$
Regression Equations	
$SOM_{i,t} = C_{i,1} + C_{i,2} \cdot SOM_{i,0} + C_{i,3} \cdot SOM_{i,t-1}$ (3 separate sets of data and 3 separate regressions, one for each *i*)	$SOM_t = \sum_i \theta_i d_i + C_2 \cdot SOM_0 + C_3 \cdot SOM_{t-1}$ (1 set of data and 1 single regression, for all *i*)
Obtaining parameters from regression constants	
$\begin{cases} \alpha_i = \frac{-ln\left(C_{i,3}\right)}{\Delta t} \\ K_i^{'} = \frac{C_{i,1} \cdot \alpha_i}{1 - e^{-\alpha_i \Delta t}} \\ a_i = \frac{C_{i,2} \cdot \alpha_i}{1 - e^{-\alpha_i \Delta t}} \end{cases}$ (9 parameters)	$\begin{cases} \alpha = \frac{-ln\left(C_3\right)}{\Delta t} \\ K_i^{'} = \frac{\theta_i \cdot \alpha}{1 - e^{-\alpha \Delta t}} \\ a = \frac{C_2 \cdot \alpha}{1 - e^{-\alpha \Delta t}} \end{cases}$ (5 parameters)

SOM – Soil Organic Matter; $K^{'}$ - specific SOM input; α - mineralization rate; a – contribution of initial plot conditions for SOM increase; C – regression constant; i – grassland system; NG –

[29] This version of the model, and corresponding results, were published in Teixeira *et al.* (2008a, 2010a).

Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.

SBPPRL plots are tilled in the first year, and thus there is increased SOM mineralization for t = 1. Besides, in the first year plants blossom only from the seeds which were sown. It is only from the second year on that a seed bank is created (much larger than the amount of seeds initially sown) from which the pasture permanently blossoms every year. Less seeds imply a lower quantity of biomass produced in the first year, unlike in the following years.

Therefore, we also test the hypothesis that SOM dynamics is different in the first year. In order to do so, we estimate the same regression in Equation (1.7), but consider one more dummy variable, namely to separate results from SBPPRL in the first year. For simplicity in the calculation, we introduce the dummy in the parameter *K'*. The regression equation for the specific-data model for SBPPRL, in Table 9, thus becomes:

$$SOM_t = \sum_j \theta_j d_j + C_2 \cdot SOM_{t-1} + C_3 \cdot SOM_0, \tag{1.8}$$

where $j = \{SBPPRL(t=1), SBPPRL(t>1)\}$. Note that, for FNG and NG, the model is the same as before, and is still represented by the regression equation in Table 9. The pooled-data model for all grassland systems in Table 9 is now represented for all $k = \{SBPPRL(t=1), SBPPRL(t>1), FNG, NG\}$ by

$$SOM_t = \sum_k \theta_k d_k + C_2 \cdot SOM_{t-1} + C_3 \cdot SOM_0, \tag{1.9}$$

where d is a dummy variable equal to one for observations of grassland system k and zero otherwise, and θ is the respective regression coefficient.

We estimate the parameters of each model using software SPSS Statistics 17.0 with an Ordinary Least Squares (OLS) method (Verbeek, 2001).

We considered other specifications of the model, namely:

- Logarithmic filling-in of data;
- Using soil and precipitation variables instead of the initial SOM concentration;
- Linear approximation of the model.

All of these options were rejected, but results are shown in Appendix I – Alternative estimations of the SOM model.

2.3.4 Application and validation of the SOM model

The procedures referred in this section are a simplified approach intended only for validation of some parameters obtained.

To determine the grassland system in which the increases in SOM was highest, we calculate SOM increases in all systems starting from the same arbitrary initial SOM. Since we need a scenario for initial SOM, we assumed a starting hypothetical concentration of 0.87%, which is believed to be representative of Portuguese soils in the

Alentejo region[30]. We also apply the model to the initial SOM concentrations measured in each farm. This procedure tests the validity of the model by comparing measured and calculated results.

All results in this section are shown in percentage points, equal to $g_{SOM}.100\ g_{soil}^{-1}$. However, for comparison purposes, we need to determine the carbon equivalent to SOM increases. Stated in another way, we find the equivalent to 1% SOM in terms of t $C.ha^{-1}$. We begin by using the Adams (1973) equation to correct mineral soil density:

$$BD = \frac{100}{\frac{SOM}{0.244\,g.cm^{-3}} + \frac{100 - SOM}{MBD}}, \tag{1.10}$$

where BD is soil bulk density ($g.cm^{-3}$), MBD is soil mineral bulk density ($g.cm^{-3}$), SOM is SOM concentration (percentage point, equal to $g_{SOM}.100\ g_{soil}^{-1}$), and 0.244 $g.cm^{-3}$ is the MBD for which MBD = BD regardless of the SOM concentration.

Based on Rawls and Brakensiek (1985), MBD of mineral soils varies between 1.20 and 1.69 $g.cm^{-3}$. The indicative MBD in Portugal is 1.25 $g.cm^{-3}$, but soils which are not tilled are more compact, and thus MBD is around 1.40 $g.cm^{-3}$ (Mário Carvalho, personal communication). In the following calculations we will use both.

Results are shown in Table 10. Starting from the MBD values, we consider 1% SOM and calculate BD. Since the soil samples were collected at up to 10 cm, the SOM mass per unit is subsequently determined per unit area. Final values are obtained by converting to tons per hectare.

Table 10 - Carbon sequestration equivalent to the increase in SOM of 1 pp in 10 cm.

MBD ($g.cm^{-3}$)	SOM (pp)	BD ($g.cm^{-3}$)	g(SOM).cm^{-3}	Depth (cm)	g(SOM).cm^{-2}	t (SOM).ha^{-1}	t $C.ha^{-1}$
1.25	1	1.20	0.0120	10	0.120	12.0	6.96
1.40	1	1.34	0.0134	10	0.134	13.4	7.77

MBD – Mineral Bulk Density, SOM – Soil Organic Matter, BD – Bulk Density, pp – percentage point.

We use Table 10 to convert *K'* from % (in mass) per unit of time into t(SOM).ha^{-1}, which is then converted to equivalent plant production. In order to do so, we assume that only humus is captured in SOM analysis, which is at most 80% of belowground biomass in pastures (Bot and Benites, 2005). Furthermore, the IPCC (1997) indicates 2.8 as the default root to shoot ratio (R:S) for semi-arid grasslands. This value is consistent with the R:S of 0.5 to 4.8 in grazed pastures, which is the range of the comprehensive data for several regions gathered by Coupland (1976). Dividing *K'* by 80% and then by R:S, we obtain an estimate of aboveground production.

The final values in Table 10 may also be used to determine the equivalent of SOM increases in terms of carbon. In order to do so, we had to assume a conversion factor between SOM and soil organic carbon (SOC). Since approximately 58% of SOM is SOC (IPCC, 1997 and 2003), and both are measured as $g.100\ g_{soil}^{-1}$, then the mass of SOC is also 58% the mass of SOM. Therefore, 1% of SOM increase corresponds to the

[30] This value was obtained as the average SOM concentration in the pasture plots of a systematic grid where soil samples were collected (Fátima Calouro, personal communication). Results are unpublished.

sequestration of 6.96 – 7.77 t C.ha^{-1}. These values are used when comparing our results to other studies.

2.4 Results of the calibration of the SOM model

2.4.1 Results from soil analyses

Results from soil analyses for SOM concentration are shown in Table 11 (Carneiro *et al.*, 2005). Considering the difference between the first and the last year, the minimum SOM increase for SBPPRL was obtained in Coruche (Farm #5), where the pasture blooming after the first year establishment was poor. The highest increases for SBPPRL were obtained in the more productive gneiss soil in Vaiamonte and the schist soils of Herdade de Refróias. In Herdade de Refróias, SOM at the beginning was already 3%. Farms #7 and #8 increased SOM concentration in SBPPRL by 0.35 pp and 0.43 pp per year. Table 11 also shows filled-in data underlined. Direct comparison with natural (non-fertilized) grasslands at each site shows that increases are usually higher for SBPPRL.

Table 11 – SOM concentration in each grassland system for experimental sites (0-10 cm).

Farm No.	Grassland system	SOM (%)				
		2001	2002	2003	2004	2005
1	SBPPRL	1.55	2.17	3.05	3.60	3.80
1	FNG	1.30	1.84	2.60	3.40	3.00
2	SBPPRL	1.75	2.15	2.65	2.70	5.40
2	FNG	1.95	2.42	3.00	4.50	3.50
2	NG	1.95	2.29	2.70	4.00	4.00
3	SBPPRL	0.45	0.73	1.20	1.63	1.60
3	FNG	0.68	0.86	1.10	1.40	2.00
3	NG	0.92	1.01	1.10	1.20	1.15
4	SBPPRL	3.40	3.08	5.10	4.60	5.60
4	FNG	3.80	4.23	4.70	5.40	5.60
4	NG	3.80	4.23	4.70	5.60	-
5	SBPPRL	0.65	0.81	1.00	1.28	1.50
5	FNG	0.55	0.78	1.10	1.15	1.25
5	NG	0.55	0.61	0.68	0.75	0.55
6	SBPPRL	1.82	2.09	2.40	2.18	2.70
6	FNG	1.75	2.25	2.90	2.70	2.70
6	NG	1.75	2.33	3.10	2.40	-
7	SBPPRL	0.55	0.83	1.14	1.60	-
7	NG	1.10	1.20	1.20	1.33	-
8	SBPPRL	0.80	1.40	1.54	2.08	-
8	NG	0.84	1.06	1.10	1.45	-

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; missing data was filled in using the geometric average of the increase rates, and is shown underlined; when result is "-", data could not be filled-in.

Table 12 shows averages for SOM concentration in each grassland system and each year (using only original data). In 2005, FNG have SOM concentrations similar to SBPPRL, but they also start from higher initial SOM concentrations. We can also see that even though the drought in 2005 is shown in stocking rate averages, SOM did not decrease significantly. This means that the lack of water decreased plant production, and therefore also SOM input, but it also decreased SOM mineralization.

Table 12 – Average SOM concentration in each grassland system and year (samples taken in Autumn).

Grassland system	SOM concentration per year and grassland system (%)				
	2001	2002	2003	2004	2005
SBPPRL	1.50	1.51	2.26	2.46	3.43
FNG	1.87	-	2.57	3.09	3.01
NG	1.67	1.13	2.32	2.39	1.90

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

All plots were natural pastures before the beginning of the project. Results for SOM in 2001 are previous to the installation of SBPPRL and to the fertilization of FNG making them representative of the initial soil conditions. Therefore, SOM initial value in each location is equal to SOM in the first year, 2001:

$$SOM_0 = SOM_{2001}\,. \tag{1.11}$$

The use of SOM_0 as a proxy for representative local conditions (soil parameters, climate conditions, former management), and the use of the first year as the initial SOM concentration, are both justified by visual inspection of results in Table 11 and by statistical results shown in Table 13. SOM_{2001} changes more between farms than between grassland systems within the same farm.

Table 13 shows that the average of all observations for SOM_0, considering all farms and grassland systems (calculated without the filled-in values), is 1.66 %, with a standard deviation of 1.06 pp (n = 18). If we first calculate the average SOM_0 in each of the seven farms, and only then calculate the overall average, the result will be the same, namely 1.66% (average weighted by the number of observations in each farm, which are either 2 or 3). But the standard deviation is much lower, 0.14 pp.

This means that in each farm (average of results for all grassland systems) the standard deviation of the average SOM_0 varies much less than in the overall sample (average of all observations for SOM_0). But when we calculate the average $SOM_{i,0}$ for each grassland system *i*, then we obtain standard deviations similar to that of the whole sample. This fact supports the assessment made from the results in Table 11: the initial SOM concentration is correlated with the farm but not with the type of pasture. It is, therefore, a logical choice for proxy for the specific conditions of a plot in the model.

Table 13 – Average and standard deviation for initial SOM concentration in each farm and grassland system.

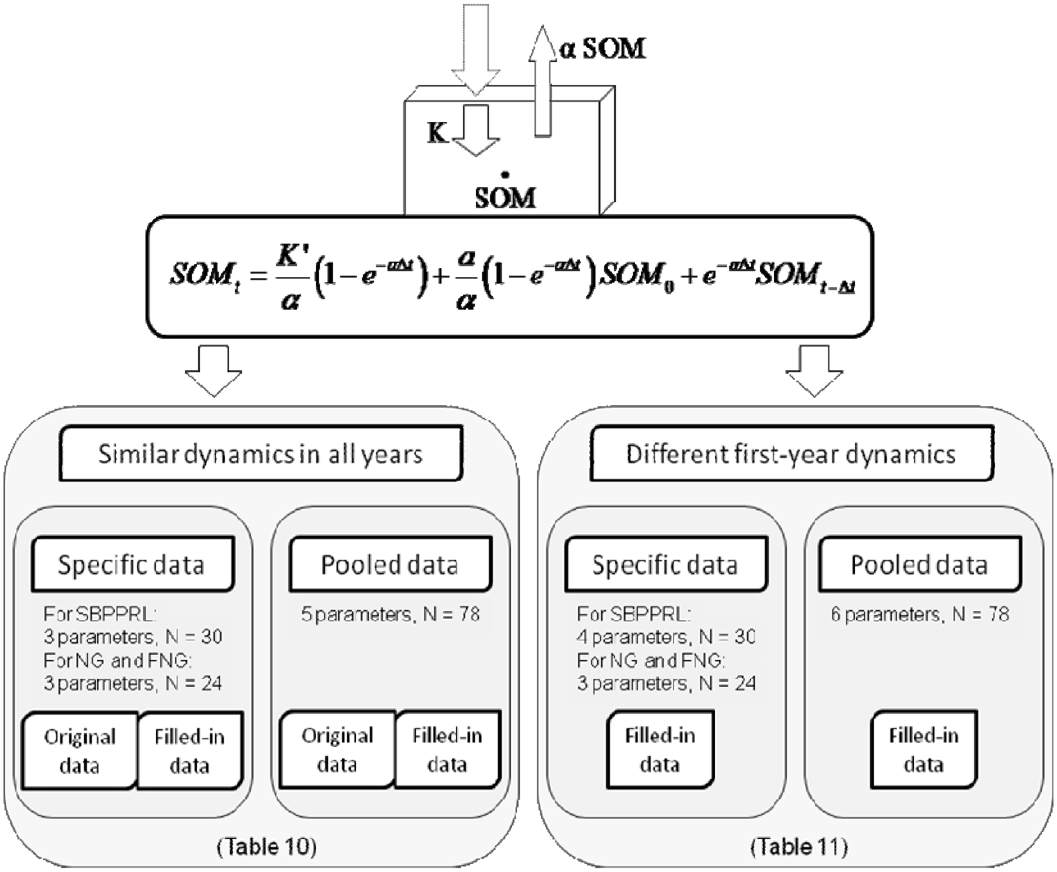
<table>
<tr><th>Farm</th><th>Grassland system</th><th>Initial SOM concentration (%)</th><th>Average (%)</th><th>Standard deviation (pp)</th><th>N (#)</th></tr>
<tr><td rowspan="2">1</td><td>SBPPRL</td><td>1.55</td><td rowspan="2">1.43</td><td rowspan="2">0.18</td><td rowspan="2">2</td></tr>
<tr><td>FNG</td><td>1.30</td></tr>
<tr><td rowspan="3">2</td><td>SBPPRL</td><td>1.75</td><td rowspan="3">1.88</td><td rowspan="3">0.12</td><td rowspan="3">3</td></tr>
<tr><td>FNG</td><td>1.95</td></tr>
<tr><td>NG</td><td>1.95</td></tr>
<tr><td rowspan="3">4</td><td>SBPPRL</td><td>3.40</td><td rowspan="3">3.67</td><td rowspan="3">0.23</td><td rowspan="3">3</td></tr>
<tr><td>FNG</td><td>3.80</td></tr>
<tr><td>NG</td><td>3.80</td></tr>
<tr><td rowspan="3">5</td><td>SBPPRL</td><td>0.65</td><td rowspan="3">0.58</td><td rowspan="3">0.06</td><td rowspan="3">3</td></tr>
<tr><td>FNG</td><td>0.55</td></tr>
<tr><td>NG</td><td>0.55</td></tr>
<tr><td rowspan="3">6</td><td>SBPPRL</td><td>1.82</td><td rowspan="3">1.77</td><td rowspan="3">0.04</td><td rowspan="3">3</td></tr>
<tr><td>FNG</td><td>1.75</td></tr>
<tr><td>NG</td><td>1.75</td></tr>
<tr><td rowspan="2">7</td><td>SBPPRL</td><td>0.55</td><td rowspan="2">0.83</td><td rowspan="2">0.39</td><td rowspan="2">2</td></tr>
<tr><td>NG</td><td>1.10</td></tr>
<tr><td rowspan="2">8</td><td>SBPPRL</td><td>0.80</td><td rowspan="2">0.82</td><td rowspan="2">0.03</td><td rowspan="2">2</td></tr>
<tr><td>NG</td><td>0.84</td></tr>
<tr><td colspan="2">Average of farms</td><td>-</td><td>1.66</td><td>0.14</td><td>7</td></tr>
<tr><td colspan="2">All observations</td><td>-</td><td>1.66</td><td>1.06</td><td>18</td></tr>
<tr><td colspan="2">Average for SBPPRL</td><td>-</td><td>1.50</td><td>0.99</td><td>7</td></tr>
<tr><td colspan="2">Average for FNG</td><td>-</td><td>1.87</td><td>1.21</td><td>5</td></tr>
<tr><td colspan="2">Average for NG</td><td>-</td><td>1.67</td><td>1.17</td><td>6</td></tr>
</table>

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

2.4.2 Regression results

Absolute values used in the regression to calibrate the parameters of the model were those in Table 11 (percentage values, equal to $g_{SOM}.100\ g_{soil}^{-1}$). Regression results are shown in Table 14 and in Table 15. The difference between the two tables is the estimation procedure of the model, which is resumed in Figure 13.

Figure 13 – Different estimation procedures used for the SOM dynamic model.



K', K, α***, a*** **– Parameters in Equation (5); N – number of observations.**

2.4.2.1. <u>Results assuming similar dynamics in the first and following years</u>

We begin our analysis by Table 14. Results show that the statistical fit of the models to base data, measured with the adjusted-R^2, is higher for the pooled-data model. The root Mean Standard Error (rMSE), however, is sometimes lower for specific-data models.

Regarding the relative order of parameters for each grassland system, inspection of Table 14 shows lower *K*' and lower α for NG. A low α is consistent with the fact that the previous management in NG was maintained, and therefore the SOM pool is stabilized. Note however that negative *K'* in the specific-data model for NG, obtained using the initial SOM concentration, does not denote a negative SOM input. The total SOM input is *K*, equal to the fixed term *K'* plus the variable term which depends on the initial SOM concentration. Therefore, the sum of both is positive for initial SOM higher than 0.11%, which is the case in every test plot (and in virtually all Portuguese agricultural land).

Table 14 also shows parameters *K'* and *a* are higher for FNG as compared to SBPPRL in equivalent specific-data models. But field evidence on stocking rates in Table 8 states otherwise. Since stocking rates are always higher for SBPPRL plots, *K* should be higher for SBPPRL (assuming that there is a direct relation between stocking rate and biomass production). Note that the overall SOM balance is still favourable to SBPPRL because the mineralization rate is also higher for FNG.

Table 14 also shows that the use of filled-in data slightly improves the statistical fit of the models. Its conclusions, however, are qualitatively similar to those obtained using only original data. This indicates that filling-in does not bias results. This is due to the fact that increases between consecutive years in the method chosen for filling-in are calculated in a way which is qualitatively similar to the model (decreasing growth rate).

Therefore, we can use filled-in data to calculate values using the first-year dummy for SBPPRL. The results are shown in Table 15. We would not be able to use this dummy variable using only original data, since in 2002, which corresponds to SOM changes in the first year of settlement, there are only three plots (out of eight) with measured data for SBPPRL (Table 11).

2.4.2.2. Results assuming different dynamics in the first year for SBPPRL

Comparing Table 15 and Table 14, we see that the use of the first-year dummy significantly improves the adjusted R^2 of the estimation for the SBPPRL specific-data model (0.951 instead of 0.794). The adjusted R^2 of the estimations are equal for both models (0.970 and 0.969), while the rMSE for the pooled-data model is slightly lower (0.477 against 0.489).

Table 15 also shows a higher SOM input in SBPPRL than in FNG in specific-data models, using the first-year dummy. Parameters *K'* and *a* are higher for SBPPRL with t > 1. Unlike Table 14, this result is now consistent with field observations in Table 8, which has the stocking rates in each plot: SBPPRL produce more biomass, and thus support a stocking rate which is systematically twice or more that of natural pastures. However, while *K'* is significantly higher for SBPPRL than for FNG, *a* is very similar in the two systems, and lower for NG. This means that SOM pools respond to fertilization in a similar way (results for *a*), and the effect of sowing is only captured by the specific SOM input (*K'*). This result is supported by the fact that the seed mixes were designed specifically for each plot, and thus maximize SOM input regardless of baseline natural conditions. The difference in effect between both forms of input is another validation of the use of the initial SOM concentration as a variable in the model.

Regarding the absolute values of $K_i^{'}$ in Table 15, we may notice, for example, that $K_i^{'}$ in the pooled-data model (t > 1) is equal to 0.604 $\%.yr^{-1}$, which using Table 10 is equivalent to an input of 7.25-8.09 $t.ha^{-1}$ of SOM. Now, we need to transform this value of *K* into equivalent production. Dividing *K* by 80% and then by the average R:S of 2.8, we find that aboveground production is 3.2 – 3.6 $t.ha^{-1}$. Using two extreme R:S, aboveground production would be 1.9 – 2.1 $t.ha^{-1}$ or 18.1 – 20.2 $t.ha^{-1}$ The average production falls within the range of dry matter productivity of SBPPRL of 2 to 9 $t.ha^{-1}$ of dry matter (Carneiro *et al*. 2005).

The mineralization rates obtained in specific-data models for SBPPRL and FNG are similar, and are both higher than for NG. Increased SOM inputs due to the installation of SBPPRL and fertilization in FNG provide the soil labile forms of SOM, which are easily mineralized. Since there was no land use change or management activity during the sampling period in NG, their SOM pool remains stable.

Table 14 – Results of the estimation for pooled-data and specific-data models for grassland systems *i*.

Model	Grassland system	Using SOM_0?	Original data							Data with filled-in missing values						
			Adj. R^2	rMSE	K' SBPPRL	K' FNG	K' NG	α	*a*	Adj. R^2	rMSE	K' SBPPRL	K' FNG	K' NG	α	*a*
Pooled-data	All	No	0.952	0.677	0.422	0.171	0.136	-0.017		0.966	0.554	0.370	0.248	0.165	-0.034	
		Yes	0.956	0.659	0.500	0.303	0.128	0.403	0.630	0.969	0.489	0.415	0.289	0.109	0.267	0.430
Specific-data	SBPPRL	No	0.760	0.745	0.413			-0.020		0.794	0.649	0.353			-0.042	
		Yes	0.731	0.781	0.531			0.237	0.348	0.794	0.585	0.379			0.151	0.276
	FNG	No	0.810	0.622		0.428		0.071		0.886	0.470		0.442		0.043	
		Yes	0.841	0.603		1.083		1.105	1.434	0.912	0.397		0.508		0.443	0.566
	NG	No	0.899	0.615			-0.034	-0.105		0.935	0.487			0.011	-0.113	
		Yes	0.920	0.549			-0.282	0.512	1.073	0.943	0.438			-0.048	0.190	0.432

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; *K'*, α, *a* – Parameters in Equation (1.7); rMSE – root Mean Squared Error.

Table 15 – Results of the estimation for pooled-data and specific-data models for grassland systems *i*, including a first-year dummy in SBPPRL.

Model	Grassland system	Using SOM_0?	Filled data							
			Adj. R^2	rMSE	K' SBPPRL t=1	K' SBPPRL t>1	K' FNG	K' NG	α	*a*
Pooled-data	All	Yes	0.970	0.477	0.097	0.604	0.349	0.130	0.365	0.544
Specific-data	SBPPRL		0.951	0.610	0.135	0.697			0.429	0.588
	FNG		0.912	0.397			0.508		0.443	0.566
	NG		0.943	0.438				-0.048	0.190	0.432

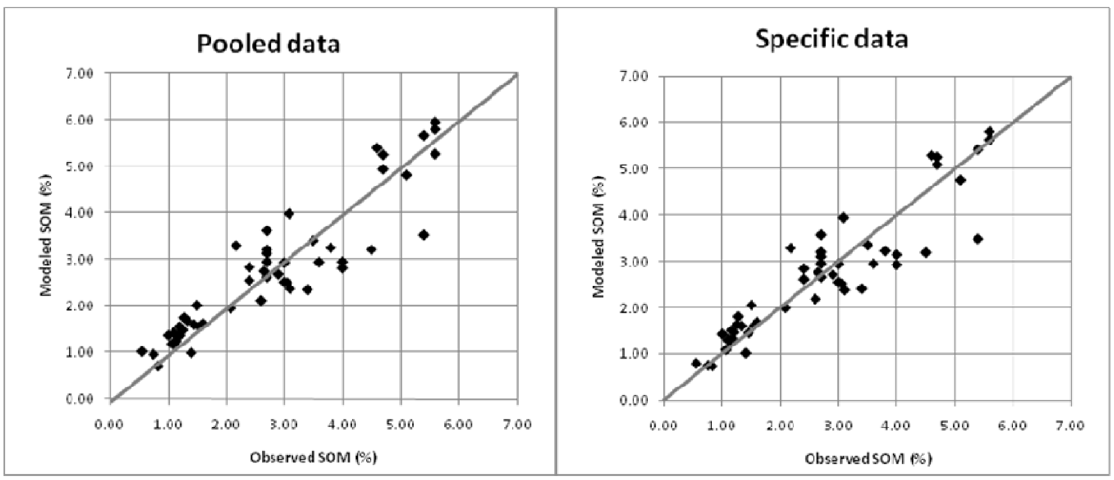
NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; *K'*, α, *a* – Parameters in Equation (1.7); rMSE – root Mean Squared Error.

2.4.3 Assessment of model quality

In order to verify the adjustment to the original data provided by the model, we applied it to each farm. We used parameters indicated in Table 16, and adjusted a model to each plot in each farm, using the plot-specific initial SOM concentration. The initial SOM concentration in each plot corresponds to the first column of data in Table 11.

Results are shown in Figure 14, which plots all observed and modelled results, except points corresponding to initial SOM concentration (which are by construction over the grey line) and except points which were filled-in. The closer that the points are to the grey 45º line, the better the fit.

Figure 14 – Observed and simulated SOM concentration for all farms and grassland systems, using an analytical model (on the left) and the linear approximation (on the right).



SOM – Soil Organic Matter.

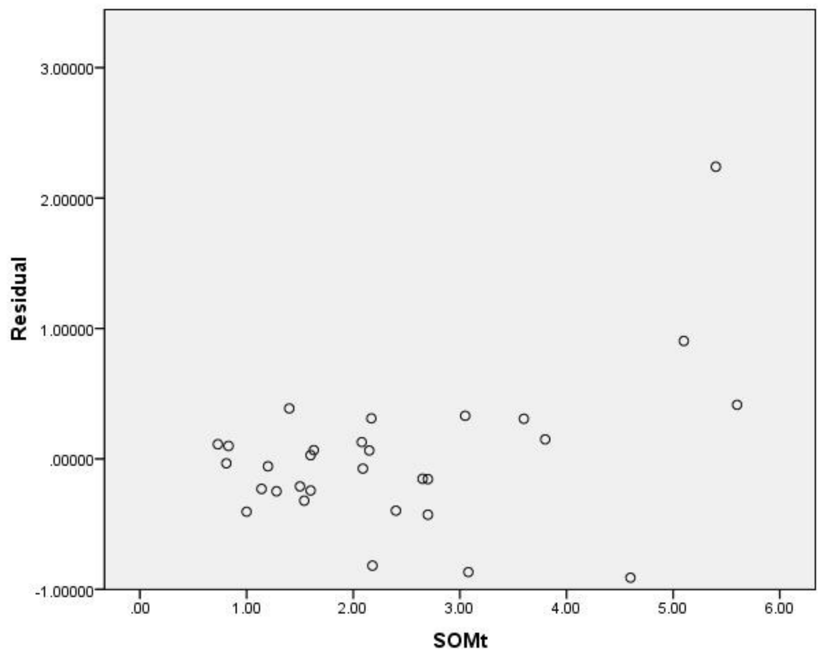
Visual inspection of Figure 14 does not show significant differences between the two models. However, in both cases, even though there seems to be no overall systematic bias, for SOM lower than 2.0 %, models seem to overestimate the values observed. In order to verify these hypotheses, namely that (1) there is no overall model bias and (2) there is a slight bias for SOM lower than 2.0%, we obtained two series, one for the results obtained with pooled data, and another for specific data. The series is calculated as the difference between modelled SOM and observed SOM.

A paired samples t-test for means of the two series (one for the results of each model) shows no evidence to reject the null hypothesis of equal means ($p < 0.05$). Therefore, the two models are equivalent, even though there is a slight tendency for the specific-data model to overestimate results. A paired samples t-tests comparing the respective series for each model with a series where all observations are zero also indicates that the means of both series cannot be rejected to be equal to zero ($p < 0.05$), and therefore there is no overall bias of any model.

To show that the bias does exist for SOM lower than 2.0%, we truncated the series data, separating each series into two more series: one for observations with SOM lower than 2.0%, and another one for SOM higher than 2.0%. Results were now different. The conclusions for observations with SOM higher than 2.0% are equal to the overall conclusions, and there is no bias ($p < 0.05$). However, in any model, we can reject that the series with observations with SOM lower than 2.0% has a zero mean value ($p > 0.05$). Models systematically overestimate lower SOM. This result, which is true for all pasture systems, means that the initial dynamics is not well captured by the asymptotic curve that we model. However, the deviation is relatively small. In fact, even though there is no bias, as we can see in Figure 14, variance is higher for higher SOM.

An alternative representation to Figure 14 may be found in Figure 15, which depicts the series of residuals (difference between estimated and observed values) as a function of SOM. For illustration purposes, only results for the specific model for SBPPRL are shown. A Phillips-Perron Test rejects the null hypothesis of a unit root ($p < 0.05$). This means that the series of residuals is stationary.

Figure 15 – Series of residuals as a function of SOM.



SOM – Soil Organic Matter.

Furthermore, a one series Kolmogorov-Smirnov Test of normality to the series of residuals shows that the null hypothesis of the normal distribution cannot be rejected (K-S Z value 0.928, asymptotic sigma 0.356). Therefore, there is no evidence of a spurious regression due to a bias in residuals.

2.4.4 Projections of average SOM increases and carbon flows

For illustration purposes, and due to the results stated before, we will use the parameters from the pooled and specific-data models, including SOM_0, the complete filled-in data table and using a first-year dummy for SBPPRL.

Table 16 shows results for K, which is a measure of the total input per year. In both formulations of the model, K is higher for SBPPRL than for FNG, which translates to higher SOM increases since the fraction of existing SOM in one year which is mineralized in the next, $1-e^{-\alpha}$, is constant in the pooled-data model and approximately equal in the specific-data model. Table 16 also shows that SBPPRL in the first-year have a much lower (but still positive) specific input than in the following years, but their total SOM input K is approximately equal to that of NG. Therefore, even in the first year, SBPPRL produce at least as much biomass as NG.

Table 16 – Models parameters for each grassland system.

<table>
<tr><th rowspan="3">Grassland system</th><th colspan="10">Parameters in analytical and linear models</th></tr>
<tr><th colspan="5">Pooled-data model</th><th colspan="5">Specific-data model</th></tr>
<tr><th>K'
$pp.yr^{-1}$</th><th>a
yr^{-1}</th><th>K (SOM_0= 0.87%)
$pp.yr^{-1}$</th><th>α
yr^{-1}</th><th>$1-e^{-\alpha}$
-</th><th>K'
$pp.yr^{-1}$</th><th>a
yr^{-1}</th><th>K (SOM_0= 0.87%)
$pp.yr^{-1}$</th><th>α
yr^{-1}</th><th>$1-e^{-\alpha}$
-</th></tr>
<tr><td>SBPPRL (t=1)</td><td>0.097</td><td rowspan="4">0.544</td><td>0.570</td><td rowspan="4">0.365</td><td rowspan="4">0.306</td><td>0.135</td><td rowspan="2">0.544</td><td>0.609</td><td rowspan="2">0.429</td><td rowspan="2">0.349</td></tr>
<tr><td>SBPPRL (t>1)</td><td>0.604</td><td>1.078</td><td>0.697</td><td>1.171</td></tr>
<tr><td>FNG</td><td>0.289</td><td>0.763</td><td>0.508</td><td>0.566</td><td>1.000</td><td>0.443</td><td>0.358</td></tr>
<tr><td>NG</td><td>0.109</td><td>0.583</td><td>-0.048</td><td>0.432</td><td>0.328</td><td>0.190</td><td>0.173</td></tr>
</table>

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; *K', K,* α*, a* – Parameters in Equation (1.7).

Using the parameters in Table 16, we determined the average SOM increase in 10 years from each grassland type shown in Table 17 and depicted in Figure 16 and Figure 17. Results show that SBPPRL on average increase their SOM concentration by 0.19-0.20 $pp.yr^{-1}$, which is equivalent to 1.33-1.40 t $C.ha^{-1}.yr^{-1}$ or 1.48-1.56 t $C.ha^{-1}.yr^{-1}$, depending on the soil density, using Table 10. This increase is higher than for FNG (0.13-0.14 $pp.yr^{-1}$, equivalent to 0.94-0.95 t $C.ha^{-1}.yr^{-1}$ or 1.05-1.07 t $C.ha^{-1}.yr^{-1}$, depending of soil density) and NG (0.07-0.08 $pp.yr^{-1}$, equivalent to 0.51-0.53 t $C.ha^{-1}.yr^{-1}$ or 0.57-0.59 t $C.ha^{-1}.yr^{-1}$, depending on soil density). The difference of the average increase in 10 years between the pooled-data and the specific-data models is lower than 0.01 pp per year or 0.08 t $C.ha^{-1}.yr^{-1}$ for all grassland systems, which shows that both models are very similar in the results obtained. Note that all plots were previously NG. Plots remaining NG were only tilled in the first year, and so results for NG must be read as post-tillage dynamics.

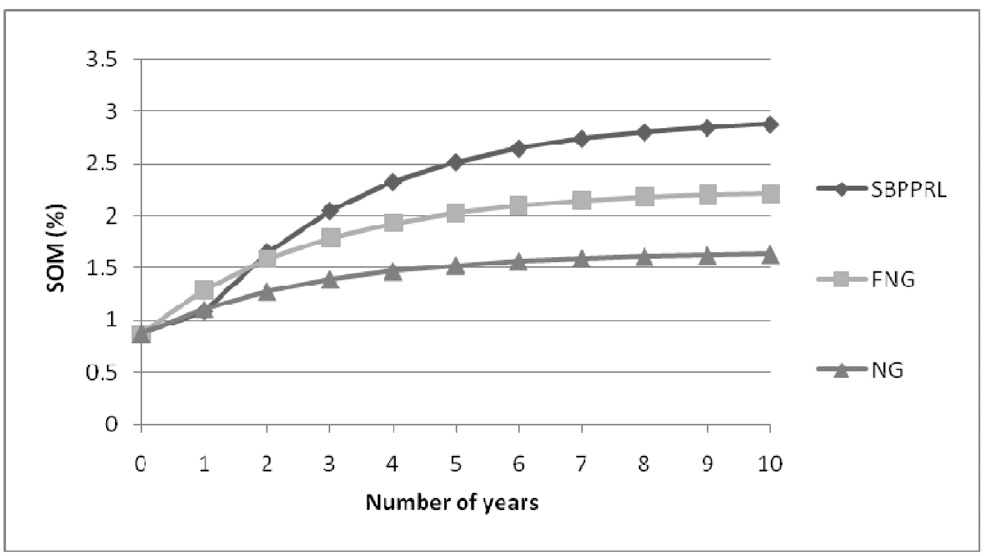
The intersection of the SOM increase curves for SBPPRL and FNG happens due to the first-year loss imposed by the dummy variable. Part of this loss is due to tillage, and so happens in the beginning (year 0). However, tillage has an effect on the mineralization rate. We could not estimate an α for the first year, since that would imply the use of a specific-data model for first-year observations, which are only eight. The dummy we use for the first-year in SBPPRL only affects the coefficient for K', which correctly depicts the decreased productivity due to the existence of less seeds, and therefore less biomass production, in the plot, which happens throughout the first year. Therefore, the first-year dummy also has to capture the effect of tillage, which should be instantaneous and only seen in α. The only way to do so is by spreading the SOM loss throughout the first-year.

Table 17 – Estimated SOM concentration per year, starting from SOM_0 = 0.87%.

<table>
<tr><th rowspan="3" colspan="3">Year</th><th colspan="6">Estimated SOM concentration (%)</th></tr>
<tr><th colspan="2">SBPPRL</th><th colspan="2">FNG</th><th colspan="2">NG</th></tr>
<tr><th>Pooled-data model</th><th>Specific-data model</th><th>Pooled-data model</th><th>Specific-data model</th><th>Pooled-data model</th><th>Specific-data model</th></tr>
<tr><td colspan="3">0</td><td>0.87</td><td>0.87</td><td>0.87</td><td>0.87</td><td>0.87</td><td>0.87</td></tr>
<tr><td colspan="3">1</td><td>1.08</td><td>1.29</td><td>1.11</td><td>1.09</td><td>1.37</td><td>1.02</td></tr>
<tr><td colspan="3">2</td><td>1.65</td><td>1.59</td><td>1.28</td><td>1.69</td><td>1.69</td><td>1.14</td></tr>
<tr><td colspan="3">3</td><td>2.05</td><td>1.79</td><td>1.39</td><td>2.09</td><td>1.89</td><td>1.24</td></tr>
<tr><td colspan="3">4</td><td>2.33</td><td>1.93</td><td>1.47</td><td>2.34</td><td>2.02</td><td>1.33</td></tr>
<tr><td colspan="3">5</td><td>2.52</td><td>2.03</td><td>1.53</td><td>2.51</td><td>2.11</td><td>1.40</td></tr>
<tr><td colspan="3">6</td><td>2.65</td><td>2.10</td><td>1.57</td><td>2.61</td><td>2.16</td><td>1.46</td></tr>
<tr><td colspan="3">7</td><td>2.74</td><td>2.14</td><td>1.59</td><td>2.68</td><td>2.20</td><td>1.50</td></tr>
<tr><td colspan="3">8</td><td>2.81</td><td>2.18</td><td>1.61</td><td>2.73</td><td>2.22</td><td>1.54</td></tr>
<tr><td colspan="3">9</td><td>2.85</td><td>2.20</td><td>1.62</td><td>2.76</td><td>2.23</td><td>1.58</td></tr>
<tr><td colspan="3">10</td><td>2.88</td><td>2.21</td><td>1.63</td><td>2.78</td><td>2.24</td><td>1.60</td></tr>
<tr><td rowspan="3">Average increase in 10 years</td><td colspan="2">$pp.yr^{-1}$</td><td>0.20</td><td>0.13</td><td>0.08</td><td>0.19</td><td>0.14</td><td>0.07</td></tr>
<tr><td rowspan="2">$t\ C.ha^{-1}.yr^{-1}$</td><td>MBD = 1.25 $g.cm^{-3}$</td><td>1.40</td><td>0.94</td><td>0.53</td><td>1.33</td><td>0.95</td><td>0.51</td></tr>
<tr><td>MBD = 1.40 $g.cm^{-3}$</td><td>1.56</td><td>1.05</td><td>0.59</td><td>1.48</td><td>1.07</td><td>0.57</td></tr>
</table>

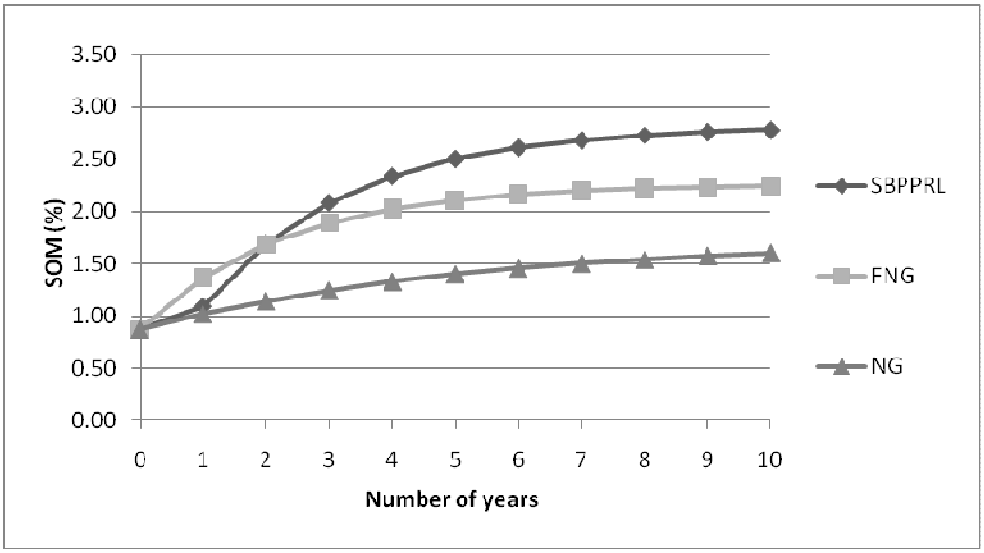
NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; pp – percentage points; MBD – Mineral Bulk Density.

Figure 16 – Simulated SOM concentration in each year, as estimated by the pooled-data model, using data filled-in using geometric averages, starting from 0.87%.



NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter concentration.

Figure 17 – Simulated SOM concentration in each year, as estimated by a specific-data model, using data filled-in using geometric averages, starting from 0.87%.



NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

2.5 Synthesis of results and discussion

Since SOM is the key parameter that determines other environmental services of pastures, in this Chapter we studied SOM dynamics in sown biodiverse and natural types of rainfed grasslands: SOM concentration is important for many agronomic and environmental reasons. We defined a model to calculate average SOM increases as a balance between accumulation of organic material in soils and mineralization of the available SOM pool. In time, as SOM increases, it eventually reaches an upper bound. Therefore, in a grassland system with no change in management, the entry and the mineralization eventually cancel each other. Other studies in literature indicate that SOM also increases asymptotically after land use changes (Sollins *et al.,* 1996; West and Six, 2007), as well as in the response to exogenous inputs of carbon or fertilizers (Six *et al.*, 2002; Six *et al.*, 2004).

Our model was calibrated using five years of soil analyses from eight locations. In each location there were two or three plots with two or three of the different grassland systems considered. We use statistical calibration to adjust an asymptotic curve to the data and obtain the model parameters. The parameters in our model were statistically determined using two methods which yielded a significant ($p < 0.10$) fit to the data. This is an important result, since we only had soil analyses for the first five years after installation (for one of them, 2002, there is practically no data). Considering that SOM saturation in soils is assessed in the long term (Stewart *et al.*, 2007), the fact that the asymptotic pattern is already picked up in the data is a strong conclusion.

We find that the expected steady-state long term SOM concentration in undisturbed SBPPRL is higher than in NG and FNG. In 10 years, there is an average increase of 0.19-0.20 percentage points per year in SBPPRL (equivalent to 1.33-1.40 t $C.ha^{-1}.yr^{-1}$ or 1.48-1.56 t $C.ha^{-1}.yr^{-1}$, depending on the soil density). In turn, SOM increases in FNG and NG are respectively 0.13-0.14 (0.94-0.95 t $C.ha^{-1}.yr^{-1}$ or 1.05-1.07 t $C.ha^{-1}.yr^{-1}$) and 0.07-0.08 percentage points per year (0.51-0.53 t $C.ha^{-1}.yr^{-1}$ or 0.57-0.59 t $C.ha^{-1}.yr^{-1}$). SBPPRL induces SOM increase due to sowing and fertilization, but mineralization rates are equal to those from FNG. NG have stable SOM pools, with lower input and mineralization rates.

Using a pooled-data model involves the estimation of fewer parameters, but requires one more assumption: that the mineralization rate is approximately independent of the grassland type. This hypothesis was validated for SBPPRL and FNG due to the fact that using a specific-data model also results in equal mineralization rates for both systems. Since there was no land use change in NG, SOM pools are already stabilized, and thus both the input of SOM and the mineralization rate were lower. The adjustments to the data of both the pooled and specific-data models are also equally good. Therefore, results are consistent despite the use of alternative estimation methods.

The parameter α is negative for the pooled-data model and in the specific-data models for SBPPRL and NG, when the initial SOM concentration is not used. There are three alternative explanations for why the model in these situations is adjusting a non-saturating exponential curve (positive second derivative) to the data.

The first explanation is that the parameter K is also a function of SOM concentration. The base model in Equation (1.4) would then be, for example:

$$\frac{dSOM}{dt} = K' + kSOM - \alpha SOM \,. \tag{1.12}$$

In this case, the parameter related to the SOM term, which is then estimated in the regressions, is not the mineralization rate: it is the mineralization rate minus the coefficient for the variable SOM input. Therefore, when $\alpha - k < 0$, the model would be merely stating that the variable SOM input is higher than the mineralization rate. Note that it would not be possible to include two variables in the model instead of just α, because then we would not be able to estimate the parameters in the regression.

The second explanation is that, in SBPPRL, SOM increases with the consolidation of the installation of the pasture. However, since in this model SOM always increases with time, it becomes a proxy for the number of years since the installation of the pasture. Therefore, there is a time-dependent effect on K, which is indistinguishable in practice from the one mentioned before, and so Equation (1.12) also states this approach.

However, if these hypotheses were correct, then it would be an effect that happens mainly (in the first case) or exclusively (in the second) in SBPPRL. The results, however, do not confirm it. For the specific-data models, α is always lower (and sometimes negative) in SBPPRL, but there is also a negative α for NG.

Therefore, the explanation is most likely that the model requires the initial SOM concentration. Table 14 shows that when SOM_0 is considered α is always positive. If more productive soils (those that begin with a higher initial SOM concentration) will increase their SOM level more regardless of the grassland system, then our model cannot cope with that exogenous fact unless it considers a negative mineralization rate. This third explanation is the one which led us to use the initial SOM concentration as an independent variable.

In fact, even if the first two explanations are true, SOM_0 is in fact capturing some of the effect mentioned. We can also see in Table 14 that the introduction of the initial SOM concentration as an independent variable in the model increases the values of K and α. This is due to the fact that part of the supposed variable effect of K (which we know from the first and the second explanations) is captured by the initial SOM concentration, thus increasing α. The increase in α is then balanced by the increase in the fixed part of K, so that the same long term equilibrium is met.

The use of the initial SOM concentration slightly increases the adjusted R^2, when taking the increased number of degrees of freedom into account. In the short term, and given the rationale stated above, a negative α could only be stating a transitory compensation of mineralization by the specific input. But in the long run this effect eventually plays out and a steady state is reached. Since we are using data obtained during five years, but wish to extrapolate for a longer period, the use of the initial SOM concentration stabilizes the model, translating the long term dynamics.The main objective of the model was to obtain a long-term trend for SOM dynamics in the different grassland systems. We did not wish to obtain a model which predicts year-by-year SOM concentration, but rather to show the average SOM accumulation potential of each system. Therefore, the main conclusions of this study are that SBPPRL provide soils an increased organic matter pool. Starting from an arbitrary 0.87% SOM initial value, in 10 years there is an average increase of 0.19-0.20 $pp.yr^{-1}$ in SBPPRL. SBPPRL increase their SOM concentration more than other grassland systems. SOM increases in FNG and NG are respectively 0.13-0.14 and 0.07-0.08 $pp.yr^{1}$.

The increased input in SBPPRL is due to two factors. First, production responds to fertilization, and this response seems to depend only on soil intrinsic quality, and so it is a similar effect for SBPPRL and FNG. Second, production responds to the improved seed bank independently of soil characteristics. It is important to notice that SBPPRL in the first-year were tilled and have a small seed bank (the seeds sown). This productivity loss effect in the first-year is needed to interpret results.

The actual difference between SBPPRL and natural pastures may be even greater than we show here. FNG and NG may be overestimated due to the fact that plots were contiguous, since Carneiro *et al.* (2005) explain that there was some contamination of natural grasslands by sown species. Note also that, even though we used 0.87% as the arbitrary starting SOM, Table 11 shows that few farms had such low initial SOM concentrations. If initial SOM concentration is higher, and since a is positive (and higher for SBPPRL), then yearly increases are higher as well.

To our knowledge, there are no other internationally published studies on SBPPRL in Portugal or elsewhere. We find that our results are similar to those found in other preliminary Portuguese studies. Some early results hint that in 10 years SBPPRL increase SOM from 1 to 3% (Crespo, 2004). At Herdade dos Esquerdos, in Vaiamonte (Portalegre, Portugal), following a programme of SBPPRL installation, SOM concentration across the farm increased from between 0.7% and 1.2% in 1979 to between 1.45% and 4.40% in 2003 (Crespo *et al.*, 2004; Crespo, 2006a,b). This SOM increase is higher than that of any natural grassland under any form of management found in the literature.

In a related study, Aires *et al.* (2008) measured carbon fluxes over a pasture in Southern Portugal, similar to natural pastures in our study. They found that, in 2004-2005 (drought year), pastures emitted 0.49 t $C.ha^{-1}.yr^{-1}$, while in 2005-2006 (normal precipitation year) they sequestered 1.91 t $C.ha^{-1}.yr^{-1}$. This high intra-annual variability is also captured by our results, since yearly measurements of SOM concentration oscillates around a medium/long-term increasing trend (the trend is given by our model).

Regarding the first year in Aires *et al.*'s (2008) study, the drought year, it was also a sampling year in our study. Table 11 shows that in FNG and NG there was also a decrease in SOM from 2004 to 2005, which is the period for which Aires *et al.* (2008) concluded that soils had been emitters. FNG lost, on average, 0.08 pp, and NG lost 0.49 pp. These values are equivalent to a loss of 0.56 – 0.62 t $C.ha^{-1}.yr^{-1}$, and 3.41 – 3.81 t $C.ha^{-1}.yr^{-1}$, for FNG and NG respectively. When comparing these results to the results of Aires *et al*. (2008), we see that our approach does not under-estimate emissions.

Still regarding the same period, it is interesting to notice that SBPPRL increased SOM concentration even in the drought year by 0.97 pp on average (Table 11). This indicates that SBPPRL have one further advantage, which is increased resilience in drought years.

Regarding the second sampling year in Aires *et al*. (2008), it was a normal precipitation year. Table 15 shows that the lowest average SOM increase in our study is for NG (specific-data model), and is 0.07 pp per year, and the highest result is 0.20 pp per year for SBPPRL (specific-data model). These values correspond, respectively, to 0.48 – 0.54 t $C.ha^{-1}.yr^{-1}$, and 1.39 – 1.54 t $C.ha^{-1}.yr^{-1}$ (two values provided, depending on soil density).

The similarity in results between our work and Aires *et al.*'s (2008) work also indicates consistency between our method, using soil samples, and theirs, using flux measurements.

In the next chapter, we turn to the quantification of the environmental services provided by SBPPRL.

3. Quantifying the environmental services provided by SBPPRL

Chapter 3 deals with the quantification of some environmental effects mentioned in Chapter 1, starting from the quantification of SOM dynamics. We namely determine the carbon balance of NG and SBPPRL systems, including the emissions from animals, legumes and liming. We compare rainfed SBPPRL to other agricultural land uses and practices, to determine which has the highest potential to sequester carbon. We also compare how rainfed SBPPRL compare to their alternative, which in this case is not NG but maize used for biofuel production.

Then, we perform an LCA of the systems, and finally we calculate soil loss in both systems, and refer studies on biodiversity. By doing so, we determine whether SBPPRL are worse that NG in any other environmental impact theme.

3.1 From SOM accumulation to carbon sequestration

3.1.1 Equivalency factors

In Section 2.3.4, we calculated the C sequestration equivalent to an increase in 1% SOM. To convert those increases from C to CO_2, we multiply that value for the molecular weight of CO_2 (44) and divide it by the atomic weight of carbon (12).

Table 18 shows final results for carbon sequestration in t $CO_2.ha^{-1}$ equivalent to 1% increase in SOM (Teixeira, 2008, 2010b).

Table 18 - Carbon sequestration equivalent to the increase in SOM of 1 pp in 10, 20 and 30 cm.

Depth (cm)	MBD ($g.cm^{-3}$)	t $C.ha^{-1}$	t $CO_2.ha^{-1}$
10	1.25	6.96	25.5
10	1.40	7.75	28.4
20	1.25	13.9	51.0
20	1.40	15.5	56.8
30	1.25	20.9	76.6
30	1.40	23.3	85.3

MBD – Mineral Bulk Density; C – carbon; CO_2 – carbon dioxide; SOM – Soil Organic Matter; pp – percentage points.

3.1.2 Which value to use?

Based on these values, and since soil samples for pastures were collected at 10 cm, the SOM increase of 0.20 percent points per year in SBPPRL estimated in the last chapter corresponds to the sequestration of 5.10 or 5.68 t $CO_2.ha^{-1}.yr^{-1}$, depending on the soil density we consider.

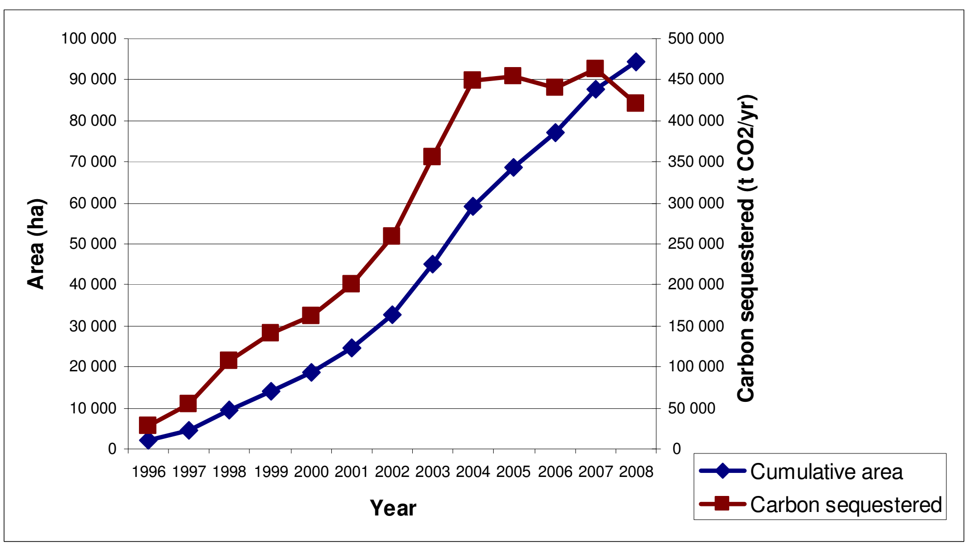
Therefore, the question arises: which soil density should we consider if no information is available? If pastures were sown using tillage, then it is very likely that during the first years after installation the MBD is close to the national average. However, we are considering as the relevant soil layer only the first 10 cm of topsoil. This is the layer which is more affected by interactions with the aerial system, and the one whose density varies the most easily. Therefore, as time goes by, livestock stomping and the lack of mobilization will undoubtfully create compactation in this upper layer of soil, increasing its MBD. If pastures are sown using no-tillage, this effect is even more dramatic.

Whenever we need to use a value throughout this thesis, we opted to use the round number of 5.0 t $CO_2.ha^{-1}.yr^{-1}$. This number is merely indicative; it is not the average of sequestration for the range of possible density values, or even an choice regarding the density we believe to be representative in the country.

3.1.3 How much carbon have SBPPRL sequestered lately?

Using sales data from Fertiprado, shown in Section 1.4.4 (João Paulo Crespo, personal communication), we calculated how the service of carbon sequestration has evolved in the past years. The rate of sales had been increasing since the beginning of the firm, but then started decreasing since 2004. Since SBPPRL sequester more carbon in the first years after the installation, and the area of first-year SBPPRL has decreased, the carbon sequestration service has remained relatively constant since 2004, as shown in Figure 18. To obtain Figure 18, we considered the specific yearly increases for each installation year. Overall, SBPPRL have sequestered around 3.5 Mt CO_2 from 1996 to 2008[31].

Figure 18 – Evolution of the environmental service of carbon sequestration provided by SBPPRL.



[31] Note that, in these calculations, we used the whole area sold by Fertiprado, which is around 70 000 ha. PNAC (2007) has a more modest estimate of 70 000 ha of SBPPRL in Portugal.

3.2 Calculating the carbon balances of pastures

3.2.1 Can we really expect high SOM sequestration in pastures?

To answer to that preliminary question, we turn to the literature. The use of grassland management as a carbon sink is well documented. Net Primary Productivity (NPP) and Net Ecosystem Productivity (NEP) in pastures are positive saturating functions of plant species and functional diversity (Catovsky *et al.*, 2002). NEP is measured by above and belowground production. Belowground production (roots), when incorporated into soils, corresponds to carbon storage. Therefore, some kinds of pasture may have high sequestration potentials. Several studies, such as Soussana *et al.* (2007) and Byrne *et al.* (2007), have tried to determine a correct number for the carbon sequestration potential via grassland management worldwide. Table 19 is a review of available data.

Freibauer *et al.* (2004) reviewed the potential for European soils to sequester carbon. They also evaluated suitable land for carbon farming. In their survey, they do not refer improved grasslands, but they indicate global grassland potential (with grazing management) as 0.8-2.6 t $CO_2.ha^{-1}.yr^{-1}$. Frank (2002) measured CO_2 flux over a grazed wheatgrass pasture during growing season as 2.18 t $CO_2.ha^{-1}.yr^{-1}$. Higher values were found by Smith (2004) – 4.4-6.2 t $CO_2.ha^{-1}.yr^{-1}$ – and Tschakert (2004) – 5.35 t $CO_2.ha^{-1}.yr^{-1}$. Both studies refer to the conversion of cropland to grassland, but the first study results from a survey, and the second from the application of a biogeochemical model.

As for the effect of increased grazing, Reeder and Schuman (2002) used 12 year data for mixed grass and 56 year data for short grass rangeland to conclude that grazed land has higher soil organic carbon than non-grazed areas. Their conclusions are supported by the bulk of available literature. For example, Freibauer *et al.* (2004) state a value for the increase of carbon sequestration by grazing management of 0.8-2.6 t $CO_2.ha^{-1}.yr^{-1}$.

Table 19 – Literature review for the potential of cropland and grassland soils to sequester carbon

Land Use	CO_2 sequestration (t CO_2 $.ha^{-1}.yr^{-1}$)	Method	Reference
Grassland	0.4-11.1	Survey	Conant *et al.*, 2001
Organic input on arable land	1.0-3.0	Survey	ECCP, 2003
Revegetation of set-aside land and introduction of perennials	2.0-7.0	Survey	ECCP, 2003
Promote organic farming	0-2.0	Survey	ECCP, 2003
Water table in peatland	5.0-15.0	Survey	ECCP, 2003
Temperate grassland	2.2 - 2.8	Flux Measurement	Frank, 2002
Eliminate bare fallow	0.6-2.8	Survey	Freibauer *et al.*, 2004
Grassland grazing management	0.8-2.6	Survey	Freibauer *et al.*, 2004
Grassland and pastures	0.18-0.37	Survey	Freibauer *et al.*, 2004
Grassland	0.62	Measurement	McLauchlan *et al.*, 2006
Semi-natural grassland	0.7 - 1.0	Modelling	Sindhoj *et al.*, 2006
Convert cropland to grassland	4.4 – 6.2	Survey	Smith, 2004
Convert cropland to grassland	5.4	Modelling	Tschakert, 2004
Enhancing rotation complexity	0.07 +- 0.04	Survey	West and Post, 2002

However, and regarding the SBPPRL, there were hints that suggested the possibility that the potential for carbon sequestration was even higher.

To understand what the balance means, we must first model them.

3.2.2 Overall C and N models of pastures

As noted before, we used field data collected from 2001 to 2005 (Table 11, location in Figure 12) in several locations in Portugal, during Projects AGRO 87 and AGRO 71. We will, then, focus on results from those projects, subjected to the experimental setting.

The model of carbon (C) and nitrogen (N) cycle processes contributing to the GHG balance in grasslands can be read as follows. Legumes and grasses grow in consociation using atmospheric CO_2. Symbiotic associations of legumes and microorganisms, namely *Rhizobium* (Bot and Benitez, 2005), fix atmospheric nitrogen as well. Belowground, a complex set of reactions between plant (roots), soil mineral particles, microorganisms and macrofauna (earthworms, etc.) takes place (Ostle *et al.*, 2009).

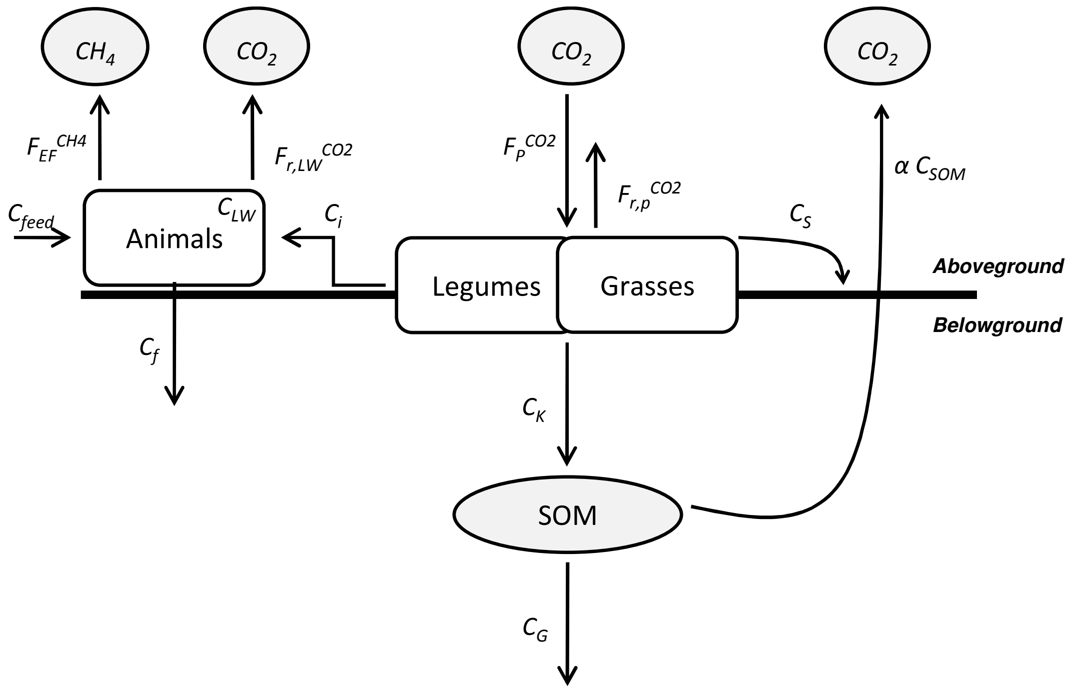
Three important outputs, in terms of GHG come out of the sum of these processes: SOM accumulation in soils, leaching of SOM particles and dissolved N, and emission of N_2O to the atmosphere. The aboveground part of the plants is eaten (on site or off-site) by animals, and be complemented by feeds (C and N import). Either through the process of enteric fermentation or from the degradation of wastes, there is some emission of CH_4 and N_2O. Finally, there is application of limestone to increase pH, and there is a corresponding emission of CO_2.

In these grassland systems there is no off-site digestion by animals (only direct grazing), and there are no applications of organic fertilizers (for a discussion on the implications of both fluxes on the global balance, see Soussana *et al.*, 2007).

3.2.2.1. Carbon balance

The carbon balance of pastures can thus be defined as the overall balance from several sub-systems, namely the soil system, the plant system and the animal system. All of these exchange greenhouse gases (GHG) with the atmosphere. The overall carbon balance is shown in Figure 19.

Figure 19 – Carbon balance in the grassland systems.



C – Carbon; CH_4 – methane; CO_2 – Carbon dioxide; F – flux; SOM – Soil Organic Matter; P – Photosynthesis; r – respiration; S – Senescence; SOM – Soil Organic Matter; LW – Live weight; G – Groundwater; α – mineralization rate; f – faeces or manure; i – ingestion; K – organic matter input to soil from roots.

The carbon balance of the plant system (all variables in in kg C ha^{-1} yr^{-1}) is:

$$F_P^{CO_2} - F_{r,plant}^{CO_2} = P_a^C + P_b^C \ . \qquad (3.1)$$

This means that the flux of CO_2 sequestered by the plant through photosynthesis (F_P^C) minus the plant respiration ($F_{r,plant}^{CO_2}$) will incorporate the total plant biomass production (P_t^C), which is equal to the sum of above (P_a^C) and belowground (P_b^C) biomass. Total production is separated in above and belowground according to the root to shoot (R:S) ratio of pasture plants (equal to P_b^C / P_a^C). Therefore, the aboveground production is equal to:

$$\frac{1}{1+R:S}\ P_t^C = C_S + C_i\ , \qquad (3.2)$$

where C_S is input (output from plant) of biomass to the soil by leave senescence, and C_i is ingestion by livestock. Under correctly managed grazing, around 30% of plant biomass escapes the intake by animals (Mazzanti *et al.,* 1994, cit. in Sanaulluah *et al.* 2009), and therefore $C_S = 0.30\ P_a^C$, which means that $1/R:S\ P_t^C = (1\text{-}0.30)^{-1}\ C_i$. Intake by animals is, on average, C_i = 4.8 kg C.LU^{-1}.day^{-1} (Thornley, 1998, cit. in Soussana *et al.*, 2007). Assuming that livestock grazes half of the year (during Spring and Autumn, even though the intake is highly variable), then C_i = 0.88 t C.LU^{-1}.yr^{-1}, or converting to equivalent CO_2, C_i = 3.2 t CO_2e.LU^{-1}.yr^{-1}. Furthermore, the IPCC (1997) indicates 2.8 as the default root to shoot ratio (R:S) for semi-arid grasslands. This value is consistent with the R:S of 0.5 to 4.8 in grazed pastures, which is the range of the comprehensive data for several regions gathered by Coupland (1976).

Belowground production will be equal to:

$$\frac{R:S}{1+R:S}\, P_t^C = K_{roots} + C_G\,, \tag{3.3}$$

where K_{roots} is SOM input due to soil processes of humus formation from roots, and C_G is emissions from leached organic matter to groundwater involved in leaching. In this section, we consider $C_G = 0$.

Therefore, the soil stock will in turn be:

$$C_K - \alpha\, C_{SOM} = K_{roots} + C_S + C_f\,, \tag{3.4}$$

where $C_K - \alpha\, C_{SOM}$ is the balance between the total carbon input (from roots, leave senescence and faeces) and the mineralized fraction of the existing SOM pool, and C_f is input from livestock faeces. According to APA (2009), daily excretion by cows is 2.79 $kg_{dry\ matter}$.LU^{-1}.day^{-1}. Considering that livestock grazes half of the year, and assuming that 58% of the excreted dry matter if C, then C_f = 1.8 t CO_2e.LU^{-1}.yr^{-1}.

Regarding the livestock balance, it may be put as:

$$C_i + C_{feed} = F_{r,LW}^{CO_2} + C_{LW} + F_{EF}^{CH_4} + C_f\,, \tag{3.5}$$

where C_{feed} is concentrated feed intake, $F_{r,LW}^{CO_2}$ is livestock respiration (which, according to Soussana *et al.*, 2007, is around 1 t CO_2.LU^{-1}.yr^{-1}), C_{LW} is net C incorporated in the animal live weight exported from the field, and $F_{EF}^{CH_4}$ are carbon losses from CH_4 emissions from enteric fermentation. We assume that C_{feed} is negligible and do not consider it here (as for the N balance).

Breeding cows are kept in pastures but each cow has one steer per year, which is typically removed at birth or after six months. In the second case, each 6 month-old steer weighs around 250 kg. 0 - 20% of this weight is carbon, and therefore the C exported, translated into CO_2e, is at most C_{LW} = 0.18 t CO_2e.LU^{-1}.yr^{-1}. We considered this figure negligible, and therefore removed it from calculations.

The overall balance, from the sum of all the sub-systems is:

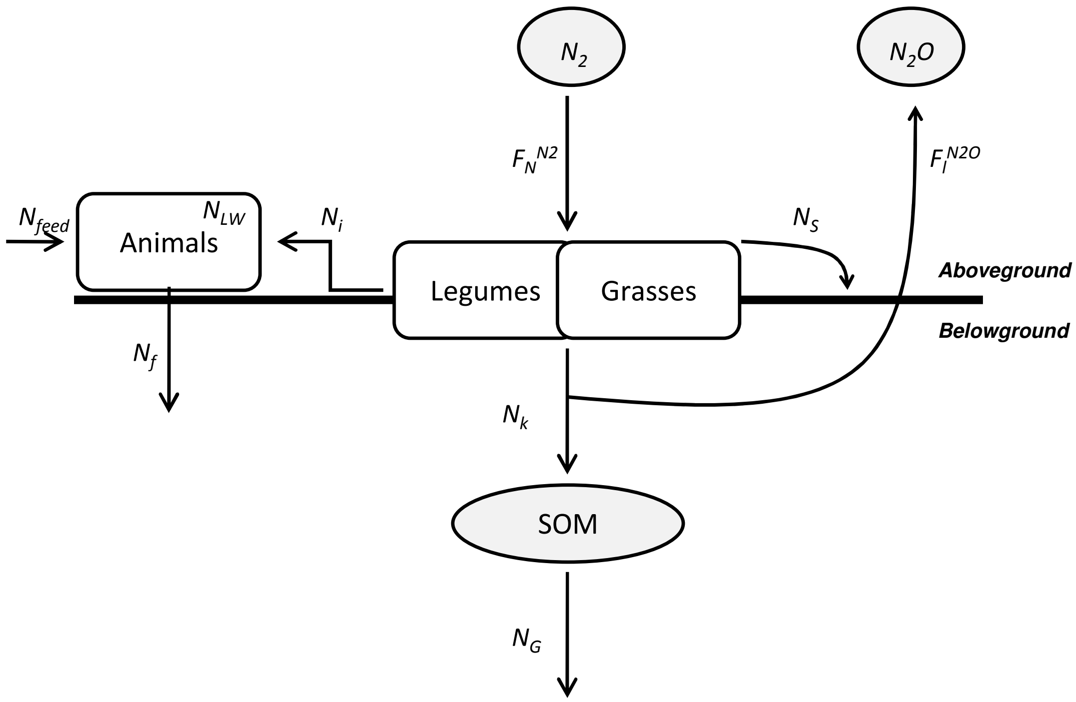
$$F_P^{CO_2} + C_{feed} = C_K - \alpha\, C_{SOM} + C_{LW} + C_G + F_{r,plant}^{CO_2} + F_{r,LW}^{CO_2} + F_{EF}^{CH_4}\,. \tag{3.6}$$

This means that the overall carbon inputs to the system are sequestration of C during photosynthesis by plants and (possible) introduction via feeds. This carbon is either stored as SOM (balance between entry and mineralized fraction), exported as animal live weight, lost to groundwater, emitted from respiration by plants and livestock, or is emitted by enteric fermentation. Note that $P_t^C = F_P^{CO_2} - F_{r,plant}^{CO_2}$, and therefore the two terms may be estimated together measuring total biomass production.. Note also that C_{feed}, C_{LW} and C_G are equal to zero.

3.2.2.2. Nitrogen balance

The nitrogen balance can also be defined according to the three sub-systems defined for the carbon balance, as shown in Figure 20.

Figure 20 – Nitrogen balance in the grassland systems.



N – Nitrogen; N_2O – nitrous oxide; N2 –Nitrogen gas; F – flux; SOM – Soil Organic Matter; P – Photosynthsis; l – legumes; S – Senescence; SOM – Soil Organic Matter; LW – Live weight; G – Groundwater; α – mineralization rate; f – faeces or manure; i - ingestion; k – organic matter input to soil from roots.

In this case, there is an N_2 input due to legumes (instead of CO_2), and so the plant sub-system may be defined as (all variables in in kg N ha^{-1} yr^{-1}):

$$F_l^{N_2} = P_a^N + P_b^N . \tag{3.7}$$

This means that the flux of N_2 sequestered by legumes ($F_l^{N_2}$) will be incorporated either its above (P_a^N) or belowground (P_b^N) biomass, the sum of which is N in the total biomass (P_t^N). As for carbon, total production is separated in above and belowground according to the root to shoot (R:S) ratio of pasture plants. Therefore, the aboveground part is equal to:

$$\frac{1}{1+R:S} P_t^N = N_S + N_i , \tag{3.8}$$

where N_S is nitrogen in input (output from plant) of biomass to the soil by leaves' senescence, and N_i is N ingestion by livestock. Analogously to the C balance, $P_a^N = (1\text{-}0.30)^{-1} N_i$, due to the relation between production and senescence.

Belowground production will be equal to:

$$\frac{\text{R:S}}{1+\text{R:S}}\, P_t^N = N_K + N_G\,, \tag{3.9}$$

where N_K is the N incorporated in the SOM stock due to soil processes of humus formation from roots and N_G is N lost from leaching. We consider $N_G \approx 0$ for both grassland systems: for NG, because there are few legumes and no N-fertilization; for SBPPRL, because experience shows that practically no N is leached. An unpublished study (Rodrigues *et al.*, 2010) from the Mountain Research Centre of the Superior Agronomic School in Bragança (Northern Portugal) has determined that soil inorganic-N levels are low in both grassland systems, and there were no significant risks of nitrogen losses (via nitrate leaching and nitrification and NH_3 volatilization).

Therefore, the soil stock will in turn be:

$$\Delta N_{SOM} = N_K + N_S + N_f + F_l^{N_2O}\,, \tag{3.10}$$

where N_{SOM} is N in the SOM pool, N_f is input from livestock faeces and $F_l^{N_2O}$ is the part of soil N which is emitted as N_2O during the complex nitrification/denitrification processes that occur in the soils. Regarding the livestock balance, it may be put as:

$$N_i + N_{feed} = N_{LW} + N_f\,, \tag{3.11}$$

where N_{feed} is the N input in concentrated feeds, and N_{LW} is N exported in the animal live weight. As in the C balance, one steer is exported per year, 0 - 2% of which is N, and therefore N_{LW} = 5 kg $N.LU^{-1}.yr^{-1}$. We also assume that this figure is negligible, and therefore removed it from calculations.

The overall balance, from the sum of all the sub-systems is:

$$F_l^{N_2} + N_{feed} = \Delta N_{SOM} + N_{LW} + N_G + F_l^{N_2O}\,. \tag{3.12}$$

This means that nitrogen fixed by legumes is used by legumes themselves and also by grasses to increase biomass. Some feed N may also enter the system. These N inputs are either incorporated in the soil SOM, exported as live weight, re-emitted to the soil as faeces or emitted as N_2O to the atmosphere. Note that $F_l^{N_2} = P_t^N$, and therefore the flux may be estimated using N in total biomass production.. Note also that N_{feed}, N_{LW} and N_G are equal to zero.

3.2.2.3. Net greenhouse gas balance

The net greenhouse gas balance (NGHGB) is the sum of the contribution from all GHG weighted by their global warming potential (GWP) in a 100-year time horizon (Forster *et al.*, 2007):

$$\text{NGHGB} = GWP_{CO_2}F^{CO_2} + GWP_{CH_4}F^{CH_4} + GWP_{N_2O}F^{N_2O}\,, \tag{3.13}$$

where $GWP_{CO_2} = 1$, $GWP_{CH4} = 25$ and $GWP_{N_2O} = 298$. This balance is equal to the sum of the stock and GHG flux terms in the C and N balances.

We consider that CO_2 sequestration is equal to the only stock term, which is SOM increase through carbon sequestration (already shown). Carbon stocked in soil pools is the only immobilized form of carbon in the system, and thus is the only effective sink that can be measured. There is, however, C exportation in steers and CO_2 emissions from livestock. There are also CO_2 emissions that do not enter the overall plant-soil-atmosphere system, namely carbon emitted due to the application of limestone for acidity correction (C_{lime}). Net CO_2 emissions are thus equal to:

$$F^{CO_2} = \Delta C_{SOM} + C_{LW} + F^{CO_2}_{r,LW} + C_{lime}, \tag{3.14}$$

Optimum conditions for legume production require not-too-low pH. When soil pH(H_2O) is lower than 5.3, which happens in 20 to 30% of Portuguese soils (David Crespo, personal communication), $CaCO_3$ or $CaMg(CO_3)_2$, depending on the type of limestone used, are applied to SBPPRL. During this operation, limestone is applied on the surface (except when the pasture is installed). After reacting with the soil particles, the cations are incorporated in the soil structure and CO_2 is emitted. The emission factor, attending the stoichiometry of the substances, is around 0.12 kg CO_2 kg^{-1} limestone applied (IPCC, 2003). We assume that SBPPRL require the application of 2 t $CaCO_3$ in the installation and then every 4 years (year 4 and year 8 after installation).

The only source of CH_4 is livestock, via enteric fermentation. Therefore, $F^{CH_4} = F^{CH_4}_{EF}$. There is one further emission of N_2O from soil processes associated with nitrification/denitrification cycles from bacterial activities related to N fixation by legumes. Therefore, $F^{N_2O} = F^{N_2O}_{l}$.

3.2.2.4. <u>CH_4 emissions</u>

The overall balance of emissions from animals, in a general situation, ($F_{livestock}$ in kg CO_2e ha^{-1} yr^{-1}) is

$$E_{livestock} = GWP_{CH_4} F^{CH_4}_{faeces/manure} + GWP_{CH_4} F^{CH_4}_{EF} + GWP_{N_2O} F^{N_2O}_{faeces/manure}, \tag{3.15}$$

where $F_{faeces/manure,CH_4}$, $F_{faeces/manure,N_2O}$ and F_{EF,CH_4} are, respectively, CH_4 and N_2O emissions from faeces or manure, and CH_4 emissions from the process of enteric fermentation (EF). Each of these emissions are calculated as

$$F_{source,GHG} = k_{GHG}\ SR, \tag{3.16}$$

where k_{GHG} is the emission rate of each GHG (CH_4 or N_2O), measured in kg GHG $head^{-1}$ yr^{-1}; SR is the average annual stocking rate (head ha^{-1}).

Values for k_{GHG} are obtained from the Portuguese National Inventory Report (NIR) for 2007 (APA, 2009), as shown in Table 20. Manure is only produced in stables, where aerating conditions are poor and thus digestion of waste is anaerobic. Grazing livestock produce faeces which are well aerated. Therefore, as noted before, the only CH_4 emissions from livestock are due to enteric fermentation.

Table 20 – Emission factors for livestock sources.

	Emission factor ($kg.head^{-1}.yr^{-1}$)						LU ($head\ ha^{-1}$)
	EF		Manure		Faeces		
Animal type	CH_4	N_20	CH_4	N_20	CH_4	N_20	
Non-dairy cattle	56.10	0.00	1.60	0.96	0.00	0.00	1.00
Steers	33.66	0.00	0.96	0.58	0.00	0.00	0.60

EF – Enteric Fermentation; CH_4 – methane; N_2O – nitrous oxide; LU – Livestock Unit.
Source: APA (2009).

Field results for stocking rates were already shown in Section 2.3.1 (Carneiro *et al.*, 2005). Measured average stocking rates for NG and SBPPRL are, respectively, 0.42 and 1.03 $LU.ha^{-1}$. For simplicity purposes, we assume round figures of 0.5 and 1.0 $LU.ha^{-1}$ in calculations.

3.2.2.5. N_2O emissions

There are basically two ways in the literature to rapidly calculate nitrogen emissions from legumes. The first uses an emission factor per fixed nitrogen unit. The second uses a factor per production of dry matter.

IPCC (1997) considers an N_2O emission factor from legumes of 0.0125 $kgN_2O\text{-}N.kg^{-1}$ fixed N. Carneiro *et al.* (2005) state for several locations values for fixed N ranging from 100 (Coruche) to 300 $kg.ha^{-1}$ (Cercal). The average is about 180 $kg.ha^{-1}$. Therefore, emissions would be 1.25 - 3.75 kg $N_2O\text{-}N.ha^{-1}$, or 0.4 – 1.2 t $CO_2e.ha^{-1}$, considering the GWP of N_2O. The average would be about 0.7 t $CO_2e.ha^{-1}$

An alternative calculation may be done considering that sown grasslands have higher dry matter (DM) productivity. According to Carneiro *et al.* (2005), productivity varies from 2 000 kg $DM.ha^{-1}$ (Coruche, Portugal) to 9 000 kg $DM.ha^{-1}$ (Quinta da França, Portugal). On average, about 60% of such production is due to legumes (Carneiro *et al.,* 2005). Therefore, and considering an emission factor of 0.001 kg $N_2O\text{-}N.kg^{-1}$ DM, emissions would range from 1.2 to 5.4 kg $N_2O\text{-}N.ha^{-1}$, or 0.3 to 1.5 t $CO_2e.ha^{-1}$, with an average of 0.9 t $CO_2e.ha^{-1}$.

In this section we use the average of the two average values, which is 0.8 t $CO_2e.ha^{-1}$.

3.2.2.6. Biomass production in SBPPRL

Results obtained may be validated using field data to confirm that the C and N balances close for each sub-system (verifying the equality in each Equation). The parameters we require are the overall production of SBPPRL and N content, which were obtained by Carneiro *et al.* (2005) and are shown in Table 21. These results are the average from the first 4 years after installation.

Table 21 – Dry matter production and average N content of SBPPRL biomass (Carneiro *et al.*, 2005).

Farm #	Total dry matter ($kg.ha^{-1}$)	Average N content (%)	Average N sequestered ($kg.ha^{-1}$)
2	9 040	2.42	220
3	5 660	2.47	140
4	10 595	2.81	300
5	3 940	2.50	100
6	8 765	2.72	240
7	6 720	2.81	190
8	4 300	2.47	100
Average	**7 000**	**2.60**	**184**

N – nitrogen.

3.2.2.7. Results for the GHG balance

Results from the application of the emission factors in Table 20 to the changes in stocking rate considered are shown in Table 22. The doubling of the stocking rate will double emissions (1.4 instead of 0.7 t $CO_2e.ha^{-1}.yr^{-1}$), since there are no transfers and no changes in feed consumption.

Table 22 – CH_4 and CO_2e emissions from cattle.

	Breeding cows	
GHG	**NG**	**SBPPRL**
t $CH_4.ha^{-1}$	0.028	0.056
t $CO_2e.ha^{-1}$	0.701	1.403

NG – Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; CH_4 – methane; N_2O – nitrous oxide.

Table 23 shows results for NGHGB for NG, and Table 24 for SBPPRL. NGHGB is negative (sink) for both NG and SBPPRL. However, sink in SBPPRL is about three times that of NG, even though there are more emissions in this system than in NG. This is due to the fact that the average ΔC_{SOM} is higher for SBPPRL. Even in the first year, when SBPPRL lose carbon due to tillage and less production, they still increase the SOM pool more than NG. However, once the SBPPRL system becomes an emitter (more emissions from soils and cattle than carbon sequestration), which happens at the 6th or 7th year, it emits more than NG even in the most favourable scenario for SBPPRL.

Table 23 – NGHGB for NG.

Year	ΔC_{SOM} MBD = 1.25 g.cm^{-3}	ΔC_{SOM} MBD = 1.40 g.cm^{-3}	C_{LW}	$F^{CO_2}_{r,LW}$	C_{lime}	$F^{CH_4}_{EF}$	$F^{N_2O}_{l}$	NGHGB MBD = 1.25 g.cm^{-3}	NGHGB MBD = 1.40 g.cm^{-3}
1	-4.96	-5.54	0.18	0.50	0.00	0.70	0.00	-3.58	-4.15
2	-3.69	-4.12	0.18	0.50	0.00	0.70	0.00	-2.31	-2.74
3	-2.77	-3.10	0.18	0.50	0.00	0.70	0.00	-1.39	-1.72
4	-2.10	-2.34	0.18	0.50	0.00	0.70	0.00	-0.72	-0.96
5	-1.60	-1.79	0.18	0.50	0.00	0.70	0.00	-0.22	-0.40
6	-1.23	-1.37	0.18	0.50	0.00	0.70	0.00	0.15	0.01
7	-0.95	-1.06	0.18	0.50	0.00	0.70	0.00	0.43	0.32
8	-0.74	-0.83	0.18	0.50	0.00	0.70	0.00	0.64	0.55
9	-0.58	-0.65	0.18	0.50	0.00	0.70	0.00	0.80	0.73
10	-0.46	-0.51	0.18	0.50	0.00	0.70	0.00	0.92	0.87
Average	**-1.91**	**-2.13**	**0.18**	**0.50**	**0.00**	**0.70**	**0.00**	**-0.53**	**-0.75**

NGHGB – Net Greenhouse Gas Balance; NG – Natural Grasslands; MBD – Mineral Bulk Density; C – Carbon; SOM – Soil Organic Matter; LW – Live Weight; r – Respiration; EF – Enteric Fermentation; CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

Table 24 – NGHGB for SBPPRL.

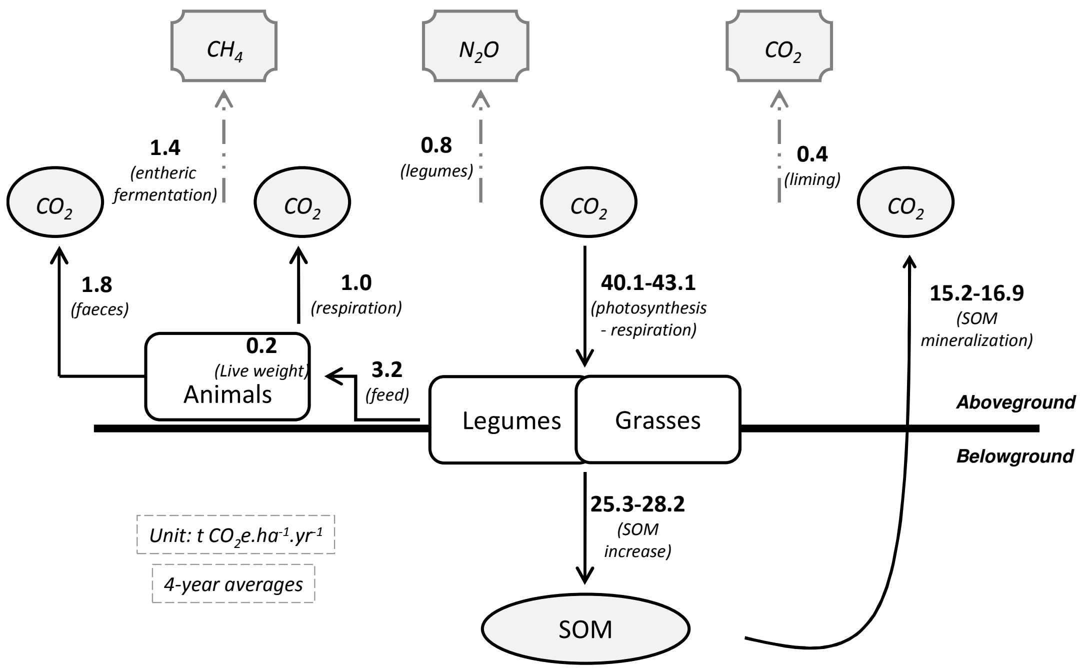
Year	ΔC_{SOM} MBD = 1.25 g.cm^{-3}	ΔC_{SOM} MBD = 1.40 g.cm^{-3}	C_{LW}	$F^{CO_2}_{r,LW}$	C_{lime}	$F^{CH_4}_{EF}$	$F^{N_2O}_{l}$	NGHGB MBD = 1.25 g.cm^{-3}	NGHGB MBD = 1.40 g.cm^{-3}
1	-5.53	-6.18	0.18	1.00	0.24	1.40	0.80	-1.91	-2.56
2	-14.97	-16.72	0.18	1.00	0.00	1.40	0.80	-11.59	-13.33
3	-10.06	-11.23	0.18	1.00	0.00	1.40	0.80	-6.68	-7.85
4	-6.77	-7.56	0.18	1.00	0.24	1.40	0.80	-3.15	-3.93
5	-4.56	-5.09	0.18	1.00	0.00	1.40	0.80	-1.17	-1.70
6	-3.07	-3.43	0.18	1.00	0.00	1.40	0.80	0.31	-0.05
7	-2.07	-2.31	0.18	1.00	0.00	1.40	0.80	1.31	1.07
8	-1.40	-1.56	0.18	1.00	0.24	1.40	0.80	2.22	2.06
9	-0.95	-1.06	0.18	1.00	0.00	1.40	0.80	2.44	2.33
10	-0.64	-0.71	0.18	1.00	0.00	1.40	0.80	2.74	2.67
Average	**-5.00**	**-5.58**	**0.18**	**1.00**	**0.07**	**1.40**	**0.80**	**-1.55**	**-2.13**

NGHGB – Net Greenhouse Gas Balance; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; MBD – Mineral Bulk Density; C – Carbon; SOM – Soil Organic Matter; LW – Live Weight; r – Respiration; EF – Enteric Fermentation; CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

These results are divided between the several sources as shown in Figure 21. Aires *et al.* (2008) found a global primary production for natural grasslands in a normal precipitation year equal to 46.2 t CO_2e.ha^{-1}.yr^{-1}. This figure is similar to the production of SBPPRL shown in Figure 21. If we repeat calculations for natural pastures (using the model for NG obtained in section 1), we obtain a production of 19.1 – 20.4 t CO_2e.ha^{-}

$^{1}.yr^{-1}$. This figure is approximately half of the one obtained in Aires *et al.*'s (2008) measurements.

Figure 21 – NGHGB of the SBPPRL system.



NGHGB – Net Greenhouse Gas Balance; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

3.2.2.8. Closure of C and N balances

The C and N balances of SBPPRL and NG are quantified in Table 25. On average, the simplified C balance states that the overall plant production has to be equal to the SOM balance in soils, livestock respiration and emissions from enteric fermentation:

$$P_t^{CO_2} = C_K - \alpha\, C_{SOM} + F_{r,LW}^{CO_2} + F_{EF}^{CH_4}. \qquad (3.17)$$

For example, for SBPPRL, summing the three terms (notice that SOM is the average from the first four years), we obtain an estimated balance of production of 11.73 and 12.82 t $CO_2e.ha^{-1}.yr^{-1}$. Aboveground production, according to Table 21, was on average 7 $t.ha^{-1}.yr^{-1}$. Considering that 58 % of production is carbon, and multiplying by 44 and dividing by 12 (respectively the molecular and atomic weights of CO_2 and C), we obtain 14.89 t $CO_2e.ha^{-1}.yr^{-1}$. Note that production is equal to the difference between photosynthesis and respiration. Therefore, the balance closes with 79 or 86 % of accuracy. This means that production is higher than the sum of the outputs. For NG, we did not have production figures, and therefore we estimate that NG produce half of the biomass as SBPPRL, since the stocking rate is also half. Balance closure is lower for NG, at around 62 – 68 %.

The simplified version of the nitrogen balance is

$$P_t^N = \Delta N_{SOM} + F_l^{N_2O}. \quad (3.18)$$

According to Rodrigues *et al.* (2010), the C:N content of SOM in SBPPRL is around 10. The average SOM_N in the first four years is then 255 or 284 kg $N.ha^{-1}.yr^{-1}$. N emissions from legumes are 0.8 t $CO_2e.ha^{-1}$, which is equal to 3 kg N_2O-N. The sum of both is about 258 or 284 kg $N.ha^{-1}.yr^{-1}$. According to Table 21, the average N in SBPPRL plants is 184 kg $N.ha^{-1}.yr^{-1}$. This means only 44 or 60 % accuracy. For NG, we used the same assumption for half of the production with the same N content. The percentage of closure is higher, at 88 – 100 %. However, for both grassland systems, output is higher than estimated N in biomass.

Table 25 – C and N balances for SBPPRL and NG.

C balance (t $CO_2e.ha^{-1}.year^{-1}$)			N balance (kg $N.ha^{-1}.year^{-1}$)		
Term	**SBPPRL**	**NG**	**Term**	**SBPPRL**	**NG**
$C_K - \alpha C_{SOM}$ (first 4 years)	9.33 - 10.42	3.38 - 3.77	ΔN_{SOM} (first 4 years)	255 - 284	92 - 103
$F_{EF}^{CH_4}$	1.40	0.70	$F_l^{N_2O}$	3	0
$F_{r,LW}^{CO_2}$	1.00	0.50			
$C_K - \alpha C_{SOM} + F_{r,LW}^{CO_2} + F_{EF}^{CH_4}$	11.73 – 12.82	4.58 - 4.97	$N_k + N_G + N_{LW} + F_l^{N_2O}$	258 - 284	92 – 103
$P_t^{CO_2} = F_P^{CO_2} - F_{r,plant}^{CO_2}$	14.89	7.44	P_t^N	184	92
% closure	**79 – 86 %**	**62 – 67%**	**% closure**	**(-) 44 – 60%**	**(-) 88 - 100%**

C – Carbon; N – Nitrogen; CO_2 – Carbon dioxide; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; NG – Natural Grasslands; SOM – Soil Organic Matter; LW – Live Weight; r – Respiration; EF – Enteric Fermentation; G – Groundwater; l – Legumes; CH_4 – methane; N_2O – nitrous oxide; (-) output higher than production.

We can also calculate the balance for each sub-system and see where the closure error comes from. We only show results for SBPPRL.

The carbon balance for aboveground biomass states that production ($P_a^{CO_2}$) is equal to the sum of carbon loss via leaf senescence (C_s) and ingestion by animals (C_i), which in turn is equal to 1/0.7 C_i. Average aboveground production is equal to total production divided by 1 + R:S. We obtain 3.9 t $CO_2e.ha^{-1}.yr^{-1}$. C_i is equal to 3.2 t $CO_2e.ha^{-1}.yr^{-1}$ (82 % balance closure). This indicates that average production in SBPPRL may not be completely consumed by grazing animals (and thus a higher sustainable stocking rate is possible), or that the intake is higher than estimated in SBPPRL.

The nitrogen balance for aboveground biomass states that aboveground N is equal to N lost by leaves senescence and ingestion by animals. Assuming a C:N of biomass also equal to 25 (standard for grasses and legumes), then the sum of both outputs is equal to 183 kg $N.ha^{-1}.yr^{-1}$, while the average N in SBPPRL plants is 184 kg $N.ha^{-1}.yr^{-1}$ (100% closure).

The carbon balance for aboveground biomass states that production ($P_b^{CO_2}$) is equal to SOM accumulation in soils due to incorporation of roots (K_{roots}). Belowground production is equal to aboveground production multiplied by the R:S, and therefore is 10.9 t $CO_2e.ha^{-1}.yr^{-1}$. However, K_{roots} is not obtained directly, and can only be calculated using the balance of the soil sub-system in Equation (3.4), which states that K_{roots} is equal to the overall SOM yearly accumulation minus the C entry from leave senescence and faeces. In that case, K_{roots} = 5.43 – 6.52 t CO_2e.ha^{-1}.yr^{-1} (58 – 63 % closure). This indicates that belowground production may be overestimated (R:S used is too high), or that SOM storage is underestimated.

The nitrogen balance for belowground biomass also requires N_K, the N output to soil, to be determined using the balance to soils, as the N stock is SOM minus the N entering soils from senescence, faeces and the N emitted as N_2O. The balance of these terms is 125 – 157 kg N.ha^{-1}.yr^{-1}. In the belowground balance, this value must be equal to N in belowground production, which is obtained as N in aboveground production times the R:S, and therefore is 516 kg N.ha^{-1}.yr^{-1} (24 – 30 % closure). Note that this balance assumes that all N is stocked in SOM, since we did not account any more gases leaving the system.

Regarding the balance from animals, removing the null terms, it states that the ingested C is either respirated, emitted as CH_4 or expelled as faeces. As indicated before, C_i is equal to 3.2 t CO_2e.ha^{-1}.yr^{-1}. The sum of the other terms is equal to 4.2 t CO_2e.ha^{-1}.yr^{-1} (76% closure), which means that either ingestion is underestimated, or emissions are overestimated.

Finally, the N balance from livestock states that N ingestion is equal to N in faeces. Since N_i = 183 kg N.ha^{-1}.yr^{-1}, and N_f = 72 kg N.ha^{-1}.yr^{-1}, there is only 39 % closure. This means that not all terms were accounted, and ingested N is emitted or expelled from the animals in other forms.

3.2.2.9. Discussion of results for the carbon balance

In this section we calculated the carbon balance of two grassland systems in Portugal, namely NG and SBPPRL. For SBPPRL, we obtained an average carbon sink (in 10 years) of 1.55 – 2.13 t CO_2.ha^{-1}.yr^{-1}, depending on the mineral bulk density (1.25 or 1.40 g.cm^{-3}). NG also have a significant sink potential of 0.53 – 0.75 t CO_2.ha^{-1}.yr^{-1}. The fertilized NG system was not consider here, since it is an intermediate situation between NG and SBPPRL.

The sink potential is mainly due to carbon sequestration and storage as SOM. The yearly sink may be as high as around 15 t CO_2.ha^{-1} for SBPPRL. During this stage, NBP is negative, which means that more production is stored than exported. However, the sink effect phases out as SOM pools stabilize, and the NBP becomes positive. After around 8 years, pastures become emitters. Since the stocking rate is higher and legumes emit more, after eight years NG have a lower carbon balance than SBPPRL. Still, it is important to notice that we used one hectare as the functional unit in the calculation, since it is also the base for Kyoto reporting. If we had used kilogram of meat produced, quilogram of protein in feed, or Euro of meat sold, results could change. SBPPRL are richer in legumes than NG, and provide a better quality feed to livestock.

Even though the functional unit used here is conservative, after 10 years SBPPRL are almost at 3% SOM. Soils with such high SOM concentration are more productive and farms benefit from overall environmental effects (decreased soil erosion, increased water holding retention). From that point on, those other agronomic, economic and environmental benefits of the systems may justify is maintenance for a longer term.

We obtain two sets of values, one for each soil MBD. The lowest MBD (1.25 $g.cm^{-3}$) is the indicative in Portugal (Maria de Fátima Calouro, personal communication), but soils which are not tilled are more compact, and thus MBD is likely higher (1.40 $g.cm^{-3}$) (Mário Carvalho, personal communication). Unpublished results (Carlos Aguiar, personal communication) from tests in one of the SBPPRL plots in the sample, namely Quinta da França, show an average MBD of 1.55 $g.cm^{-3}$, with a standard deviance of 0.09 $g.cm^{-3}$ (results from samples ranging from 1.37-1.82 $g.cm^{-3}$). More compact soils are usually problematic, particularly in the upper layer due to livestock stomping (Greenwood *et al.*, 2001). In those conditions, a rigid layer is created that imposes a barrier for plant growth and water infiltration (Bot and Benitez, 2005). However, in these grasslands, plants are annual, and the root system renews every year, leaving aeration channels which maintain a macropore structure in soils crucial for production (Drewry *et al.*, 2007). Furthermore, field work done in the context of Project Extensity showed that SBPPRL increase soil biodiversity, particularly macrofauna (Henriques *et al.*, 2006). Earthworms and insects create holes from which water and other organic matter penetrates the soil, which balances the effect of increased MBD (Bot and Benitez, 2005).

The calculations made for the C and N balances of the overall system and of each sub-system (aboveground and belowground plant, soil and animal), showed that the percentage of balance closure is usually higher than 50 % but significantly lower than 100 %, and higher for C than for N balances. This is mainly due to the fact that we did not include all N emissions to the atmosphere. We also did not use a grassland-specific R:S, turning instead to the literature. But the R:S used (2.8) seems to be too high, which was the cause for some of the balances not closing. In general, the limitations in our approach seem to be conservative for SBPPRL, since results suggest that the sustainable stocking rate can be higher or animals can eat more, and emissions are overestimated. Results are summed up in Table 26.

Table 26 – % of closure for C and N balances for SBPPRL, and main explanations.

Balance (SBPPRL)	Closure (%)	Likely explanation(s)
C – overall	79 – 86	Production overestimated, or either SOM storage or emissions underestimated
N – overall	(-) 44 – 60	N in biomass underestimated
C – aboveground	82	Not all production is consumed, and thus a higher stocking rate is possible, or C intake is underestimated
N – aboveground	100	-
C – belowground	58 – 63	Belowground production overestimated (R:S too high), or SOM storage underestimated
N – belowground	24 – 30	There are many more emissions from soil (NH_3, N_2, etc.) which were not included
C – soil	-	Cannot be calculated (no measured K_{roots})
N – soil	-	Cannot be calculated (no measured N_K)
C – animal	(-) 76	Ingestion is underestimated, or emissions are overestimated
N – animal	39	Ingested N is emitted or expelled from the animals in other forms

C – Carbon; N – Nitrogen; CO_2 – Carbon dioxide; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; (-) output higher than production.

Regarding the unaccounted effects in the balance, two parameters were left out of our study. First, even though there are no harvests for forage, in a natural system, there is some invasion by shrubs, which are intermediate stages in natural succession. Tillage is the most widely used system of shrub control in NG (Pinheiro *et al.*, 2008). During tillage, all shrub biomass in incorporated in soils, thus never leaving the plot. Therefore, for NG, $F_{harvest} \approx 0$ (but SOM accumulation is soils is also minimized).

However, in well managed SBPPRL, shrubs never grow, and therefore there is a loss in organic matter which corresponds to shrub production. We consider that $F_{harvest}$ in SBPPRL is equal to the aboveground production (since belowground production in NG is captured by SOM increase) of woody biomass removed during tillage in NG. Ehleringer and Mooney (1983) estimated aboveground net primary productivity of shrubs as 1.1 – 1.3 $t.ha^{-1}.yr^{-1}$, while Castro and Freitas (2008) find values around 1.9 – 2.7 $t.ha^{-1}.yr^{-1}$. As an intermediate value, we may consider $F_{harvest} = 2$ $t.ha^{-1}.yr^{-1}$.

The second effect regards losses that may occur in NG. We considered NG as unmanaged areas. However, two important disturbance events may occur in NG: tillage for shrub control, or forest fires. Tillage would mineralize most of the SOM accumulated. If it did not occur, forest fires could take place. During a forest fire, large scale emission of greenhouse gases occurs. This is likely in NG, but not in SBPPRL, because, as stated before, shrub growth is limited. Therefore, two effects deter fires in SBPPRL, namely:

- The grazed agro-forestry mosaic landscape interrupts large areas of forest monocultures prone to fire;
- Unlike in NG, increased grazing pressure in SBPPRL stops invasion by shrubs, which are also good fuel for forest fires.

We also did a farm scale analysis, which did not take into account life cycle effects. One important effect regards feeds. Instead of increasing stocking rates, farmers who install SBPPRL may opt to transfer steers from intensive to extensive production, thus decreasing stable periods and feed consumption. Van der Werf *et al.* (2005) indicate an average emission for pig feed of 528 kg $CO_2e.t^{-1}$ feed. Their value is close to the 498 kg $CO_2e.t^{-1}$ feed found by Blonk *et al.* (1997) and the 591 kg $CO_2e.t^{-1}$ feed found by Carlsson-Kanyama (1998), also for pig feed. Casey and Holden (2005) consider the emission of 1156 kg $CO_2e.t^{-1}$ feed for lactating cows, which need more protein-intensive ingredients (thus produced with more fertilizers). Therefore, emissions from feeds are not neglectible, and are the reason for one of the next Sections of this Chapter, in which we determine the life cycle impacts of pastures.

One important assumption regards N_2O emissions. These emissions from soils change as legume percentage changes, as well as soil C:N ratios (Bot and Benitez, 2005; Ostle *et al.*, 2009). Legume percentage is known to be higher in the first years in SBPPRL, but C:N ratios decrease in pastures (Rodrigues *et al.*, 2010). These two effects have contradictory results in N_2O emissions, and the overall balance is unknown. Besides, calculations made for N balances suggest that N_2O emissions may be overestimated.

Some authors have already claimed that N_2O emissions, estimated using typical emission factors from IPCC (1997), could be overestimated. Li *et al.* (2005), modelled carbon sequestration enhancement strategies, and found that carbon dynamics influences nitrogen dynamics. They estimate that, when SOM increases, the increase in N_2O emissions diminishes or even eliminates the carbon sequestration. However, empirical studies like Crews and Peoples (2004), Kammann *et al.* (1998), Ledgard (2001) or Rochette and Janzen (2005) state that emission factors are systematically overestimated, and find values close to the lowest extreme of the interval presented here. These studies refer to mixes of grass and legumes, and therefore are a valid approximation of the system studied here. In equilibrium, it is very likely that fixed nitrogen is fully consumed by grasses.

But even if emissions were of the magnitude of Li's (Li *et al.*, 2005), these emissions may be offset by their alternative. Nitrogen fixed by legumes is a direct substitute of synthetic nitrogen in fertilizers. To achieve close productivities without legumes, pastures would require nitrogen fertilization. The other option would be to use commercial feed for cattle, but then fertilized crop production would be required. Therefore, we may compare the emissions due to biological N fixation to the emissions due to fertilizer production and use.

Considering the emission factor in IPCC (1997) stated above, each fixed N unit corresponds to 3.88 kg $CO_2e.ha^{-1}$ emitted. We found on the database of the Life Cycle Assessment software SimaPro 6.0 the CO_2 emissions for which fertilizers are responsible during their production. We found that for each kg of N in the fertilizer urea 3 kg of CO_2e are emitted, which is about the same as what is emitted by legumes. But for each kg of N in the fertilizer ammonium nitrate, 7.75 kg of CO_2e are emitted, and this is about the double of the emissions due to biological fixation. Therefore, each unit of nitrogen is used more efficiently if it is biological rather than synthetic.

It is important to remark that, for Kyoto accounting purposes, the effects mentioned here contribute to different items. Carbon sequestration is accounted under the optional Article 3.4, "Grassland Management". Emissions from legumes, liming and livestock are reported for the agricultural sector, while feed and fertilizer consumption are reported under industrial activities. Furthermore, some effects deal with international transfers. For instance, if the increase in stocking rate would substitute meat imports and decrease the Portuguese meat deficit, the corresponding emissions would now be reported by Portugal, even if substituted meat was produced intensively. This is because Kyoto agreements place the entire burden of emissions on producers. It is crucial in future climate negotiations to account for implicit emissions in trading and develop a fair indicator of the allocation of CO_2 emissions between producers and consumers. Preliminary work on what a fair indicator would be has been done, for example, by Rodrigues *et al.* (2006).

A final word goes to emissions from livestock. In the previous results, we were subjected to the experimental setting of the AGRO projects. Since the AGRO projects were set at a plot scale, there was an effective increase in stocking rate. But it is not plausible that it is so when we move to bigger scales. At the country scale, change from NG to SBPPRL implies maintenance in stocking rate, and decrease in feed consumption by cows. This scenario is the most likely. Table 27 shows that practically all breeding cows in Portugal are subsidized, and so, if animal quotas are maintained, then there can be almost no global stock rate increase.

Table 27 - Registered (INE[32]) and supported (IFADAP/INGA, 2004) breeding cows in Portugal.

Year	Registered animals	Supported animals	% of Supported Animals
1998	341 000	321 948	94.4
1999	342 000	303 700	88.8
2000	342 000	307 093	89.8
2001	351 000	307 731	87.7
2002	359 000	321 978	89.7
2003	371 000	332 243	89.6
2004	384 000	365 050	95.1

Our results would also change if, instead of using the NIR emission factors, we used factors from the IPCC (1997). Unlike the NIR, the IPCC considers that faeces also emit CH_4 and N_2O. Therefore, in our first scenario, we start from a degraded grassland with a stocking rate of 0.5 LU, and then introduce 0.5 LU of breeding cows after the grassland is sown. Corresponding emissions would rise by 1.1 t CO_2 eq.ha^{-1}.yr^{-1}, as shown in Table 28.

Table 28 – Emissions from breeding cows in pastures.

Gas	Emission factors		Stocking rate	Emissions	
	Enteric fermentation	Faeces[33]		Enteric fermentation	Faeces
	kg.head^{-1}.yr^{-1}	kg.head^{-1}.yr^{-1}	head.ha^{-1}	kg.ha^{-1}.yr^{-1}	kg.ha^{-1}.yr^{-1}
CH_4	73[34]	2.156[35]	0.5	36.5	1.078
N_2O	0[36]	1.927[37]	0.5	0	0.964
CO_2e				766.5	321.478

CH_4 – Methane; N_2O – Nitrous oxide; CO_2e – Carbon dioxide equivalent.

For Portugal, this is the worst scenario possible of increased animal stocking rate, and it represents only about 20% of total carbon sequestration. Therefore, global carbon balance would still be very favourable, and SBPPRL would still be carbon sinks. Considering that meat demand in Portugal does not change by implementing such policies, this would represent a transfer: instead of buying meat produced elsewhere, Portugal would produce it locally. Therefore, in terms of global world emissions, they would remain the same.

[32] http://www.ine.pt/prodserv/quadros/quadro.asp

[33] Figures in this column have excessive precision. However, such are the exact figures provided by APA, and so we opted to maintain them as such.

[34] Enteric fermentation CH_4 emission factor for breeding cows in grasslands, in 2004.

[35] Faeces CH_4 emission factor for breeding cows in grasslands, in 2004.

[36] IPCC (who establishes the Kyoto accounting method) does not consider N_2O emissions from enteric fermentation.

[37] Faeces N_2O emission factor for breeding cows in grasslands, in 2004.

Finally, a third option would be to assume that there is a transition from a stocking rate of 0.5 $LU.ha^{-1}$, composed only by breeding cows, to a stocking rate of 1.0 $LU.ha^{-1}$ where for each cow, a steer is being fed and finished during a year (from 6 to 18 months). There are three major effects:

1. Breeding cows' population increases from 0.5 $LU.ha^{-1}$ to 0.625 $LU.ha^{-1}$.
2. Number of steers being fed in the pasture increases from 0 $LU.ha^{-1}$ to 0.375 $LU.ha^{-1}$, corresponding to an increase of 0.625 $steer.ha^{-1}$ (since 1 steer represents 0.6 LU).
3. Steers are withdrawn from intensive feeding, and so emissions corresponding to 0.625 $steer.ha^{-1}$ are avoided.

Effects 1, 2 and 3 are quantified in Table 29. Table 29 shows that global balance is 176 kg $CO_2e.ha^{-1}$ emitted. This value is minor when compared to carbon sequestration by SBPPRL. Detailed calculations are shown in Appendix II – Estimation of CO_2e emissions from livestock.

Table 29 – Effect on greenhouse gases' emissions of the stocking rate increase

Gas	Effect	Emission factors		Stocking rate	Emissions	
		Enteric fermentation	Faeces/ Manure		Enteric fermentation	Faeces/ Manure
		$kg.head^{-1}.yr^{-1}$	$kg.head^{-1}.yr^{-1}$	$head.ha^{-1}$	$kg.ha^{-1}.yr^{-1}$	$kg.ha^{-1}.yr^{-1}$
CH_4	1	73	2.156	0.125	9.125	0.270
	2	50.2	0.679	0.625	31.375	0.424
	3	-50.2	-1.156	0.625	-31.375	-0.723
N_2O	1	0	1.927	0.125	0	0.241
	2	0	0.659	0.625	0	0.412
	3	0	-1.122	0.625	0	-0.701
CO_2e	1			0.125	191.625	80.331
	2			0.625	658.875	136.593
	3			0.625	-658.875	-232.560
Total (kg CO_2e)					191.625	-15.636

CH_4 – Methane; N_2O – Nitrous oxide; CO_2e – Carbon dioxide equivalent.

In this case, Portugal would have a benefit from switching from an intensive to an extensive production system. In terms of global effects, since transfers would remain domestic, there would also not be a significant effect. But the overall conclusion remains the same: SBPPRL are still a carbon sink.

As we have shown, the whole SBPPRL system is a sink. But is it a more powerful sink than other possible land uses? We now answer this question when we refer available data for carbon sequestration in uses such as agriculture and forests.

3.3 Comparison with other agricultural land uses for carbon sequestration[38]

Portugal will have to report, in the context of Article 3.4 of the KP, the carbon balance of all land uses. One methodological distinction will be made between areas under specific management activities, and unmanaged areas. Two specific activities will be reported. The first one is SBPPRL, under the grassland management item, and the second one is no-tillage, under the cropland management item.

3.3.1 No-tillage

Like SBPPRL, the use of no-tillage as a specific management activity has been pinpointed in the literature as having a substantial potential for carbon sequestration. No-tillage is particularly important for annual crops (Basch, 2002).

To assess the potential of no-tillage for carbon sequestration, we start by presenting next a small literature review of available studies, to frame the figures obtained next. We then estimate the potential for carbon sequestration using the (default) IPCC method. Finally, national data is used to conclude the final figures to be used for national accounting.

3.3.1.1. Literature review

Table 30 shows a short literature review on carbon sequestration from no-tillage in annual crops. Figures found in these studies range from 0 to 3 t $CO_2.ha^{-1}$ (ECCP, 2003).

Table 30 – Literature review of available studies on carbon sequestration from no-tillage.

Reference	**Study method**	**Carbon sequestration ($t\ CO_2 \cdot ha^{-1} \cdot yr^{-1}$)**
ECCP, 2003	Survey	0-3.0
Six *et al.*, 2004	Survey	0.8
Six *et al.*, 2004	Survey	0.4
Smith, 2004	Survey	1.4
Cambardella and Elliott, 1992	Medição	1.22
Six *et al.*, 2002	Survey	1.2 +- 0.4
West and Post, 2002	Survey	0.21 +- 0.05
Freibauer *et al.*, 2004	Survey	0.6
Bernacchi *et al.*, 2005	Measurements (*Eddy covariance*)	2.2
Marland *et al.*, 2003	Survey	0.34
Marland *et al.*, 2004	Survey	0.57 +- 0.14

3.3.1.2. Calculations using the IPCC default method

The IPCC (2006) has a method to calculate the carbon sequestration potential of no-tillage using the expression

[38] In this sub-section, we do not consider carbon sequestration from forest management. Carbon storage in aboveground biomass is important for forests, but not for the rest of the agricultural land uses, which makes them hardly comparable. However, and as a curiosity, according to Pereira *et al.* (2009a), cork oak *montados* sequester around 1-5 t $CO_2.ha^{-1}.yr^{-1}$, pine forests sequester around 15-26 t $CO_2.ha^{-1}.yr^{-1}$, and eucalyptus forests sequester around 15-32 t $CO_2.ha^{-1}.yr^{-1}$.

$$\Delta C_i = \Delta C_{ABi} + \Delta C_{BBi} + \Delta C_{DWi} + \Delta C_{LIi} + \Delta C_{SOi} + \Delta C_{HWPi}, \tag{3.24}$$

where ΔC_i is the change in carbon stock for soil use *I*, and the subscripts have the following meanings:

AB – Aboveground biomass, assumed zero for non-woody crops;

BB – Belowground biomass, assumed zero for non-woody crops;

DW – Dead woody biomass, assumed zero for non-woody crops;

LI – Litter, which is biomass encompassed in soils by the death of plants and management actions;

SO – Soil carbon;

HWP – Biomass removed as woody products, assumed to be zero for non-woody crops.

Therefore, for no-tillage of annual (non-woody) crops, Equation (3.24) becomes

$$\Delta C_{NT} = \Delta C_{SO_{NT}}, \tag{3.25}$$

where the soil carbon balance is calculated by the following expression

$$\Delta C_{SO} = \frac{C_t - C_0}{D}. \tag{3.26}$$

In Equation (3.26) *C* represents soil organic carbon (SOC), *t* is the inventory year (under no-tillage), and *0* represents the baseline year (conventional tillage). *D* is the reference time period for the change in SOC from C_0 to C_t. *C* is calculated using

$$C = C_{REF} \bullet F_{LU} \bullet F_{MG} \bullet F_I, \tag{3.27}$$

where:

C_{REF} – reference carbon stock in managed soils; for no-tillage, we assumed the value 18.8 t $C.ha^{-1}$ (Fátima Calouro, comunicação pessoal).

F_{LU} – stock change factor for land-use systems or sub-system for a particular land-use, dimensionless; for annual crops, the figure is 0.80 (IPCC, 2006);

F_{MG} – stock change factor for management regime, dimensionless; for conventional tillage, the value is 1.00, and for no-tillage 1.10 (IPCC, 2006);

F_I – stock change factor for input of organic matter, dimensionless; we considered that when crop residues are left on the field, this parameter is *medium* (1.00) or *high* (1.37), and it is *low* (0.95) otherwise.

Note that all equations mentioned before are applied on a hectare basis.

We thus estimate, from Equations (3.24) to (3.27), carbon sequestration from no-tillage as 0.26 t $CO_2.ha^{-1}.yr^{-1}$, in case no crop residues are left on the field. If residues are left, the value rises to 0.28 or 0.38 t $CO_2.ha^{-1}.yr^{-1}$ (medium or high input of SOM).

3.3.1.3. Calculations using data from the Évora University

However, an analysis of existing data for Portugal shows that this value may be underestimated. Most studies were conducted in research projects by the Évora University, and all refer to rainfed crops.

Data for Herdade da Revilheira in Évora, Portugal, (Carvalho and Basch, 1995) shows that in no-tilled crops, leaving residues on the field, SOM increases by of about 0.47 percent points in the first 30 cm and in five years (Table 31). The test took place in a plot field of 2.8 ha in a four year rotation. The rotation was sunflower – wheat – forage – triticale. Sunflower was replaced in some plots during the test, due to the fact that it was highly attacked by birds (Mário Carvalho, personal communication).

Table 31 – SOM change (1999-2004) in no-tilled luvisoil areas in Herdade da Revilheira, leaving residues on the field (Carvalho and Basch, 1995).

Depth (cm)	Increase (%)
0-10	0.86
10-20	0.42
20-30	0.15
0-30 (average)	0.47

SOM – Soil Organic Matter.

Other results are shown in Table 32, and confirm that SOM increases by about 0.1 percent points per year in 30 cm in no-tilled plots, when compared with conventional tillage. Increases below that depth occur due to Summer cracks, which allow residues to fall to the interior of the layer.

Table 32 - SOM concentration and soil respiration (mineralization) in a cromic vertisol (Almocreva, Barros de Beja, Portugal) after 8 years under different tillage systems.

Depth (cm)	SOM concentration (%)		Soil respiration ($mgCO_2.10g^{-1}$ soil)	
	NT	CT	NT	CT
10	2,53	1,91	3,17	0,69
20	2,15	1,67	3,87	2,88
30	2,25	1,62	5,80	3,70
40	2,22	1,33	-	-

SOM – Soil Organic Matter; CO2 – Carbon dioxide; NT – No-Tillage, CT – Conventional Tillage. Source: Carvalho and Basch (1995).

Soils in which all crop residues are removed have expectedly lower SOM concentrations, since without cover soil respiration (another term used for the SOM mineralization rate) increases. Available studies corroborate this statement, as shown in Table 33. In four years, no-tilled soils show higher SOM concentrations of about 0.1 percent points (or 0.025 percentage points per year) in relation to other tillage methods. Note that SOM concentration is higher in the 20-30 cm layer for conventional tillage due to the burying of organic components by the plow. Note however that, in the case of these trials, the rotation did not include a forage stage.

Table 33 - SOM concentration and distribution in a luvisoil area, after four years of tillage.

Depth (cm)	SOM concentration (%)			
	NT	R	Chi	CT
0-10	1.70	1.50	1.60	1.30
10-20	1.30	1.40	1.20	1.30
20-30	1.00	0.90	0.90	1.10
0-30 (average)	1.33	1.27	1.23	1.23

SOM – Soil Organic Matter; CO2 – Carbon dioxide; NT – No-tillage; R – Ripper; Chi – Chisel; CT – Conventional tillage.
Source: Carvalho, unpublished.

For winter crops, Carvalho *et al*. (2002) also determined the SOM concentration in soils under no-tillage for three years, in the 0-30 cm layer, and four different quantities of straw left on the fields. Data obtained was used to calibrate a statistical model to relate SOM concentration and the quantity of residues left (*R*, measured in tons), which is,

$$SOM(\%) = 0,89 + 0,13R(t) \,. \tag{3.28}$$

From this expression, we may extrapolate that in a situation without residues, SOM concentration is 0.89%. For each ton left, SOM increases by 0.13 percent points.

We may assume that, even when all straw is removed, 1 $t.ha^{-1}$ of residues always stay on the field (Mário Carvalho, personal communication). The average straw produced for oats, wheat and triticale is around 2.2 $t.ha^{-1}$ (average straw productivity from the GPP (2001) database). Therefore, the difference between leaving straw on the field is 1.2 $t.ha^{-1}$. From Equation (3.28), this represents a 0.156 percent points difference in SOM concentration. Since the parameters in the equation were calculated for a three year period, this is equivalent to 0.052 percent points per year.

Considering the equivalences in Section 3.1, and assuming the 30 cm depth, the yearly increase of 0.025 percent points which corresponds to no-tillage without residues left on the field is equivalent to the sequestration of 1.9-2.1 t $CO_2.ha^{-1}$. Similarly, the increase of 0.1 percent points from no-tillage with residues is equivalent to 7.7-8.5 t $CO_2.ha^{-1}$. The difference between these two situations is, then, of 5.8-6.4 t $CO_2.ha^{-1}$. All these figures are much higher than those obtained from the IPCC (2006) method shown before.

We use results from Carvalho *et al*. (2002) to confirm the previous ones. As shown before, the difference between the case with and without straw was 0.052 percent points per year, which is equivalent to 4.0-4.4 t $CO_2.ha^{-1}$, which is similar to, but still 30 lower than, the value calculated above. Note that this previous value is, however, subjected to higher uncertainty, since we had to look at a different source to determine typical straw production.

3.3.1.4. Results and discussion

The analysis done has allowed us to obtain the following conclusions regarding the carbon sequestration potential of no-tillage:

1. Without residues left on the field – 1.9-2.1 t $CO_2.ha^{-1}.yr^{-1}$;
2. Leaving residues on the field – 7.7-8.5 t $CO_2.ha^{-1}.yr^{-1}$.

We again note that these values are yearly averages of a four year rotation where, in the second case, soil cover in one year was a forage crop, and therefore no crop residues are produced. Hence, the situation in that year is similar to the one where no residues are left on the field.

Regarding the sequestration factor to use for each crop, there are several possibilities, which result from combining the following conditions:

- Whether it is a Spring/Summer crop, a Fall/Winter crop, or a fallow;
- Whether it is a rainfed or an irrigated crop.

Typical Fall/Winter crops are wheat and triticale. The majority of farmers sell straws from these crops, and therefore only low-quality residues remain on the field (Mário Carvalho, personal communication). Carbon wise, we are in situation 1 above.

The main rainfed Spring/Summer crop is sunflower, and the main irrigated Spring/Summer crop is maize. Grain maize residues always remain on the field, since grain maize straw has no commercial value. Carbon wise, we are in situation 2 above. Silage maize, however, is a different story, since the whole plant is harvested, and so we are in situation 1.

Since all test results shown before were obtained in rainfed plots, the question arises of which sequestration factor to use for irrigated crops. There are two reasons why we can believe that irrigated and rainfed crops are different:

- Irrigated areas are more productive, with more root formation, and hence more organic matter enters the soil;
- Since there is water available in the hot seasons, the conditions for higher soil respiration are gathered, hence decreasing SOM accumulation.

Since there are no know systematic studies of irrigated crops, we considered that the first effect is predominant (Mário Carvalho, personal communication). Therefore, the use of the rainfed sequestration potential is a conservative option for irrigated areas.

Therefore, Table 34 shows a summary of which factores to use in each combination of cases.

Table 34 - Carbon sequestration factors for no-tillage.

Crop	Sequestration factor (t $CO_2.ha^{-1}.yr^{-1}$)			
	Spring/Summer crop	Fall/Winter crop with straw	Fall/Winter crop without straw	Fallow
Rainfed	1.9-2.1	7.7-8.5	1.9-2.1	1.9-2.1
Irrigatd	7.7-8.5	7.7-8.5	1.9-2.1	1.9-2.1

Note also that the studies indicated here were done on luvisoils and vertisols. We can admit that, in other soil types, results will be different. However, a few considerations are in order:

- In soils with higher clay content, the use of a luvisoil sequestration factor may be conservative, since these are usually more productive and therefore expectedly will have higher SOM increases.

- In sandy soils, the use of a carbon sequestration potential from luvisoils may be an overestimation. However, due to their frequent low productivity, hardly any crops are produced in sandy soils in Portugal.

3.3.2 Other land uses

For other land uses, we studied whether there is SOC stock change for cropland remaining cropland, for each crop. Since other crops are either (1) permanent or (2) annual with soil mobilization, it may be argued that there is no significant carbon stock change. In the first case, there is no SOC accumulation because the long-term SOM equilibrium is already reached. In case (2), there is also no accumulation because periodic soil mobilization creates the conditions for SOC inputs to deep layers of soil to be mineralized quickly.

To assess if these statements are true, we used two methods: a Paired Samples T Test, and a time series model to determine the cointegration order of all sample series. We explain both briefly next.

3.3.2.1. Method

We intended to study whether SOC remains constant in Portuguese croplands that remain croplands. For some available trials there were only two sampling years, and for others there were as many as twelve. Two methods were used:

1. For samples consisting on two or more years, we study if the average soil organic matter in all sampling years is equal (or, to be accurate, if we cannot statistically reject the hypothesis that averages vary significantly amongst sampling years);
2. For samples consisting on more than two years, determine if the time series of measured SOC is stationary (which means that there is no upward or downward trend).

In the first case, we used a Paired Samples T Test. This test consists on determining the difference between the average values of two series, and determining if that difference is statistically different from zero (given the standard deviation of each sample). Therefore, we compute the average SOC for any two years of a given sample; if both average values are equal, then we assume that there was no change in soil carbon stocks between the two years.

This test is only valid if both series are normally distributed. Therefore, we started by performing a Kolmogorov-Smirnov test on all series. This tests for a normal distribution.

The Paired Samples T Test is especially helpful when there are only two sampling years, since it is used to compare the average value of any two series. When there are more than two sampling years, we ran the test coupling all years in pairs. Therefore, we tested all possible combinations.

However, the Paired Samples T Test has some serious limitations. First of all, it requires series to have the same number of observations (since the difference between observations is calculated first, and only then the average of the differences). When the number of observations is different, some must be discarded. Therefore, not all available information is used.

Besides, this test is too restrictive. All years must be compared in pairs, and so it tests if averages are constant for all pairs (perfectly stationary series). However, SOC may vary significantly between particular years and, in the long term, the times series may still be stationary.

Therefore, we used a second method whenever there was a time series of more than two sampling years. We used the Box-Ljung statistics to determine the integration level of all series. To say that a series is stationary (statistically constant in time) is the same as to say that it is "zero order integrated". A "first order integrated" series has a linear trend, as so forth.

However, we needed a time series with all sampling years between the first and the last. Therefore, we linearly interpolated all missing values.

We used the statistical software SPSS 16.0.

3.3.2.2. Data

Data used were the result of two decades of soil sampling and analysis in several test sites. All trials were made by the former Laboratório Químico Agrícola Rebelo da Silva (LQARS). Soil samples were collected at two depths: 0-20 cm and 20-40 cm (for annual crops) or 20-50 cm (for permanent crops).

Data for SOC are structured as an incomplete time series for each crop in each site. Most series are incomplete since some sample years are missing. Soil use and available sample years for each test site are described in Table 35.

Table 35 – Trials done by LQARS, and available sample years for each type of land use.

Land Use	Species	Trial location	Trial #	Available sample years
Olive	“Verdeal Transmontana”	Mirandela	1	1987, 1990, 1993 to 1997
	“Cobrançosa”	Mirandela	2	1987, 1990, 1995
	“Galega”	Castelo Branco	3	1998 to 2002
	“Picual”	Santarém	4	1995, 1998
	“Cobrançosa”	Santarém	5	1987, 1989
	“Blanqueta”	Santarém	6	1989, 1995
	“Blanqueta”	Elvas	7	1988, 1991, 1998
	“Carrasquenha”	Elvas	8	1989 to 1999
Vineyard	“Bical”	Bairrada	9	1987, 1989, 1996
	“Castelão (Camarate)”	Bairrada	10	1988, 1990, 2000
	“Periquita (Castelão)”	Palmela	11	1988, 1989, 1992, 1996, 2000
	“Loureiro”	Vinhos Verdes	12	1987, 1990, 1992, 1996, 2000, 2004
Fall/Winter grains	Triticale	Viseu	13	1986, 1988 to 1990
	Triticale	Évora	14	1986 to 1988, 1990,1992
	Wheat	Coimbra	15	1985 to 1987
	Wheat	Serpa	16	1986, 1988 to 1992
Maize and rice	Rice	Coimbra	17	1986, 1990
	Maize	Estarreja	18	1984 to 1985, 1987 to 1991
	Maize	Aveiro	19	2002 to 2004
	Maize	Viseu	20	1980, 1982 e 1983, 1985
	Maize	Pegões	21	1988 to 1990
	Maize	Beira Interior	22	1981, 1983 to 1985
	Maize	V. Franca Xira	23	1987,1988,1989, 1990, 1991
	Maize	Chamusca	24	1985, 1987, 1989
	Maize	Elvas	25	1981, 1983, 1985, 1987

3.3.2.3. Results and discussion

Paired samples T-test

Results for the Paired Samples T Test are shown in Table 36. A test is inconclusive if there are more than two sampling years, and results differ for each pairing. They show downward or upward trends if, for available years, the average soil carbon levels decreased or increased in all pairings.

Results for annual crops (trials 13-25) mostly support that SOC does not change between years or, if so, there is an upward trend. Conclusions for perennial crops (trials 1-13) cannot be drawn from Table 36.

Table 36 – Conclusions on the stationarity of the soil carbon time series according to the Paired Samples T Test, per trial.

Trial #	Number of sampling years	Constant average?		If not, trend?		Conclusion
		0-20 cm	20-40/50 cm	0-20 cm	20-40/50 cm	
1	12	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
2	4	No	Inconclusive	Inconclusive	Downward	Non stationary
3	4	No	No data	Inconclusive	No data	Inconclusive
4	2	No	No	Downward	Downward	Downward trend
5	2	No	No	Downward	Downward	Downward trend
6	2	No	No data	Downward	No data	Downward trend
7	2	No	No	Upward	Upward	Upward trend
8	10	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
9	3	Inconclusive	No	Inconclusive	Downward	Inconclusive
10	3	No	No	Inconclusive	Inconclusive	Non stationary
11	5	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
12	6	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
13	4	Inconclusive	No data	Inconclusive	No data	Inconclusive
14	5	No data	No data	No data	No data	Inconclusive
15	2	No	No data	Downward	No data	Downward trend
16	6	Inconclusive	No data	Inconclusive	No data	Inconclusive
17	2	No	No data	Upward	No data	Upward trend
18	7 (0-20 cm); 2 (20-40 cm)	Inconclusive	No	Inconclusive	Upward	Inconclusive
19	3	No	No data	Upward	No data	Upward trend
20	4	Inconclusive	No data	Inconclusive	No data	Inconclusive
21	3	Inconclusive	No data	Inconclusive	No data	Inconclusive
22	4	Yes	No data	-	No data	Stationary
23	4	No	No data	Upward	No data	Upward trend
24	3	Inconclusive	No data	Inconclusive	No data	Inconclusive
25	4	Yes	No data	-	No data	Stationary

Time series model

Results for time series modelling are shown in Table 37. A series is considered stationary if it is zero order integrated. If it is first order integrated, then it is possible to determine if the trend is up or downward.

Results confirm the claim that SOC does not change between years for annual crops, but are again fuzzy for perennial crops.

Table 37 – Conclusions on the stationarity of the soil carbon time series according to time series modelling, per trial.

Trial #	# sampling years	# interpolated years	Stationary series?		If not, trend?		Conclusion
			0-20 cm	20-50 cm	0-20 cm	20-50 cm	
1	12	1	Yes	Yes	-	-	Stationary
2	4	10	No	No	Downward	Downward	Downward trend
3	4	1	Yes	Yes	-	-	Stationary
4	0	0	No data	No data	No data	No data	Inconclusive
5	0	0	No data	No data	No data	No data	Inconclusive
6	0	0	No data	No data	No data	No data	Inconclusive
7	0	0	No data	No data	No data	No data	Inconclusive
8	10	1	Yes	Yes	-	-	Stationary
9	3	7	No	No	Upward	Downward	Not stationary
10	3	10	No	No	Downward	Downward	Downward trend
11	5	8	Yes	Yes	-	-	Stationary
12	6	12	No	No	Upward	Upward	Upward trend
13	4	1	Yes	No data	-	No data	Stationary
14	5	2	Yes	No data	-	No data	Stationary
15	0	0	No data	No data	No data	No data	Inconclusive
16	6	1	Yes	No data	-	No data	Stationary
17	0	0	No data	No data	No data	No data	Inconclusive
18	7	1	Yes	No data	-	No data	Stationary
19	3	0	Yes	No data	-	No data	Stationary
20	4	2	Yes	No data	-	No data	Stationary
21	3	0	Yes	No data	-	No data	Stationary
22	4	1	Yes	No data	-	No data	Stationary
23	4	0	Yes	No data	-	No data	Stationary
24	3	2	Yes	No data	-	No data	Stationary
25	4	3	Yes	No data	-	No data	Stationary

Comparison between approaches

Final results are shown in Table 38. The combination of results from both methods yields conclusive classifications for each trial. Then, conclusions for each crop are drawn from generally appreciating results for corresponding trials.

Table 38 – Conclusions on the stationarity of soil carbon in Portuguese soils, per crop.

Trial #	Conclusion		Final conclusion, by trial	Crop	Final conclusion, by crop
	T Test	Time Series			
1	Inconclusive	Stationary	Stationary	Olive	Depends on the trial **Probably: Stationary**
2	Not stationary	Downward trend	Downward trend		
3	Inconclusive	Stationary	Stationary		
4	Downward trend	Inconclusive	Downward trend		
5	Downward trend	Inconclusive	Downward trend		
6	Downward trend	Inconclusive	Downward trend		
7	Upward trend	Inconclusive	Upward trend		
8	Inconclusive	Stationary	Stationary		
9	Inconclusive	Not stationary	Not stationary	Vineyard	**Depends on the trial**
10	Not stationary	Downward trend	Downward trend		
11	Inconclusive	Stationary	Stationary		
12	Inconclusive	Upward trend	Upward trend		
13	Inconclusive	Stationary	Stationary	Fall/Winter grains	**Stationary**
14	Inconclusive	Stationary	Stationary		
15	Downward trend	Inconclusive	Downward trend		
16	Inconclusive	Stationary	Stationary		
17	Upward trend	Inconclusive	Upward trend	Maize and rice	Stationary/upward trend **Probably: Stationary**
18	Inconclusive	Stationary	Stationary		
19	Upward trend	Stationary	Stationary/Upward trend		
20	Inconclusive	Stationary	Stationary		
21	Inconclusive	Stationary	Stationary		
22	Stationary	Stationary	Stationary		
23	Upward trend	Stationary	Stationary/Upward trend		
24	Inconclusive	Stationary	Stationary		
25	Stationary	Stationary	Stationary		

Conclusions - olive

Conclusions are that, for olive, the stationarity of SOC is dependent on the trial. Note however, that trials that point to a downward trend in soil carbon stocks are:

- Trial 2 – the conclusion was obtained from 4 sampling years and 10 interpolated years. Since so many values were interpolated, results must be taken with caution;
- Trials 4, 5 and 6 – there are only 2 sampling years in each trial, and therefore conclusions must be influenced by the fact that we are comparing only two specific years. However, trial 7 yields an upward trend only based on 2 sampling years as well.

Therefore, the two most reliable trials are numbers 1 and 8. Both of them yield stationarity as the most likely conclusion.

Final conclusion: SOC is probably stationary, but results depend on the trial and number of sampling years.

Conclusions - olive

Conclusions are that, for vineyard, the stationarity of SOC is dependent on the trial. In this case, results that point for no stationarity cannot be dismissed easily, since they are corroborated by both methods.

Final conclusion: No conclusion may be drawn, since results depend on the trial.

Conclusions - Fall/Winter grains

In the case of Fall/Winter grains, there is only one trial (trial number 15) that yields a downward trend for SOC dynamics. However, this trial consists only of two sampling years, and therefore its conclusion is relatively weak and may be dismissed. All other trials point to stationarity.

Final conclusion: SOC is stationary.

Conclusions - maize and rice

In the case of maize and rice, all trials point to either stationarity of SOC accumulation. However, since stationarity is the conclusion for most trials, we consider that as the most likely result.

Final conclusion: SOC is either stationary (which is the most likely result) or increasing.

Average carbon values per crop

We have previously determined that the average values for olive, Fall/Winter crops and maize/rice are constant, regardless of sampling year. This means that SOC is constant for croplands remaining croplands with each type of crop.

Therefore, the average soil carbon stock for each crop may be obtained by calculating the average of all values for each crop. Results are shown in Table 39. Maize and rice have the highest SOC stocks, followed by olive. Fall/Winter grains have the lowest stocks.

Table 39 – Average Soil Organic Carbon stock per crop.

Crop	Depth (cm)	Soil organic carbon (%)
Olive	0-20	0.93
	20-50	0.64
Vineyard[39]	0-20	0.98
	20-50	0.92
Fall/Winter grains	0-20	0.54
	20-40	0.58
Maize and rice	0-20	1.54
	20-40	1.71

[39] Note that averages for vineyard have no statistical meaning, since we could not prove that soil organic carbon stocks are stationary.

3.4 Complementing the assessment – the case of irrigated pastures

All studies conducted in this thesis refer to rainfed pastures[40]. Even though those are the majority, as shown in Table 1 for the case study of Project Extensity, irrigated pastures gain a particular importance as an alternative to crop production. In a parallel study (Valada *et al.*, 2008, 2010) done during this thesis, the life cycles of two alternative land uses were studied:

1. Irrigated land is used for maize production. Maize is then used as a raw material for the production of ethanol. DDG, a dry distilled grain, is a by-product that can be used to feed cattle as a substitute for soybeans in concentrated feeds. Because an area is occupied with maize, it cannot be grazed by cattle, which remain in stables. Bioethanol is then used in cars' motor combustion. We consider that the CO_2 released was previously sequestered by maize.

2. Sown Irrigated biodiverse permanent Pastures, or SIP for simplification, are installed. SIP are then grazed by cattle. There is direct substitution of bioethanol and gasoline. Therefore, in this scenario, gasoline remains in use due to the non production of maize. Gasoline is used in car combustion, releasing as major pollutants CO_2, CO, NO_x and CH_4.

Since SIP benefit from having water available all year, they guarantee an increased and regular production throughout the year. Therefore, they are used for steer feed. This means that they bring out all the benefits from transferring steers from intensive to extensive production, as we referred in Section 3.2.2.9.

Those scenarios were evaluated in an LCA conducted with software SimaPro 6.0. From all the impact categories that were analysed, we present results here for two: greenhouse gas emissions and energy resources.

The functional unit chosen for the first scenario was 1 t of ethanol, which requires an area of 0.27 ha of maize production. The directly comparable functional unit for the second scenario is the quantity of gasoline that directly substitutes 1 t of bioethanol. That quantity is 0.72 t of gasoline. The same area of 0.27 ha is considered in the second scenario to be used for SIP. Both conventional tillage and no-tillage were considered for maize production. For no-tillage maize and for SIP, carbon sequestration in soils was considered.

Results for greenhouse gas (GHG) emissions are shown in Figure 22 (Valada *et al.*, 2008). If carbon sequestration is not included, the bioethanol production scenario is responsible for only slightly less GHG emissions than the gasoline scenario. However, bioethanol fuels benefit from a tax reduction. If that tax reduction was only motivated by the principle that bioethanol is responsible for low CO_{2e} emissions, at the very least our results show that it would be a very expensive policy. Each ton of CO_{2e} gained from the substitution of gasoline would cost about € 100 to the Portuguese state. Furthermore, if carbon sequestration is included, the SIP scenario gains advantage. Therefore, we concluded that SIP is a better land use for GHG emissions.

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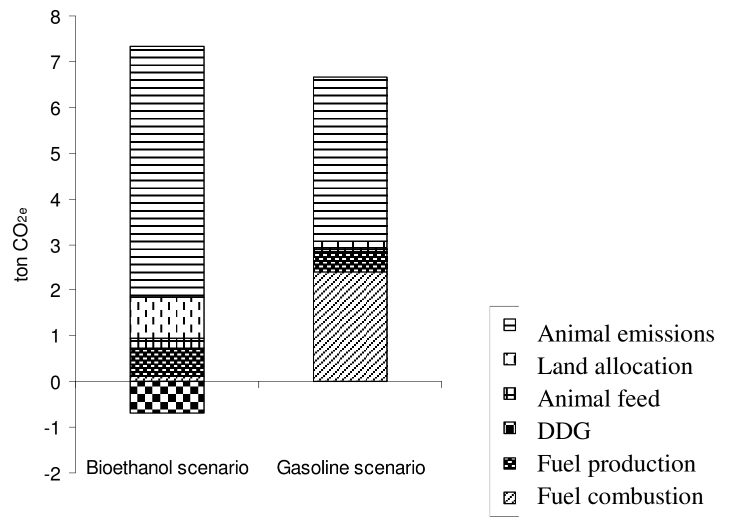
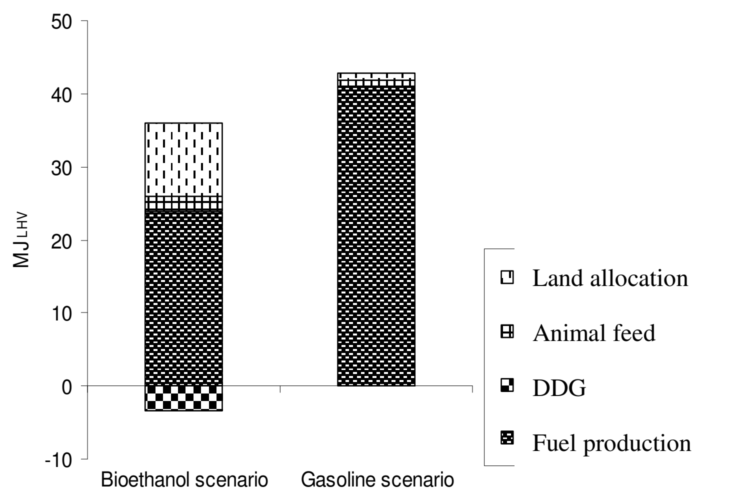
[40] For details on this study, see Valada *et al.* (2008).

Figure 22 – Results for GHG emissions of the bioethanol scenario (maize production) and the gasoline scenario (sown irrigated pastures).



Regarding energy resources (Figure 23), more energy is always necessary in the life cycle of gasoline production than in the bioethanol scenario. Our results (Valada *et al.*, 2008) show that, as an energy policy, it is favourable to produce maize for bioethanol.

Figure 23 – Results for energy resources of the bioethanol scenario (maize production) and the gasoline scenario (sown irrigated pastures).



These results complement the ones shown in this thesis for rainfed pastures. We compared rainfed pastures with the alternative land use, which are natural pastures. Irrigated pastures have to be compared with their alternative land use, which at the time of the study was maize for bioethanol production. And when we do so, we see that, at least in some impact categories, and considering both direct and indirect impacts[41] that result from the land use choice, SIP are also a positive option.

This case of SIP raises the question of whether the study of SBPRL should remain only concerning direct effects, or whether the whole life cycle would yield different conclusions regarding their positive environmental effects. In the next section, we perform the life cycle assessment of environmental impacts of SBPPRL and NG.

3.5 Life cycle assessment of pastures

One method particularly suited for the environmental assessment of indirect impacts is Life Cycle Assessment (LCA). The evaluation framework most commonly applied in LCA involves the steps shown in Table 40 (Tukker, 2000).

Table 40 – Theoretical approach of Life Cycle Assessment.

Step	Theorical approach
Goal and scope definition	Definition of: • the purpose of the study; • the functional unit.
Inventory	A system is defined that includes all relevant process chains of the product function in question: • manufacture; • use; • waste management. For each process in this chain, the relevant environmental interventions are inventoried in relation to the process' contribution to the central product function. Interventions caused by each process that is part of the system concerned are added by intervention type. The final result is a list of all environmental interventions associated with the product's function. The list is known as the inventory table.
Impact assessment (Classification, characterization, and valuation)	Aggregate the information obtained in the inventory. First, a classification of impact categories is chosen, usually reflecting a common mechanism of environmental threat (e.g., global warming, acidification, and ozone depletion). In the characterization step, the environmental interventions listed in the inventory table are translated into scores on each impact category. The result of this operation is called the impact profile of a functional unit of a product. In principle, the impact categories can be aggregated, by means of weighting, to give a single score for the environmental impact of a product.

Source: Tukker (2000).

The LCA done in this thesis recurred to software commonly used in such studies, namely SimaPro 6.0, which was developed by the National Reuse of Waste Research Programme and Pré Consultants of the Netherlands. The use of software is a precious

[41] Note that some indirect impacts come up from the consideration of causal effects related to the choice only by economic reasons – for example, emissions from livestock in intensive production in the maize scenario. In this sense, the analysis conducted was similar to the one we did in this thesis. It also included the same elements of sustainability assessment explained in the first Chapter.

time saver, since the inventory and impact assessment methods are already included therein. The main advantages of using LCA software such as SimaPro are the speed of assessment and the fact that its data base is very wide-ranging, since all existing inputs for any given activity are considered.

However, many processes are country-specific, and the fact that it uses a foreign database is a strong limitation. Its application must be carefully planned, since SimaPro does not possess a thorough database of impacts, including specific impacts for Portugal. It is always necessary to complement the inventory with country-specific data. Therefore, we decided to use SimaPro as the basis, but we incorporated national information whenever it was available or the impact resulting from the process was significant. Results were calculated iteratively – after a first run, most significant impacts were characterized again using country-specific data.

To perform an assessment of the pastures' life cycle, we divided the work in three stages (Teixeira *et al.*, 2008):

1. LCA of feed ingredients;
2. LCA of commercial feeds;
3. LCA of NG and SBPPRL.

In each level, the LCA done in the level before is included is crucial. The impacts of feed ingredients must be determined to obtain the impacts of whole feeds, and animal feed is required when studying the impacts of the whole pasture system. Each part will be explained in detail next.

3.5.1 SimaPro and the Ecoindicators

SimaPro works according to ISO 14000 environmental management standards. It consists of a data base of inputs and outputs from several processes and production of materials. Therefore, the assessment of environmental impacts consists in the sum of impacts from each step of its life cycle (inventory stage). Impacts are then added by environmental themes. The total impact in each theme is then normalized and aggregated into a single impact indicator, usually using one of two methods: "Ecoindicator 95" (EI95) and "Ecoindicator 99" (EI99) (impact assessment stage). Both methods aggregate impacts into a subjective and abstract unit called "Ecoindicator Point", or Pt. Even though conclusions drawn are often similar (Luo 2001), it is important to use both, as they present different themes and a different conception.

EI95 classifies, characterizes and normalizes the environmental impacts based on their contribution to several themes (Luo, 2001). The environmental aspects related to a given product are first aggregated into a number of effects caused, and those are then characterized according to the degree of damage inflicted on the environment; finally, these results are normalized into a single score, based on subjective evaluation (Goedkoop, 1998). The environmental impact themes in EI95 are: emissions of greenhouse gases, heavy metals and carcinogens; substances causing ozone layer destruction, acidification, eutrophication, winter smog and summer smog; and consumption of energy resources[42].

EI99 is an update and extension of EI95, which emphasises its damage-oriented methodology by considering three areas of environmental damage: human health (measured in DALY – Disability Adjusted Life Years), ecosystem quality (expressed as PAF – Potentially Affected Fraction and PDF - Potentially Disappeared Fraction) and resource depletion (expressed as $MJ.kg^{-1}$) (Luo, 2001; Goedkoop and Spriensma, 2000).

3.5.2 LCA of feed ingredients

The most basic unit that comes into animal feed is the ingredient of a commercial feed. First, we determined which are the main ingredients in commercial feeds. To do so, we spoke to farmers from Project Extensity and searched INE's database and the Portuguese Association of Producers of Commercial Feeds for Animals' year book (IACA, 2004). At the time of the study, we came up with the following list:

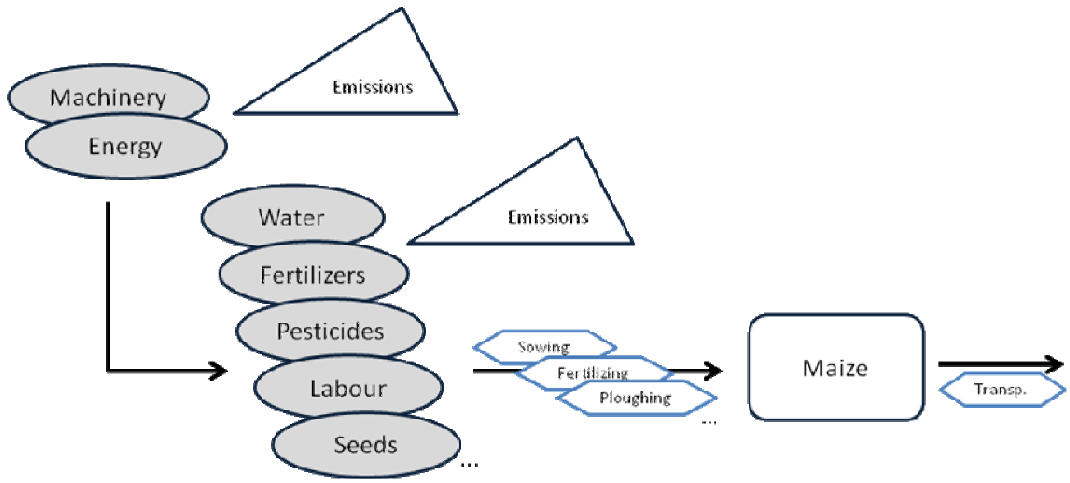
- Forages – wheat straw, silage maize;
- Cereal crops – grain maize, grain wheat, barley;
- Oilseeds – soybeans, sunflower;
- Minor by-products – DDG, corn gluten feed, palm kernel cake.

One LCA was done for each of these ingredients in each location where national consumption is mostly produced. One thorough study (Teixeira *et al.*, 2007) was done to serve as basis for the others. This study was done to compare different locations where maize could be produced, as well as techniques to optimize its environmental impact (such as no-tillage). The study is presented in full in Appendix III – Environmental analysis of maize production. In the context of this thesis, the most important part is to show how results were obtained.

First, we defined the functional unit, which is 1 t of maize, and defined the borders of the life cycle, as shown in Figure 24. We chose the most productive zones and characterized the production systems using the technical coefficients from crop fact sheets in GPP (2001). Then, every product entry (fertilizers, chemicals, …) and every action (sowing, ploughing, …) considered in the fact sheets were simulated using SimaPro 6.0. One international provenience was included, namely Argentina (the main maize selling country to Portugal at the time).

[42] Two more themes are available, namely solid waste and pesticides, but there is rarely enough information to calculate them.

Figure 24 – LCA system studied for maize production.



As mentioned before, the work was done iteratively, and therefore in each run we determined the steps in the life cycle responsible for the most impacts, and redone them using more accurate inventory data. For example, our results show that fertilization is the operation responsible for most of the environmental impact of maize production in all locations (around 70%). Therefore, we had to correct air and water emissions from fertilization operations, using data from van der Werf *et al.* (2005). One particularly crucial parameter was nitrogen leaching. In this case, and not being able to obtain any Portuguese studies on the issue, we turned to studies carried out in relatively similar conditions in Spain.

Engström *et al.* (2007) indicate that the most important environmental themes for Swedish agriculture are eutrophication, global warming and resource use. Our analysis confirms that these themes are important, but indicates some others of interest, like acidification and heavy metals.

The importance of heavy metals is striking. But it may be understood if we follow this thinking line. SimaPro allocates the impact of building the machinery used to the production in which it intervenes. Agriculture is an overcapitalized industry. Unlike other types of machinery (industrial, private transportation vehicles), agricultural machinery is used for a relatively small time frame, and only in a very specific time of year. Therefore, costs and inputs of machinery building and use must always be considered, since its impact is comparable to that of maintenance and fuel consumption. Furthermore, in Portugal, recycling or reuse is not necessarily the final destination of materials, and emissions may be aggravated by lack of adequate final destination. For example, in the case of irrigation, machinery needed stands for 46% of its heavy metal emissions, while electricity consumption stands for 38%.

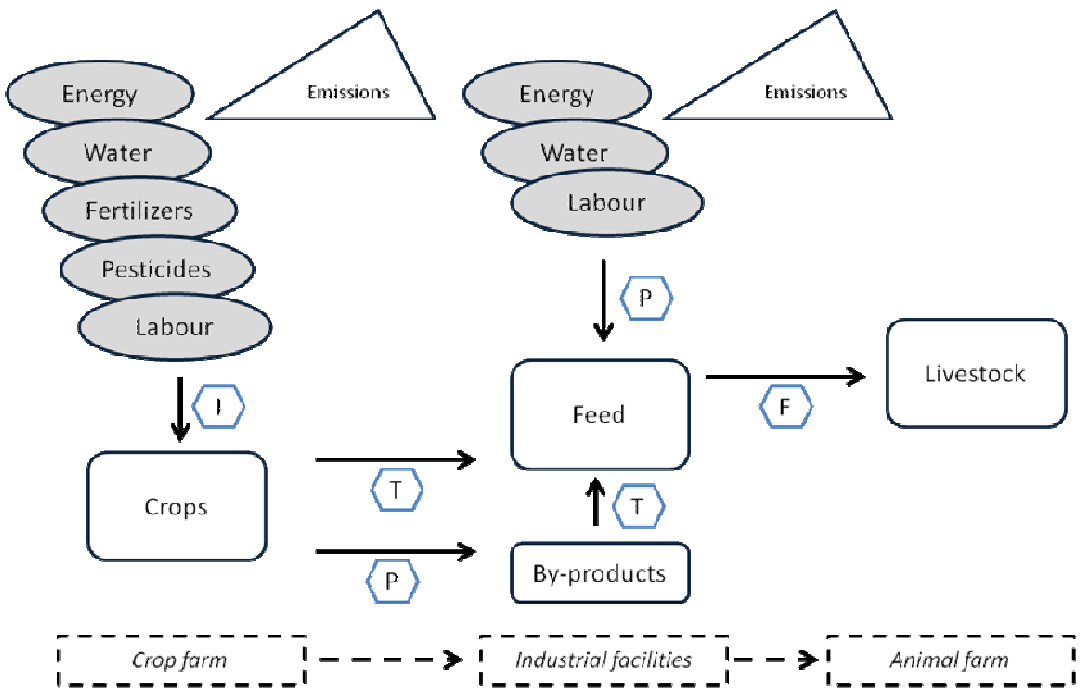
The impact of heavy metals may also be explained by fertilizer use. Fertilizers are currently the main sources of cadmium emissions to the soil, which is an important problem in The Netherlands, where the method was developed. The average European value is 3.8-6.8 $g.ha^{-1}$ in crop land, whereas in The Netherlands values are as high as 7.5-8.5 $g.ha^{-1}$ (Ferrão, 1998). For example, in BI (where the impact on heavy metals is the highest), we found that over 35% of the impact comes from fertilizing. Irrigation also has a very significant part (over 25%).

After calibrating the method for maize produced in many regions, we moved on to all the other ingredients. The last step was to aggregate them all in typical commercial feeds. That's what we did next.

3.5.3 LCA of commercial feeds

For commercial feeds, another specific LCA was conceptualized, as shown in Figure 25. There are four major steps in this life cycle, namely ingredient production (I), transportation of ingredients to an industrial facility (T), feed processing at the facility (P), and transportation of processed feed to the animal farm (F).

Figure 25 – Commercial feed's life cycle scheme.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Therefore, the impact of each feed is:

$$I_{feed} = \sum_{i=1}^{n} I_i + \sum_{i=1}^{n} T_i + P_{feed} + F_{feed}, \quad (3.23)$$

where each capital letter represents the impact of each stage of the life cycle, and i is each ingredient. Four different functional units were tested in this assessment, which were 1 kilogram of:

- feed (total mass);

- crude protein in the feed;
- digestible energy in the feed;
- crude fiber in the feed.

Three feeds were analysed:

1. A grain maize-based feed (Feed 1), which is an average typical formulation for intensive production, since it contains no complement with green forages (only wheat straw);
2. A silage maize-based feed (Feed 2), which is mostly based on silage maize, which is a green forage, mixed at the farm – it is the most "extensive" feed;
3. The average national feed, which is the average of all feed ingredients used in Portugal, and was obtained from the fodder balance (IACA, 2004).

Table 41 gives the composition of each feed. The first type of feed is a conventional maize grain-based feed. It is given to the animal in a quantity of 2.5% of its live weight per day, plus a constant amount of 1.5 kg of wheat straw (Alfredo Sendim; personal communication). The second type of feed is maize silage-based. The animals are fed with 3% of their live weight of this composition per day (Alfredo Sendim; personal communication). Since many feeds are used for each type of animal and age, it is impossible to use a given feed composition as representative (Fernando Anjos; personal communication). The average national feed for finishing calves was obtained from the fodder balance, and so there is a large uncertainty in its composition. This feed should only be considered as a benchmark for the other two, serving as a control.

The animals are fed from the age of 6 to 8 months (180 to 200 kg live weight) to the age of 12 to 14 months[43] (360 to 400 kg live weight), and considering the average value in each of those, it may be shown that the first type of feed, which provides a fixed quantity of straw and 2.5% of the animal's live weight, is given to the animal in larger amounts in the beginning, and smaller in the end, than the second type of feed, which varies the quantity of straw within the base feed (Alfredo Sendim; personal communication).

[43] Note that this is true for organic production. By choosing this specific case, we are using a adopting approach. There are many cases of intensive production in which livestock is not slayed until 24 months of age (Alfredo Sendim; personal communication).

Table 41 – Composition of the feeds studied.

	% (kg ingredient.100 kg^{-1} feed)		
Ingredient	**Feed 1**	**Feed 2**	**Average feed**
Maize (silage)		58.6	
Maize (grain)	20.0	12.5	13.7
Corn Gluten Feed	20.0	6.6	27.0
Wheat (grain)	19.0	6.6	4.9
Barley	10.0		2.1
Soy meal (44% protein)	6.0	9.1	16.9
Manioc	5.0		
DPG	5.0		
Palm kernel cake	5.0		
Sunflower	4.8		
Carbonate	2.1		
Fats	0.9		
Bicarbonate	0.8		
Salt	0.6		
Premix	0.5	0.8	
Urea	0.3		
Others			36.3[44]
Wheat (straw)	(1.5 kg.day^{-1})[45]	5.8	

Here, we considered two intervals of 2.4 months: (1) from 7.2 to 9.6 months, when the animals are fed in 60% by commercial feed and 40% from pasture; (2) from 9.6 to 12 months, animals are confined to stables and completely fed with the commercial feed. Feed 1 and the average feed allow the animals to grow more rapidly, at the rate of 1.5 kg.day^{-1}, whereas Feed 2 provides a slower growth rate of 1.2 kg.day^{-1}. Therefore, animals fed with Feed 2 end the second period with less weight than those fed in the other cases (Table 42) (Alfredo Sendim; personal communication).

Table 42 – Animal weights when fed with each feed.

Age (months)	Weight 1 and average (kg)	Weight 2 (kg)
7.2	190.0	190.0
9.6	298.0	276.4
12	406.0	362.8

Again, we used SimaPro 6.0 and an iterative approach to results, correcting the inventory for the most significant impacts. For by-products, we also considered two impact allocation methods – mass allocation, and economic value allocation.

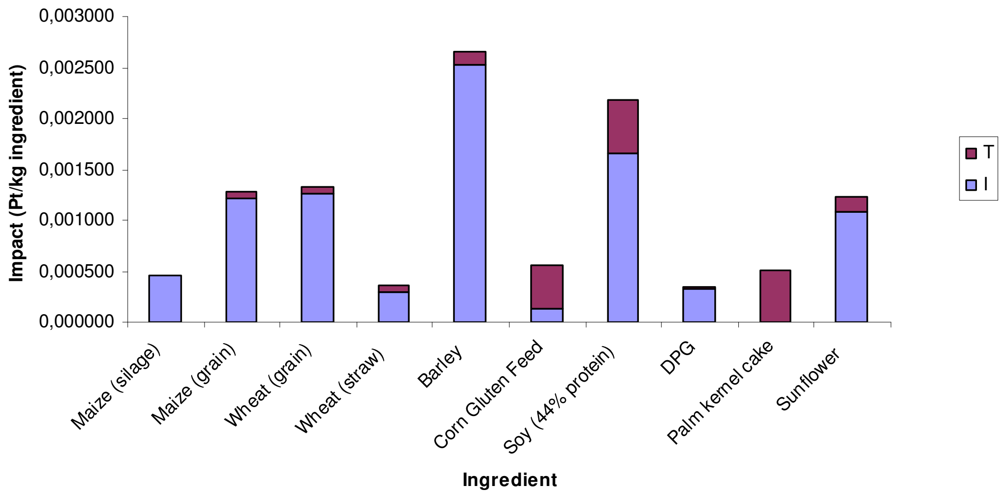
The whole study is presented in Appendix IV – Environmental analysis of concentrated feeds. Returning to individual feed ingredients, results show that by-products have lower environmental impacts. Barley and soybeans are the ones with higher impacts per kilogram, as shown in Figure 26. Barley also has the worst environmental impact per kilogram of crude protein, crude fiber and digestible energy. Soy, on the other hand, is

[44] The “others” are undisclosed cereals and by-products. Due to lack of information, it was not possible to determine what they are.

[45] Straw is given to the animals in a fixed quantity, which does not depend on the quantity of feed also given. It could not be determined whether the average feed contains straw, but since its fibre content is equal to that of Feed 1 (as shown next) it is plausible to assume that it does not (straw is mainly used for fibre).

an efficient provider of protein. Silage maize has a low impact per kilogram and per unit of crude fiber, but is not an efficient source of protein or energy. Wheat straw and by-products are efficient in all functional units.

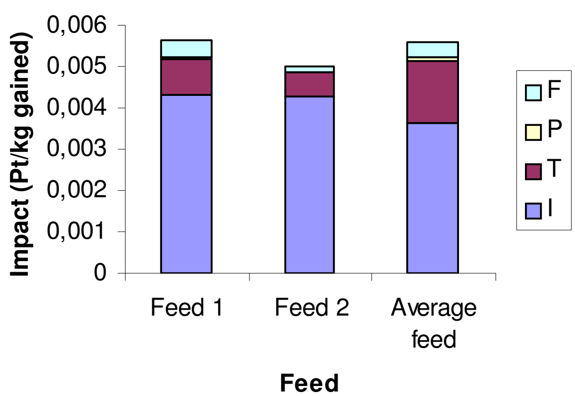
Figure 26 – Unit impact of per kilogram of each of the ingredients (Ecoindicator 95).



I – Impacts of ingredient production; T – impacts of transportation of ingredients.

As for the impact of aggregated feeds, shown in Figure 27, we found that each kilogram of the silage maize-based feed (“extensive” feed) has a lower aggregated environmental impact than the grain maize-based feed (“intensive” feed). Feed 1 has, in fact, about the same impact as the average feed. We can also see from Figure 27 that the production of the ingredient is the stage of the life cycle with higher impacts, but its transportation to the industrial facility cannot be neglected.

Figure 27 – Overall impact of each feed ($Pt.kg^{-1}$ gained), Ecoindicator 95.



I – Impacts of ingredient production; T – impacts of transportation of ingredients; P – impacts of feed processing at the facility; F – transportation of the feed to the animal farm.

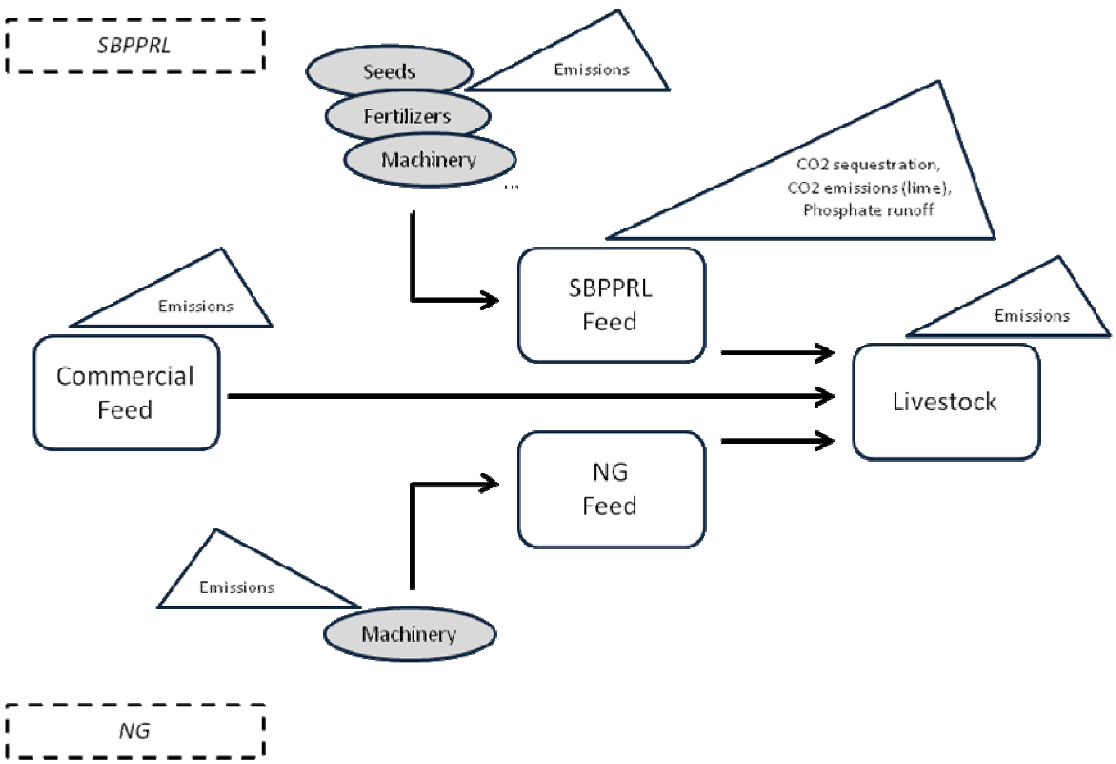
However, each of these feeds do not provide the same amounts of protein, fiber and energy to cattle (although all provide an acceptable minimum of each). Feed 2 is very rich in fiber, while Feed 1 and the average feed are slightly richer in terms of protein and energy. For this reason, all feeds have about the same environmental impact per unit of crude protein and digestible energy, while feed 2 has a lower environmental impact per unit of crude fiber.

Now that we know which feeds are typically used and their impacts, we can turn to the life cycle of pastures as a whole.

3.5.4 LCA of grassland systems

A whole LCA study is required for SBPPRL and NG due to the fact that one is more intensive than the other. SBPPRL require a wider array of inputs than NG: seeds, fertilizers, and machinery. In a way, they are a return to a pre-labor extensive systems situation. Major differences between animal production in sown and natural grasslands are shown in Figure 28.

Figure 28 – Life cycles of animal production in natural (baseline scenario) and sown (proposed scenario) grasslands.



NG – Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; CO_2 – Carbon dioxide.

The impact of 1 t of feed was determined in the previous section. Those feeds, however, are meant for steers. Nevertheless, we may consider that the silage maize-based feed is appropriate for grazing cows. This is the feed with the lower impact from those we studied before, and therefore it is also the most favorable for the NG scenario.

Therefore, we guarantee that we do not bias results towards decreasing the SBPPRL system impacts (where there are less or none commercial feeds).

The impact of 1 ha of SBPPRL and NG can be calculated using software SimaPro 6.0, using the inputs in the life cycle in Figure 28. For SBPPRL, we use the sheets that will be shown in the next Chapter, in Table 59 for installation and Table 60 for maintenance. One tenth of the installation impacts were added to the maintenance impacts, since the expected time frame of SBPPRL is 10 years. The impact of one generic legume seed in SimaPro's database were used for SBPPRL seeds. This is a simplification, since the impact of the production of the seeds is very different from species to species. For NG, only one tillage every five years was considered. Therefore, one fifth of the impact of one tillage operation was calculated to find the impact of NG.

We use two scenarios for SBPPRL: one where 100 kg of phosphate fertilizer (Single Superphosphate) are added yearly for maintenance, and one where 200 kg are added. The 200 kg are the worst-case scenario possible, corresponding to the worst levels of P in soils prior to installation. Typically, less P is needed, and after a while the operation isn't required every year. Therefore, the 100 kg scenario is included for comparison purposes. Even though the same uncertainty in quantities exsists for other inputs and operations, phosphate fertilizer is the main source of impact, as we will show next.

Results are shown in Table 43. Each hectare of SBPPRL has, naturally, higher impacts than each hectare of NG. This is due to the fact that NG only require one tillage operation around every 5 years, while SBPPRL require a wide array of operations and inputs. The only impact category where SBPPRL have a better result than NG is GHG emissions, measured as CO_2e, due to carbon sequestration.

Table 43 – LCA impacts of 1 ha of SBPPRL and NG and 1 t of feed in each impact category.

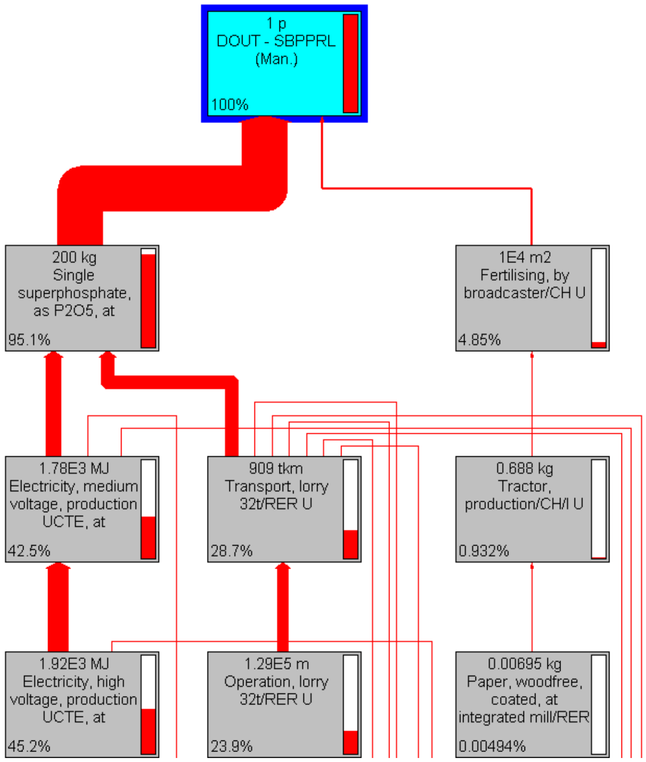
Impact category	[Unit]	$\{I\}^{SBPPRL}$ [Unit].ha^{-1}		$\{I\}^{NG}$	$\langle I\rangle^{feed}$
		100 kg.yr^{-1} phosphote	200 kg.yr^{-1} phosphote	[Unit].ha^{-1}	[Unit].t^{-1}
GHG	kg CO2	-3.9×10^{3}	-3.6×10^{3}	1.1×10^{1}	3.5×10^{2}
Ozone layer	kg CFC11	4.7×10^{-5}	7.4×10^{-5}	2.1×10^{-6}	3.5×10^{-5}
Acidification	kg SO2	5.7×10^{0}	9.9×10^{0}	9.2×10^{-2}	4.1×10^{0}
Eutrophication	kg PO4	1.2×10^{0}	1.9×10^{0}	1.6×10^{-2}	1.0×10^{0}
Heavy metals	kg Pb	2.0×10^{-2}	3.4×10^{-2}	4.7×10^{-4}	3.9×10^{-3}
Carcinogens	kg B(a)P	6.6×10^{-5}	9.7×10^{-5}	5.4×10^{-6}	2.8×10^{-5}
Winter smog	kg SPM	4.3×10^{0}	7.7×10^{0}	3.2×10^{-2}	6.8×10^{-1}
Summer smog	kg C2H4	1.5×10^{-1}	2.3×10^{-1}	7.8×10^{-3}	9.5×10^{-2}
Energy resources	MJ LHV	5.5×10^{3}	9.2×10^{3}	7.7×10^{2}	3.1×10^{3}

I – Impact; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; NG – Natural Grasslands; GHG – Greenhouse Gases; LHV – Lower Heating Value; LCA – Life Cycle Assessment.

Almost the totality of the impact of SBPPRL in every category is due to phosphorus fertilizer production, transport and application. While several studies exist on the environmental effects of the application of phosphorus (Blake *et al.*, 2000), our study indicates that it is the life cycle impacts involved in its production that are responsible for the most part of the impacts. This is particularly visible in the acidification and energy resources themes, as shown in Figure 29 for energy resource consumption.

Production of the fertilizer is responsible for 95% of all the energy use for the maintenance of SBPPRL. On many farms, however, fertilizers are applied and farmers continue to grow crops.

Figure 29 – Main sources of energy resorce consumption in the life cycle of SBPPRL, according to SimaPro 6.0.



In most impact categories, each unit of synthetic phosphate has less environmental impacts than, each unit of nitrogen fertilizer (less GHG emissions, for example), except for the themes acidification, eutrophication and winter smog, as shown in Table 44. Therefore, SBPPRL suppress the need for N fertilizers, which is also positive for GHG reduction, but require phosphate, which causes impacts in other themes.

Table 44 – Comparison of LCA impacts between a nitrogen fertilizer and a phosphate fertilizer.

Impact category	[Unit]	Ammonium nitrate [Unit].kg^{-1} N	Urea [Unit].kg^{-1} N	Single superphosphate [Unit].kg^{-1} P_2O_5	Triple superphosphate [Unit].kg^{-1} P_2O_5
GHG	kg CO2	7.8×10^{0}	3.0×10^{0}	2.5×10^{0}	1.9×10^{0}
Ozone layer	kg CFC11	4.1×10^{-7}	4.2×10^{-7}	2.7×10^{-7}	1.9×10^{-7}
Acidification	kg SO2	2.7×10^{-2}	1.3×10^{-2}	4.2×10^{-2}	3.3×10^{-2}
Eutrophication	kg PO4	4.4×10^{-3}	1.9×10^{-3}	6.5×10^{-3}	4.2×10^{-2}
Heavy metals	kg Pb	5.9×10^{-5}	4.0×10^{-5}	1.4×10^{-4}	2.4×10^{-4}
Carcinogens	kg B(a)P	3.4×10^{-7}	3.2×10^{-7}	3.1×10^{-7}	1.9×10^{-7}
Winter smog	kg SPM	6.4×10^{-3}	5.2×10^{-3}	3.4×10^{-2}	2.9×10^{-2}
Summer smog	kg C2H4	5.8×10^{-4}	5.8×10^{-4}	8.2×10^{-4}	4.8×10^{-4}
Energy resources	MJ LHV	5.6×10^{1}	6.4×10^{1}	3.7×10^{1}	2.7×10^{1}

GHG – Greenhouse Gases; LHV – Lower Heating Value; LCA – Life Cycle Assessment.

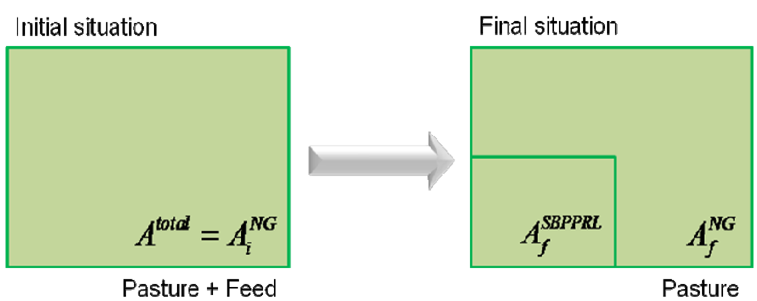
Still, as we have noticed before, the direct comparison between each hectare of SBPPRL and NG is of little interest, since it is not a complete life cycle. The NG system's life cycle is only complete considering feeds. And so we move on to determining the overall impacts of each scenario.

3.5.4.1. How to calculate the LCA impacts with the available data?

However, as may be suggested by Figure 28, it does not suffice to calculate, in one scenario, the impact of 1 ha of SBPPRL, and compare it, in the other scenario, with the impact of 1 ha of NG and corresponding substituted feed. This approach would be wrong because it does not maintain the livestock balance. In such a situation, there is no guarantee that the same number of animals are being fed in the two scenarios. Remember, for example, the case of the field plots used so far, where livestock units in SBPPRL and NG were the same cattle grazes both plots, differing only on the number of days it is kept on each (and therefore the feed is the same).

A more likely approach is to consider two scenarios regarding two moments in time (Figure 30). In the beginning, before installing SBPPRL, all plots in the farm are NG (we hereby denote all quantities regarding the initial simulation with the subscript *i*). Cattle grazes and consumes commercial feeds. Then, farmers starting installing SBPPRL. We assume that they install as many as necessary to suppress the need for concentrated feeds, or the whole area of the farm, whichever is smaller (quantities referring to the final simulation are denoted with the subscript *f*). This assumption is useful for calculations, since the amount of feeds provided by farmers with NG and SBPPRL is unknown.

Figure 30 – Scenario for substitution of feed by increased production in SBPPRL.



A – Area; i – initial; f – final; NG – Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.

In this scenario, several balances may be defined. All calculations are done for the average hectare in the farm. The first one is the area balance, which states that the total area of the farm (A^{total}) is equal to the area of NG in the beginning (A_i^{NG}), and to the sum of the area of NG in the end (A_f^{NG}) plus the area of SBPPRL installed (A_f^{SBPPRL}):

$$A^{total} = A_i^{NG} = A_f^{NG} + A_f^{SBPPRL}. \tag{3.25}$$

The second balance is the livestock balance, which simply states that, since SBPPRL replace cattle needs for commercial feeds, the number of livestock units (LU) in constant (no animals enter or leave the farm):

$$LU^{total} = LU_i = LU_f \ . \tag{3.26}$$

Given the previous condition, the energy requirements in the animal feed in the beginning and in the end are the same, and therefore, there is an energy balance:

$$\begin{aligned} LU\left(\frac{EFU}{LU}\right)^{total} &= A_i^{NG}\left\{EFU\right\}^{NG} + A_i^{NG}\left\{EFU\right\}^{feed} \\ &= A_f^{NG}\left\{EFU\right\}^{NG} + A_f^{SBPPRL}\left\{EFU\right\}^{SBPPRL} \end{aligned}, \tag{3.27}$$

where $\{EFU\}$ are the forage units, measured as digestible energy, per hectare, respectively, of the commercial feed and the feed grazed in pastures by the cattle. Regarding the feed, however, the energy content should be assessed per unit of mass. In order to do so, a conversion is required:

$$\left\{EFU\right\}^{feed} = \frac{\left\{EFU\right\}^{feed}}{\left\{M\right\}^{feed}}\left\{M\right\}^{feed} = \left\langle EFU\right\rangle^{feed}\left\{M\right\}^{feed}, \tag{3.28}$$

where $\{M\}$ is the quantity of feed required yearly per area unit of NG in the initital situation.

The total impact of the initial situation is the sum of the impact of NG and the feed in the area of the farm:

$$\left\{I\right\}_i^{total} = A_i^{NG} \cdot \left(\left\{I\right\}^{NG} + \left\{I\right\}^{feed}\right), \tag{3.29}$$

where $\{I\}$ is the environmental impact per hectare, respectively, of the commercial feed and the feed grazed in pastures by the cattle. As before, the energy content of the feed should be assessed per unit of mass. In order to do so, another conversion is required:

$$\left\{I\right\}^{feed} = \frac{\left\{I\right\}^{feed}}{\left\{M\right\}^{feed}}\left\{M\right\}^{feed} = \left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}. \tag{3.30}$$

The impact of the final situation is thus:

$$\left\{I\right\}_f^{total} = A_f^{NG}\left\{I\right\}^{NG} + A_f^{SBPPRL}\left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}. \tag{3.31}$$

Therefore, the difference in impacts between the two situations is:

$$\left\{I\right\}_f^{total} - \left\{I\right\}_i^{total} = A_f^{NG}\left\{I\right\}^{NG} + A_f^{SBPPRL}\left\{I\right\}^{SBPPRL} - A_i^{NG} \cdot \left(\left\{I\right\}^{NG} + \left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}\right). \tag{3.32}$$

Rearranging this expression, and using the area balance, we find that:

$$\left\{I\right\}_f^{total} - \left\{I\right\}_i^{total} = A_f^{SBPPRL}\left(\left\{I\right\}^{SBPPRL} - \left\{I\right\}^{NG}\right) - A^{total}\left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}. \tag{3.33}$$

Since we want to find the life cycle environmental impact of the installation of 1 ha of SBPPRL, this Equation must be divided by the area of SBPPRL:

$$\frac{\left\{I\right\}_f^{total} - \left\{I\right\}_i^{total}}{A_f^{SBPPRL}} = \left\{I\right\}^{SBPPRL} - \left\{I\right\}^{NG} - \frac{A^{total}}{A_f^{SBPPRL}}\left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}. \tag{3.34}$$

This Equation (3.34) shows that the difference in impact due to the installation of 1 ha of SBPPRL is equal to the impact of SBPPRL per hectare minus the impact of NG per hectare, and minus the impact of the commercial feed multiplied by the inverse of the fraction of the farm where, in the final situation, SBPPRL have been sown.

But in Equation (3.34) there are two missing terms. We do not have data for $\{M\}^{feed}$ (we do not know how much feed is given to cattle per hectare of NG, since there is a large variability and uncertainty in this term). We also do not know the fraction of the total area of the farm which is sown with SBPPRL (the area required to compensate the use of feeds). We define this fraction as x:

$$x \equiv \frac{A_f^{SBPPRL}}{A^{total}}. \tag{3.35}$$

Therefore, Equation (3.34) must be modified, in order to suppress this lack of data. We return to the energy balance in Equation (3.27), and use the area balance to modify it

$$\begin{aligned} &A^{total}\{EFU\}^{NG} + A^{total}\langle EFU\rangle^{feed}\{M\}^{feed} = \\ &= \left(A^{total} - A_f^{SBPPRL}\right)\{EFU\}^{NG} + A_f^{SBPPRL}\{EFU\}^{SBPPRL}. \end{aligned} \tag{3.36}$$

Dividing Equation (3.36) by the total area and using the definition of x, we find that:

$$\{EFU\}^{NG} + \langle EFU\rangle^{feed}\{M\}^{feed} = (1-x)\{EFU\}^{NG} + x\{EFU\}^{SBPPRL}. \tag{3.37}$$

Equation (3.37) may now be divided by the forage units of NG:

$$1 + \frac{\langle EFU\rangle^{feed}}{\{EFU\}^{NG}}\{M\}^{feed} = (1-x) + x\frac{\{EFU\}^{SBPPRL}}{\{EFU\}^{NG}}. \tag{3.38}$$

At this time, we may introduce a new variable, ε, which is the coefficient of forage production between SBPPRL and NG:

$$\varepsilon = \frac{\{EFU\}^{SBPPRL}}{\{EFU\}^{NG}}. \tag{3.39}$$

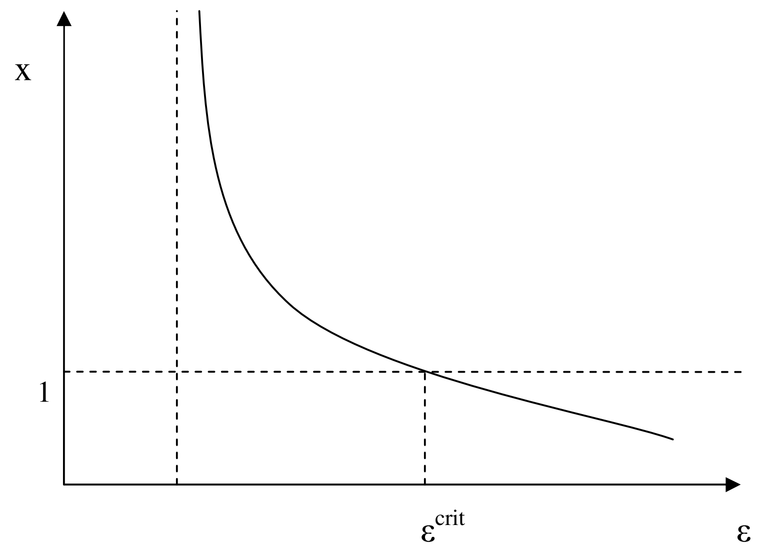
This coefficient is useful because, even though we do not know the energy content of both pastures, we can know their ratio, which is equal to the ratio between stocking rates in SBPPRL and NG plots. Field data was obtained in a situation where cattle grazed more days SBPPRL plots than NG plots, but was given the same quantity of commercial feeds. Therefore, we may assume that the ratio in stocking rates, both measured in grazing days, corresponds to the ratio in forage content of both types of pasture.

Using this coefficient in Equation (3.38), and solving for x:

$$x = \frac{1}{\varepsilon - 1}\frac{\langle EFU\rangle^{feed}}{\{EFU\}^{NG}}\{M\}^{feed}. \tag{3.40}$$

The graphical representation of Equation (3.40) is shown in Figure 31.

Figure 31 – Effect of the nutritional quality of SBPPRL (ε) on the fraction of the farm sown with SBPPRL (x).



There is an ε^{crit}, which is the value for which all area is converted to SBPPRL, and therefore is defined by x = 1, or, recurring to Equation (3.40):

$$\varepsilon^{crit} = 1 + \frac{\langle EFU \rangle^{feed}}{\{EFU\}^{NG}} \{M\}^{feed} . \tag{3.41}$$

Therefore, according to Equation (3.41), the critical value increases with the fraction of the animal feed that is assured by feeds in the initial case. For $\varepsilon < \varepsilon^{crit}$, there is no valid solution for x: the productivity of SBPPRL is not sufficient to completely eliminate the use of feed, within the available area. This means that, in this situation, all available area would be converted to SBPPRL, and feeds would still be required.

Returning to our goal, which was to calculate the difference in impacts between the final and the initial situation per unit of SBPPRL area, we may use the inverse of Equation (3.40) and substitute it in Equation (3.34), obtaining

$$\frac{\{I\}_f^{total} - \{I\}_i^{total}}{A_f^{SBPPRL}} = I^{SBPPRL} - I^{NG} - (\varepsilon - 1) \frac{\{EFU\}^{NG}}{\langle EFU \rangle^{feed}} \langle I \rangle^{feed} . \tag{3.42}$$

Using the definition of ε again, we find the final expression:

$$\frac{\{I\}_f^{total} - \{I\}_i^{total}}{A_f^{SBPPRL}} = \{I\}^{SBPPRL} - \{I\}^{NG} - \left(\frac{\varepsilon - 1}{\varepsilon} \right) \frac{\{EFU\}^{SBPPRL}}{\langle EFU \rangle^{feed}} \langle I \rangle^{feed} . \tag{3.43}$$

We have estimates available for all of the variables in Equation (3.43). We begin with LCA results of the environmental impacts.

3.5.4.2. <u>Determining the impacts of each scenario</u>

We have ε = 2 since, as we have shown before, stocking rate in SBPPRL is approximately double that of NG. The digestible energy content of 1 t of feed, and of 1 ha of SBPPRL, can be obtained in energy equivalency tables for feedstuff (Stanton, 2004). Results for 1 t of the feed we used are shown in Table 45.

Table 45 – Digestible energy content of each ton of feed.

Ingredient	Quantity ($t.t^{-1}$ feed)	DM (%)	DE ($MJ.kg^{-1}$ DM)	DE (MJ)
Maize (silage)	0.586	36	12.7	2 687
Maize (grain)	0.125	89	16.4	1 828
Corn Gluten Feed	0.066	90	15.1	899
Wheat (grain)	0.066	89	16.4	965
Soy meal (44% protein)	0.091	90	16.1	1 315
Premix	0.008	-	-	-
Wheat (straw)	0.058	88	8.1	415
Total	1	-	-	8 175

DM – Dry Matter; DE – Digestible Energy.

As for SBPPRL, since there is no single composition of the pasture, we must arbitrate significant values for *EFU*. We use clover as representative of SBPPRL energy content. The digestible energy in clover varieties may be retrieved from energy equaivalency tables for feedstuff. Table 46 shows the average, minimum and maximum values (Stanton, 2004). The interval is due to the fact that the energy content depends on the state of maturation. Green (early stage) clovers are typically more energetic (corresponding to the maximum values). To obtain the total energy in one hectare, we must multiply the values in Table 46 by the dry matter production of SBPPRL which, according to Table 21, is 7 000 kg.

Table 46 – Digestible energy content of each clover species.

Clover variety	Digestible energy ($MJ.kg^{-1}$ dry matter)		
	Average	Minimum	Maximum
Trifolium alexandrinum	19.5	15.3	22.7
Trifolium meneghianum	20.0	20.0	20.0
Trifolium pratense	21.8	21.8	21.8
Trifolium repens	20.7	20.7	20.7
Trifolium squarrosum	21.1	20.4	21.8
Trifolium subterraneum	25.6	25.6	25.6
All varieties	**20.0**	**15.3**	**25.6**

Results for the difference in impacts are shown in Table 47 for all impact categories studied. A negative value means that, for every hectare of SBPPRL sown, we obtain less impacts than in the prior situation with NG. This is the case for all themes when 100 kg of phosphate fertilizer are used. When 200 kg of phosphate fertilizer are used, the only theme where the scenario with SBPPRL has more impact is winter smog. This is one of the themes where phosphate fertilizers have more impacts than nitrogen fertilizers, and therefore there is a significant loss when substituting feeds for SBPPRL. Results are qualitatively the same if we use the extreme values of digestible energy in Table 46 instead of the average; the only difference is that, when using the maximum value, the scenario with SBPPRL has less impacts in all themes regardless of the quantity of phosphate fertilizer.

Table 47 – Difference in impacts between scenarios per area of SBPPRL sown.

Impact category	[Unit]	ε	$\{EFU\}^{SBPPRL}$ MJ.ha^{-1}	$\langle EFU\rangle^{feed}$ MJ.ton^{-1}	$\frac{\{I\}_f^{total}-\{I\}_i^{total}}{A_f^{SBPPRL}}$ 100 kg.yr^{-1} phosphote	200 kg.yr^{-1} phosphote
GHG	kg CO2	2	139.8 x 10^3	8.2 x 10^3	-6.9 x 10^3	-6.7 x 10^3
Ozone layer	kg CFC11				-2.6 x 10^{-4}	-2.3 x 10^{-4}
Acidification	kg SO2				-2.9 x 10^1	-2.5 x 10^1
Eutrophication	kg PO4				-7.5 x 10^0	-6.9 x 10^0
Heavy metals	kg Pb				-1.4 x 10^{-2}	-1.4 x 10^{-4}
Carcinogens	kg B(a)P				-1.8 x 10^{-4}	-1.5 x 10^{-4}
Winter smog	kg SPM				-1.6 x 10^0	1.8 x 10^0
Summer smog	kg C2H4				-6.7 x 10^{-1}	-5.9 x 10^{-1}
Energy resources	MJ LHV				-2.1 x 10^4	-1.8 x 10^4

I – Impact; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; NG – Natural Grasslands; GHG – Greenhouse Gases; LHV – Lower Heating Value; LCA – Life Cycle Assessment.

3.5.4.3. Consistency analysis – what fraction of a farm is sown at most?

Another interesting analysis is to determine what fraction of farm area is sown by farmers or, in other words, how much area of SBPPRL do farmers have to install in order to eliminate concentrated feeds. In order to do so, we return to Equation (3.34) and to the fact that two variables were unknown: $\{M\}^{feed}$ and *x*. From Equation (3.40), we can see that there is a direct proportionality between both, and that neither one can be determined independently. In other words, there are several combinations of the two variables that yield the same environmental impacts that we determined in Table 47. The relation between the variables may be obtained from Equation (3.34) as:

$$\frac{\{M\}^{feed}}{x}=\frac{\{I\}^{SBPPRL}-\{I\}^{NG}-\dfrac{\{I\}_f^{total}-\{I\}_i^{total}}{A_f^{SBPPRL}}}{\langle I\rangle^{feed}}.$$

From Equation (3.40), this is the equal to

$$\frac{\{M\}^{feed}}{x}=\left(\frac{\varepsilon-1}{\varepsilon}\right)\frac{\{EFU\}^{SBPPRL}}{\langle EFU\rangle^{feed}}.$$

Using the same values already shown in Table 47, we find that the ratio between the two variables is equal to 8.56 kg.ha^{-1} SBPPRL. We can thus build a table that presents, for each x, which would be the feed initialy provided to animals, in the absence of SBPPRL. Then, we can multiply that quantity by the $\langle EFU\rangle^{feed}$ to calculate the energy provided by the feed in each case. We can add that value to the $\{EFU\}^{NG}$ (which we know is half of the value for SBPPRL, since $\varepsilon = 2$), and obtain the total energy provided to animals in the initial situation. Then, to check the consistency of the results, we can calculate the energy provided in the final situation, which is equal to the fraction of SBPPRL times $\{EFU\}^{SBPPRL}$ plus the complementary fraction, which are NG, times $\{EFU\}^{NG}$. The results for each line of the table should be the same. The consistency of results is shown in Table 48.

Table 48 – Initial and final digestible energy for each scenario of SBPPRL area fraction.

x	$\{M\}^{feed}$ (t.ha^{-1})	$\{EFU\}^{feed}$ (GJ.ha^{-1})	$\{EFU\}^{NG}$ (GJ.ha^{-1})	$\{EFU\}_i^{total}$ (GJ.ha^{-1})	$\{EFU\}_f^{total}$ (GJ.ha^{-1})
0.0	0.00	0	89.5	89.5	89.5
0.1	1.09	8.9	89.5	98.4	98.4
0.2	2.19	17.9	89.5	107.4	107.4
0.3	3.28	26.8	89.5	116.3	116.3
0.4	4.38	35.8	89.5	125.3	125.3
0.5	5.47	44.7	89.5	134.2	134.2
0.6	6.57	53.7	89.5	143.2	143.2
0.7	7.66	62.6	89.5	152.1	152.1
0.8	8.76	71.6	89.5	161.1	161.1
0.9	9.85	80.5	89.5	170.0	170.0
1.0	10.95	89.5	89.5	179.0	179.0

x – fraction of SBPPRL area in farm; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; NG – Natural Grasslands; EFU – Energy Forage Units; i – initial situation; f – final situation.

The higher the value of *x*, the higher the quantity of feed that is given to animals in each hectare of NG in the initial situation. Since we considered a constant energy provision by each type of pasture, this also means that more energy is required by livestock. Therefore, if we estimate how much feed is needed in an average case, we can also know *x* in an average case.

If we assume a stocking rate of 0.5 LU.ha^{-1}; that a cow weights around 500 kg; that a cow eats between 0.5 and 1.5% of its live weight per day; then we can assume that feed consumption in the initial situation is 0.5-1.4 t feed.ha^{-1}. According to Table 48, this implies *x* between 0 and 20 %. In our case, the critical value is never reached, and we have reason to believe that NG and SBPPRL will always co-exist in farms. They are complementary, rather than purely competitive, options.

If we now return to the first Chapter, and recall that in Table 1 we showed that 15.6% of the area of pastures in Project Extensity were SBPPRL[46], then we obtain the justification we needed for our prior approach when we clamed that farmers install SBPPRL up until the point when no more commercial feeds are required.

Two environmental effects which are not accounted by LCA are soil loss and biodiversity. We now turn to those.

[46] We did not use this figure in the beginning because, even though it may be used as an indication for consistency check, it is unreliable as a parameter. The overall Extensity area is not equaivalent to a single farmer, and even so there was no indication that all farmers had no intention of sowing more pastures.

3.6 Soil protection and decreased erosion

NG require some form of mechanical operation to control shrub invasion. SBPPRL, on the other hand, do not, since animals control shrubs by themselves. The reduction of these actions, which imply some form of soil tillage, corresponds to less destruction of soil agglomerates. This effect is well documented for croplands where tillage was substituted by no-tillage (Basch *et al.*, 2001). The passage of agricultural machinery creates a crust at the surface of the field which decreases the infiltration of water, increasing water runoff and carriage of soil particles. Another main determinant of soil erosion is SOM. Soils with high SOM concentration are more stable, and less susceptible to erosion.

Soil loss may be estimated using Wischmeier and Smith's (1978) Universal Soil Loss Equation (USLE). This is the most simple and well-known model to this end. The basic formulation of the model is

$$A = R \cdot K \cdot LS \cdot C \cdot P \,, \tag{3.19}$$

where

A – average soil loss or specific erosion ($t.ha^{-1}.yr^{-1}$);

R – rainfall erosivity factor ($MJ.mm.ha^{-1}.h^{-1}.yr^{-1}$);

For Portugal, and according to Tomás (1993), R may be estimated according to an empirical equation obtained as the average for three stations in the Algarve region:

$$R = -685{,}3 + 3{,}406P \,, \tag{3.20}$$

where P is total annual precipitation (mm).

K – soil erodibility factor ($t.MJ^{-1}.mm^{-1}$);

LS – slope length-gradient factor, considering slope length and steepness simultaneously (dimensionless)

$$LS = \left(\frac{x}{22.13}\right)^m .\left(0.065 + 0.045s + 0.0065s^2\right) \tag{3.21}$$

x – average slope length (m)

m – factor determined by the interaction between slope length and steepness, as well as soil proprieties, vegetation type and agricultural practices (dimensionless, 0.2 < m < 0.6)

s – slope steepness (%)

C – crop/vegetation and management factor (dimensionless)

P – support practice factor (dimensionless)

From all these factors, the ones where grassland type has any significant influence are K, C and P. We will analyse next the effect of switching land use from NG to SBPPRL in each of the three coefficients.

3.6.1 Effect on the parameter K

According to Pimenta (1998b), the K factor for permanent pastures is 0.02, which is the same as for areas with spontaneous herbaceous or shrub vegetation. There is no value for SBPPRL. Therefore, we determined the difference between both from the difference in SOM concentration. Chapter 1 of the present thesis is dedicated to comparing SOM dynamics in different grassland types. For now, it suffices to say, for motives we will explain then, that SOM increases in SBPPRL by about 0.20 percent points per year more than NG. We will consider that, in 10 years, a NG is kept for 2 cycles of 5 years with no mobilization, keeping its SOM level in 1% throughout the period. In the same 10 years, SBPPRL are never tilled, and increase their SOM level from 1 to 3%.

To determine K, we resort to Pimenta (1998a), and the expression

$$K = \frac{2.1 \times M^{1,14} \times 10^{-4} \times (12 - SOM) + 3.25 \times (\alpha - 2) + 2.5 \times (\beta - 3)}{759.3}, \tag{3.22}$$

where

M – % of loam and thin sand multiplied by 100% less the percentage of clay (four major texture classes were considered, as shown in Table 49);

Table 49 – Percentage of each soil constituents for major texture classes.

Textura	% clay	% lome	% sand
Clay	60	10	30
Silt	20	25	55
Loam	5	65	30
Sand	5	2,5	92,5

α - factor for soil structure class (1 to 4, as shown in Table 50);

Table 50 – Parameter α to determinate soil erodibility.

Structure	α
Very thin granulate	1
Thin granulate	2
Coarse granulate	3
Compacte	4

β - factor for soil permeability class (1 to 6, where 1 is very slow and 6 is fast).

Table 51 – Parameter β to determinate soil erodibility.

Permeabilidade	β
Very slow	1
Slow	2
Slow to moderate	3
Moderate	4
Moderate to fast	5
Fast	6

Results are shown in Table 52[47]. We estimated soil erodibility for the four major texture classes.

Table 52 – Soil erodibility for the four situations analysed.

Texture	α	β	K (%SOM = 1)	K (%SOM = 3)
Clay	Very thin granulate	Very slow	0.001	-0.001
Silt	Thin granulate	Moderate	0.021	0.018
Loam	Compacte	Slow to moderate	0.072	0.061
Sand	Coarse granulate	Fast	0.016	0.015

K – Soil erodibility; SOM – Soil Organic Matter.

3.6.2 Effect on the parameter C

According to Pimenta (1998b), factor C for permanent pastures is 0.02, while for zones with spontaneous herbaceous or shrub vegetation is 0.05. While the first factor is our best guess for SBPPRL, the second one must be corrected to better depict the NG system. Also according to Pimenta (1998b), the factor C rises to 0.40 for croplands due to tillage. NG require some tillage operation for shrub control, which we considered to happen every five years (twice every ten years). Therefore, the average factor C is 0.12 (20% x 0.40 + 80% x 0.05). Therefore, the parameter C is 83% lower in SBPPRL than NG.

3.6.3 Effect on the parameter P

Parameter P refers to specific practices aimed at lowering soil erosion. Even though it could be argued that this pasture system is, in itself, a management practice to diminish erosion, there are few values available for this parameter. To our knowledge, only parameters for cross slope cultivation, contour farming and strip-cropping have been estimated. Since these are not comparable with the SBPPRL system, we have to use the default value for P, which is 1, in both scenarios (SBPPRL and NG).

[47] Note that parameters α and β were subjective choices, determined from the correspondence with each texture class.

3.6.4 Combined effect

To understand how much soil loss is avoided by the use of SBPPRL, we must calculate all parameters in the USLE. Therefore, we made some assumptions to calculate the value of A in a general case.

We assumed an average total yearly precipitation of 500 mm. Using Equation (3.20), we obtain an R of 1 017.7 $MJ.mm.ha^{-1}.h^{-1}.yr^{-1}$. We also considered a slope steepness of 5% and a slope length of 50 m, thus obtaining an m of 0.5 (Tomás and Coutinho, 1993), using Equation (3.21). Results for average soil loss, obtained with Equation (3.19), are shown in Table 53.

The absolute results are very dependent on soil erodibility (parameter K), since changes in K will have proportional differences in soil loss (parameter A). Therefore, K controls how much soil is lost. The difference between SBPPRL and NG is also sinfluenced by K, but it is mostly the management factor (parameter C) that is responsible for less soil to be lost in SBPPRL. Therefore, according to the USLE, SOM increases decrease soil loss, but the most important effect is that of no-tillage.

Table 53 – Average soil loss in NG and SBPPRL.

Texture	Average soil loss ($kg.ha^{-1}.yr^{-1}$)		
	NG	SBPPRL	Difference
Clay	8	0	8
Silt	174	25	149
Loam	598	84	514
Sand	133	21	112

NG – Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.

Two additional facts must be stressed regarding soil loss. When topsoil is lost, organic matter and nutrients are also carried away with the mineral materials. Therefore, erosion also causes SOM loss and phosphorus loss. Phosphorus has the opposite electrical charge that mineral soil particles do. Therefore, is becomes adsorbed to the soil, and so phosphorus losses occur through surface soil erosion. Even though adsorbed phosphorus is largely unusable to plants, decreasing soil loss is also a way to decrease nutrient loss.

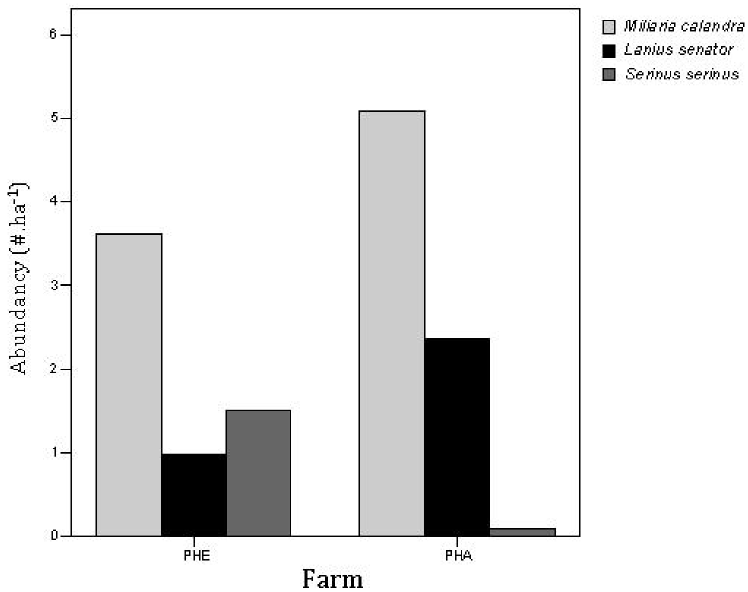
3.7 Effects on biodiversity

The effects of grassland system in wild animal biodiversity were studied in the course of Project Extensity. All results briefly explained next were obtained in studies conducted by Liga para a Protecção da Natureza (LPN).

Two types of bio-indicators were used, namely insects and birds. For coleopterous insects, pitfall traps were placed in adjacent plots of SBPPRL and NG. Birds common in agricultural zones were indentified through expert listening, also in adjacent plots of both types of grassland. This experimental setting was repeated at four regions, namely Lower Alentejo (Castro Verde), Upper Alentejo (Portalegre), Central Alentejo (Montemor-o-Novo) and Cova da Beira (Covilhã). These four locations represent four ecossystem types: pseudo-steppe (Castro Verde), *montado* (Montemor-o-Novo and Portalegre), and mountain (Covilhã).

It could be argued that, since SBPPRL are more productive, and livestock stocking rates may be higher, the quantity and diversity of birds would decrease. However, results obtained did not show significant differences in what respects to birds, as shown in Figure 32. Bird presence in grasslands seems to be more correlated with factors that do not respect to grassland type, such as the presence of trees, places to nidify, fences or other locations to rest. This is not surprising, because birds have a large area of activity, much larger than average pasture plots.

Figure 32 – Bird biodiversity in natural and sown pastures (using three species as indicators).

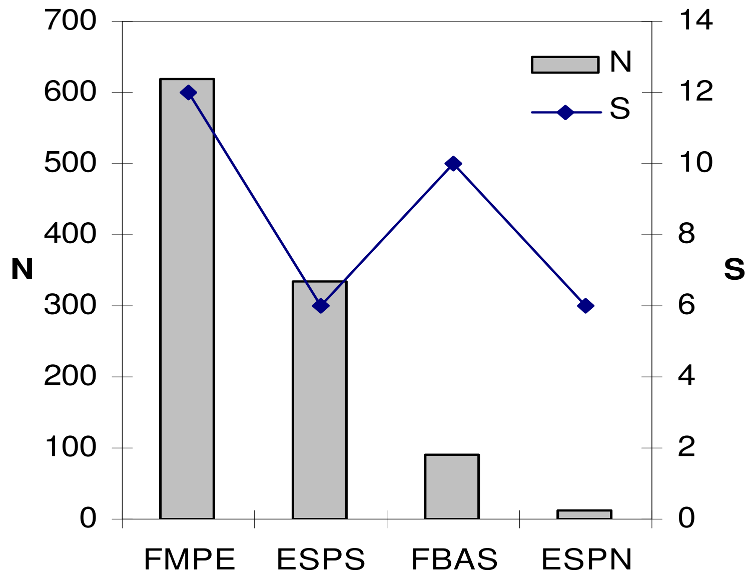


PHE – SBPPRL, Organic, (Northern Alentejo); PHA – Natural pastures, conventional (Northern Alentejo, adjacent farms)
Source: Henriques *et al.* (2008).

Regarding insects, Figure 33 shows that SBPPRL have a statistically significant higher quantity (N) than NG. The sown pasture in the sample ESPS is comparable with the natural pasture ESPN, since they were in the same region. While the specific richness (S) is similar in both farms, the number of insects is much lower in ESPN. Part of this result is also due to the fact that organic farming is practiced in ESPS. The NG in the organic farm FMPE in Northern Alentejo had more quantity and richness of insect species than the conventional farm FBAS.

Again, this confirms our initial intuition – that more productivity, together with more organic matter in soils, yields livelier micro-habitats for wild soil fauna. Soil fauna, on the other hand, is also crucial for SOM increases and stability. As Bot and Benitez (2005) notice, soil fauna is crucial for creating channels for air and water, and also for promoting the binding of mineral and organic particles. While channels are particularly important for no-tilled soils, which are more compact, binding particles decreases erosion and increases water holding capacity.

Figure 33 – Insect biodiversity in natural and sown pastures.



N – Abundancy; S – Specific richness.
FMPE – Natural pastures, organic farming (Central Alentejo); ESPS - SBPPRL, organic farming (Northern Alentejo); FBAS – Natural pastures, conventional farming (Central Alentejo); ESPN – Natural pastures, conventional farming (Northern Alentejo).
Source: Henriques *et al.* (2008).

Even though these results do not point to a clear answer of which grassland system is better, they do show the importance of landscape planning as an integrated system where all elements matter. Most grassland systems can have positive effects on biodiversity, if they are complemented by other elements such as tree cover. We referred previously that it is very likely that the SBPPRL and NG systems always coexist in the same farm. Biodiversity benefits the most from this case, since there are more niches available, and also because there are areas richer in insects (bird feed), and others with lower stocking rates (bird getaway). In this respect, SBPPRL and NG are complementary and not competitors.

3.8 Synthesis of results and discussion

In this chapter, we quantified some environmental effects of pasture systems. First, we converted previously obtained SOM increase for SBPPRL into its equivalent carbon sequestration. We determined that on average rainfed SBPPRL sequester 5 t $CO_2.ha^{-1}.yr^{-1}$. This allowed us to estimate that existing SBPPRL areas in Portugal have sequestered more than 3.5 Mt CO_2 from 1996 to 2008. Afterwards, we determined the overall balance, by subtracting from the sequestration the emissions from animals, legumes and liming. The balance is still positive – 4.1 t $CO_2.ha^{-1}.yr^{-1}$ sequestered.

This potential of SBPPRL to sequester carbon must be compared with that of other land uses and management activities. Soil carbon sequestration in no-till soils with crop residues was estimated as 7.7-8.5 t $CO_2.ha^{-1}.yr^{-1}$, and without crop residue cover is 1.9-

2.1 t CO_2.ha^{-1}.yr^{-1}. However, due to lack of data, it is unknown for how long these increases occur (when the maximum saturation is reached). Besides, we only considered the top 10 cm layer for SBPPRL, and in no-till crops 30 cm is the important depth for SOM determination (since roots go deeper). Regarding other land uses, we could not find significant evidence that there is any SOC dynamics.

We also showed that there is also an environmental advantage in sowing irrigated pastures, when we compare them to their alternative land use which is maize production. This effect comes up only when studying the whole production life cycle. This led us to calculate indirect environmental impacts also for rainfed SBPPRL, using LCA. We concluded that, for all impact themes, the impact of a scenario where farmers sown part of their farm with SBPPRL to replace commercial feeds is lower. The only significant impact of SBPPRL is the production of phosphate fertilizer.

Two environmental impacts, though, were left off the LCA method use: soil and biodiversity loss. So, we used the USLE to conclude that soils with SBPPRL are less eroded than soils with natural grasslands. We also reviewed results that show that some insect species are benefited by SBPPRL and, at least, there is no loss in some bird species in SBPPRL plots.

Therefore, in this Chapter we conclude that the whole system of SBPPRL (rainfed) or SIP (irrigated) is responsible for the sequestration of more carbon than their respective alternatives (NG or maize, respectively), and do so with positive environmental effects in all studied themes. We could not find any environmental impact theme where SBPPRL cause significantly more damage than NG. On the contrary, for most of them (carbon sequestration, soil loss) they provide environmental services.

4. Paying for the environmental services of SBPPRL

In Chapter 4 we show the several ways through which farmers can obtain revenue from pastures due to the fact that they provide environmental services. Namely, they can use direct agricultural support, they can sell products which consumers value due to its production method, and they can be paid for carbon they sequester.

We discuss consumer valuation of meat products from SBPPRL, before designing a scheme for payments for carbon sequestration in SBPPRL by the PCF. We conclude by determining the impact of project in terms of expected carbon sequestration.

4.1 Support of SBPPRL by rural development policies

The current Portuguese Rural Development Programme (MADRP, 2007) for the period 2007-2013 stipulate specific amounts to support the installation and maintenance of SBPPRL, but only if they are managed according to two specific agricultural norms: integrated production and organic farming. Base values for the maintenance of natural grasslands and SBPPRL are shown in Table 54.

Table 54 – Public support for the maintenance of natural grasslands and SBPPRL.

Land use	Integrated Production (€.ha^{-1})	Organic farming (€.ha^{-1})	Base area (ha)
Permanent (natural) grassland	106	172	30
SBPPRL	130	210	30

SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.
Source: MADRP (2007).

These values should be read as follows: for a farmer with a grassland area inferior to the base area, the indicated values will apply. If a farmer has a superior area that that indicated, the values per hectare drop. The modulation factors are shown in Table 55.

Table 55 – % of total sum attributed to each area class.

Area class, in relation to Base Area (BA)	% of total sum
≤ BA	100
> BA and ≤ 2BA	80
> 2BA and ≤ 5BA	50
> 5BA	20

The major difference in the level of support to both types of grasslands lies in the income loss for the adoption of these systems, in relation to a conventional production method. Difference lies in the need for correct phosphate (P) fertilization, especially in SBPPRL. The value for the first area class under integrated production is consistent with the one we used for running costs (130 €.ha^{-1}.yr^{-1}), which means that public financing was designed to cover running costs.

Public financing was designed in collaboration with PNAC's objectives. This means that grasslands supported by the Rural Development Programme will be fulfilling PNAC's goals. All additional carbon sequestered in grasslands has to be contracted outside from this kind of public financing.

4.2 Consumer valuation

Farmers sometimes argue that payments for environmental services, just like all forms of public support, are not enough to guarantee security in their activity (Nuno Rodrigues, personal communication). Farmers seem to prefer what they see as the main result of their activity, which is the direct revenue of whatever products they produce. Therefore, it is important to determine if we can expect an increase in revenue due to the fact that meat is produced in SBPPRL. Consumers may perceive those meat products as more valuable than conventionally produced meat, as they do with organic meat, for example, for two reasons: higher product quality, and due to being sustainable. While the first reason is straightforward[48], the second reason has been increasingly study, as it is getting more valued by some types of consumers by the hour (Harris, 2007).

A first attempt to study this subject was done during Project Extensity, by directly enquiring consumers on their preferences and willingness to pay for meat products (Jorge *et al.*, 2006; Silva *et al.*, 2006, 2007, 2008a, 2008b). Results of these studies are summed up next.

It is important to explain that during Project Extensity a norm was developed for sustainable beef production (Domingos *et al.*, 2005). This norm, named "Guaranteed Sustainability" (GSN), was based on the Forest Stewardship Council (FSC) norm for forest products. In that sense, it was guided by two main principles: the congregation of economic, environmental and social aspects, and the involvement of major stakeholders. It was subscribed by governmental entities, an association of farmers, a representative of consumers, and an environmentalist group. This norm is based on the integrated production method, but it goes beyond it by introducing some obligations in what regards to soil and biodiversity protection. It also highly promoted the use of no-tillage for annual forages and crops, and SBPPRL as the basis for animal feed. This Norm also sets some procedures for handling manure: facilities should take into account the daily production of effluents and that storage is necessary whilst manure application in soil is not advisable (3 to 4 months), assuring protection from rainwater. Also, manure piles must be kept at given distances from water streams, drains, fountains or wells. By following these measures, GHG emissions by leaching and runoffs are avoided, and only gaseous emissions are relevant.

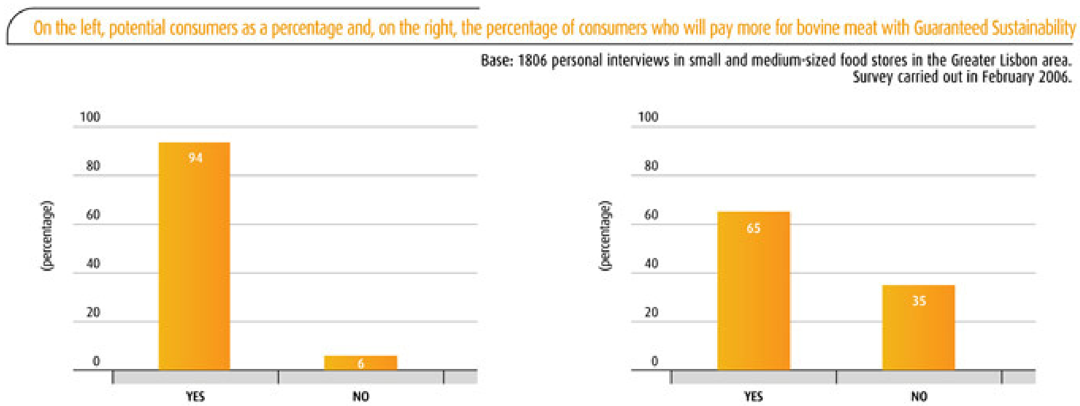
It was the specifications and obligations of the GSN that were evaluated by consumers in the surveys done during Project Extensity. Surveys were conducted by direct interviewing, either in person or by telephone. Surveys done in person took place in stores where meat was sold, in the Lisbon and Porto regions. Surveys done via telephone were done all over the country, and each had over 1 000 validated interviewees.

Results have shown that conventional bovine meat, with no specific label, is the most consumed type of meat. Meat produced by organic farming, together with meat from other certification systems, is no more than 30% of the total. The main reasons pointed by consumers to justify this consumption pattern are the high prices and low availability of high quality meat. However, consumers valued the principals that guide

[48] Note that, even though consumers may recognize that higher intrinsic quality is a reason to increase the price of the product, but it still does not mean that they are willing to pay more for it.

the GSN (on average, graded as a 4 in a scale of 1 to 5). 94% of surveyed consumers claimed to be potential consumers of meat produced following the GSN (Figure 34).

Figure 34 – On the left, potential consumers as a percentage and, on the right, the percentage of consumers who will pay more for bovine meat with GSN.



Survey carried out in the Lisbon region in February 2006. 1 806 interviews were carried out.

Another survey carried out 1 718 interviews in person, also in small and medium stores. When presented with the average conventional meat price, 79.3% of inquired consumers claimed to be willing to pay more than 10 €.kg^{-1} for GSN bovine meat. In this sample and on average, interviewed consumers claimed to be willing to pay about more 3.5 €.kg^{-1} for GSN meat than for conventional meat. However, due to budget restrictions of households, this would mean a decrease in meat consumption. A demand curve for GSN meat was built, showing that the monthly consumption of meat is reduced by 0.8 kg for each Euro paid in excess of the conventional meat price.

A telephone survey, in which 755 interviews were conducted in the beginning of 2007, showed that consumers highly value the fact that animals are allowed to graze in pastures instead of being stabled. In fact, 83% of inquired consumers claimed to be willing to pay more (about 1.5 €.kg^{-1} on average) for meat of an animal that isn't stabled after weaning. However, the main reason pointed is meat quality and not animal well-being. Afterwards, it was explained that animal feed based in green forages or pastures produces meat which is darker, while concentrated feeds are responsible for animals with clearer meat. When asked which one inquired consumers prefer, 89% claimed to prefer the first one.

In the context of this thesis, these results cannot be applied directly. Values were obtained for a whole bundle of characteristics which together make the GSN. They were never specifically asked whether they value SBPPRL over NG. However, these results show that, when explained, consumers prefer the characteristics that are accentuated by feeding animals in SBPPRL, namely the time spent grazing, and also the darker tone of the meat.

We have stated before that the darker tone of the meat comes from more grazing months for steers, and that has environmental advantages since it withdraws livestock from intensive production (Section 3.2.2.9). But we have also mentioned before that darker tones of meat seem to discourage retailers from buying animals from farmers

(Section 1.6). However, when asked wether they would prefer a dark meat over a red meat, if the dark meat meant that steers were kept in pastures, consumers answered yes.

These results hint that, in principle, consumers are willing to chose meat from SBPPRL over other types of meat, if it is presented as a package (like the GSN), and if it means that the meat has a higher quality and taste. The environmental advantages of products from SBPPRL have been shown throughout this thesis, and the quality and health standards of those products has also been studied during the course of Project Extensity. Results of food chain monitoring done in the context of Project Extensity (Ralha *et al.*, 2008) showed that meat from livestock fed in SBPPRL is of higher quality, and richer in saturated fatty acids.

4.3 Designing payments for carbon sequestration

So, farmers who install SBPPRL can resort to agricultural support and to increased market revenue due to higher quality (taste and health, for example) of their products. But is it also possible that farmers obtain direct payments for the environmental services they provide? Carbon sequestration seems to be the service to inspect, since carbon is the only environmental commodity with an established market. Next, we briefly describe how such contracts for environmental payments are established, why the Portuguese Carbon Fund (PCF) is interested in paying for carbon from SBPPRL, and finally how a Project was conceived and approved by the PCF, and what its expected results are[49].

4.3.1 Designing contracts for resource conservation

In standard economic theory, environmental co-effects of human activities are often considered externalities, in the sense that they are not reflected in commodity prices (Perman *et al.*, 1996). As this effect aggravates, we say that there is a market failure, since the current state of business is incapable of internalizing that impact. There are two main ways usually appointed to cope with this market failure: environmental taxes, and environmental contracts.

An environmental tax is imposed on polluters as a way of internalizing the environmental damage (Perman *et al.*, 1996). The tax will increase production costs, incentivating producers to switch to cleaner techniques, and the collected funds will be used to mitigate the damage effects. But it is a normative regime, and the state is required to regulate and control its operationalizitation. Alternatively, polluters may be given the option to compensate their impacts by participating in other markets created specifically for environmental assets. These markets involve a third party, the suppliers of environmental assets, with who are established contracts for the implementation of measures with positive effects.

The last case constitutes the universe of private resource conservation contracts, either voluntary (for example, biodiversity protection) or mandatory (for example, carbon emissions reduction). But even in this kind of contracts, the state plays a crucial role, since it is usually the source of most funds to such markets. Resource conservation is

[49] The next sections are partially based on a Msc. Thesis in Economics (Teixeira, 2008).

very dependant of state intervention, since in fact most environmental goods are public and non-exclusive.

It is not a surprise that most resource conservation revolves around agricultural practices and forestry activities. The agro-forestry sector has a very close link to direct environmental effects, and regulates many of the possible ecosystem services, like water cycle regulation, water quality, soil protection, carbon sequestration, air quality, and biodiversity. However, not all transactions involving environmental services from agriculture and forestry are efficient (Gulati and Vercammen, 2006).

The issue of resource conservation contracts has been fairly debated in the economic literature of the last decades. According to Antle *et al.* (2003), there are two types of costs regarding the implementation of contracts for the adoption of good environmental practices. The first type is farm opportunity costs, which includes the cost of the conversion to the required agricultural practice. The second type is contract costs, which include transaction, monitorization costs, and brokerage fees.

There are mainly two mechanisms for the payments to occur: on a "per hectare" base, and on a "per objective" base (Antle *et al.,* 2003). The per hectare scheme consists on a fixed payment for an area where a farmer adopts a land use or management practice leading to environmental benefits. This type of contract is mainly used when there is no specific monitorization method, or it is not possible to correctly assess the result of the adopted practice. The per objective scheme involves a payment based on the accomplishment of environmental objectives, measured in quantitative indicators. For example, for carbon sequestration, this would mean a "per ton" of CO_2 contract. Every time a farmer would switch its land use or farming practice, he would receive a payment for each unit of carbon incorporated in the soil. Models and field measurements are used to assess the respective sequestration (Antle and McCarl, 2002).

The difference between the two types of contract is that in the first one the farmer receives a payment regardless of the accomplishment of any environmental goal (for example, carbon sequestration). Therefore, the entity that finances the project assumes the risk, and there is only the monitorization cost of assuring that the farmer does indeed adopt the practice contracted. In the second type of contract, only the effective environmental gain is paid for, and it is the farmer who assumes the risk, which also means he has a real incentive to correctly apply management practices. There is, however, a higher cost of monitorization, due to the need of environmental studies (for example, soil analysis for SOM determination). It is thus unclear which of the two types of contract is more efficient, in the sense that more carbon is sequestered by the same total amount paid (Antle and McCarl, 2002).

4.3.2 The case of carbon sequestration

The United States of America did not sign the Kyoto Protocol (KP), but established in 2001 a voluntary initiative that, alongside other measures, included the possibility of carbon sequestration by forests and in grasslands and croplands (Mooney *et al.*, 2002). This initiative is private, consisting on contracts between industries that emit greenhouse gases (GHG) and farmers. But its contribution is not neglectable. Lal *et al.* (1998) indicate that carbon sequestration in agricultural soils could, alone, decrease U.S. emissions by 8%. The credits purchased in this way have prices competitive with those obtained by forest sequestration (Antle *et al.*, 2002).

So, in these resource conservation contracts relating to carbon sequestration, two questions are usually appointed: which economic incentives induce farmers to appoint management techniques that increase soil CO_2?, and would that form of sequestration be economically competitive with other forms of emission reduction (Antle and McCarl, 2002; Marland *et al.*, 2001a)?

Lewandrowski *et al.* (2004) studied which practices would be favoured by farmers for different ranges of incentives in the United States of America. They find that higher payments for each ton of carbon make afforestation the most adopted activity, while for lower payments changes in rotation and tillage practices occur. McCarl and Sands (2007) conclude the same: lower CO_2 prices make cropland and grassland management extremely attractive carbon sinks. They confirm that agricultural sequestration may be an economically effective carbon sink.

However, there are some critics of the use of land use, land use change and forestry (LULUCF, now named AFOLU, which stands for "agriculture, forestry and other land uses") practices for carbon sequestration. Their argument usually revolves around the issue of non-permanence.

4.3.3 The permanence issue

It is necessary to take into account that some ways to sequester carbon do not necessarily correspond to a permanent decrease. This is the case for carbon storage in masses of water (Herzog *et al.*, 2003), and also for soil carbon sequestration (Blanco and Forner, 2000; Ellis, 2001). The problem is that if there is a reversion of practices, such as the use of tillage for cropland management or deforestation for forest management, sequestered carbon will be emitted again (Antle and McCarl, 2002). Furthermore, soils may be net emitters in certain climatic adverse years. This is usually referred to as "leakage", and Murray *et al.* (2004) estimate that, for forestry activities, it may be as high as 90%.

One way to address this issue was proposed by Blanco and Forner (2000) and Chomitz (2000), who all use the concept of temporary or expiring credits. Temporary credits are basically a way to buy time while cleaner technologies are set in place, keeping all the environmental benefits of permanent sequestration. Many have followed to use this concept. Marland *et al.* (2001b) consider temporary sequestration to be completely different from permanent avoided emissions. They propose a market for temporary sequestration credits (which is equivalent to a rental market), separate from the carbon emissions credits market. More recently, Maréchal and Hecq (2006) picked up this idea, proposing the issuing of temporary credits from AFOLU activities.

But expiring credits create another market for carbon sequestration, parallel to the market of emissions reduction. Other authors have found other ways to cope with the issue of permanence. The other option is to use an equivalence factor (Kim *et al.*, 2007), which considers that a ton of carbon sequestered during a certain number of years is equivalent to a permanent reduction (Moura-Costa and Wilson, 2000). This is known as the ton-year accounting method, and has spawned several studies which yielded equivalence times from 42 to 150 years (Maréchal and Hecq, 2006). This means that there is a high uncertainty, and that for longer time spans sequestration projects become less interesting.

There are other alternatives, like the average storing capacity method (the main difference is that the variation in carbon stocks is used to generate credits) or liability mechanisms (in which each country would have to compensate AFOLU emissions by a reduction elsewhere), minimum duration or buffer credits (Maréchal and Hecq, 2006), but none of them are consensual or even desirable. Others determine equivalence factors. Keller *et al.* (2003) recognize that CO_2 sequestration is not a perfect substitute for the avoidance of CO_2 emissions, but still believe that they should be compared. In order to do so, they define an efficiency factor for CO_2 sequestration as the ratio between economically equivalent avoided and sequestered emissions. They find that afforestation is only about 60% efficient, while sequestration in water masses is about 90% efficient.

4.3.4 Problems with carbon contracts

The major problem with this type of contract is the fact that carbon sequestration may not be as easily measured as an emissions reduction in a point-source, like a factory, or above ground forest biomass (Mooney *et al.*, 2002). Direct measurement, either by flux measurement or soil analysis, is very expensive, and so many times the only possibly way to verify if a farmer does comply with the contract is to observe its practices. However, the only way to relate practices with sequestration is by using fixed sequestration factors, which usually underestimate the potential of soils, and do not consider how well the farmer manages his land. Therefore, the issue of efficiency and trade-off between monitorization costs and net CO_2 sequestration is not linear.

There is also a consequence related with contracts that only target one environmental objective, which is the fact that they ignore co-effects. Sometimes these co-effects are negative, but in the case of carbon sequestration in soils they are usually positive. Land uses and management practices that enhance the soil's carbon pool typically also reduce soil erosion, nutrient leaching and runoff, and increase the soil water retention potential (as shown in the previous chapter). The co-effects are not negligible, but they remain as positive externalities in carbon contracts (Feng *et al.*, 2007).

It is also important to notice that farmers who adopt techniques leading to carbon sequestration may face an important increase in productivity, due to the fact that an increased SOM content improves the soil fertility status (Lal *et al.*, 1998). This is usually a direct effect which, in this type of contracts, becomes a positive externality.

There are mainly two types of contracts available to farmers: private and public contracts. Private contracts occur via voluntary schemes, in which private firms finance carbon sequestration in soils (Antle and McCarl, 2002). The Portuguese state attributed maximum emissions levels for all polluting firms, but sequestration projects do not decrease their normative target. Therefore, private firms finance these projects mainly for image purposes, and in the process they help Portugal achieve their KP target for free. We present one such case in the next section, which is the contract established between EDP and Terraprima. As for public contracts, they refer to economic payments from the government, specifically with the objective of promoting carbon sequestration. One example of such is the Portuguese Carbon Fund (PCF), which we will address further down the chapter.

4.3.5 Laying the ground: The EDP-Terraprima Project

In 2006, a contract was established between EDP, the main electricity company in the country, and Terraprima, a small firm with agro-forestry activities in Quinta da França (QF), Portugal. It was the first private contract for carbon sequestration in all LULUCF practices in Portugal (forest, cropland and grassland management). It was also the first private contract in Portugal to finance no-tillage and SBPPRL as carbon sinks. In this contract, EDP will finance in the period 2006-2012 projects regarding forest management, cropland management and grassland management on a partial "per hectare" basis. Terraprima undertakes frequent monitorization of its carbon stocks, and is paid according to the effective fixation.

The yearly payment has two components: a fixed part (F_t) and a variable part (V_t). The fixed part is defined for 2006 and 2007 as $F_t = 45\,000\,€$, and for 2008 to 2012 as

$$F_t = R_t \times \min\left(c_t, 3000 \text{ tCO}_2\right), \tag{4.1}$$

where R_t is 15 €.t^{-1} CO_2 in 2006, and is actualized in each year by the national consumer price index, and c_t is the amount of fixed carbon in year t.

The variable part is calculated for each year t, according to

$$V_t = y \times \min(x_t, X_t), \tag{4.2}$$

where y is 6 €.t^{-1} CO_2 when x_t and X_t are inferior to 1500 t CO_2, and 7 €.t^{-1} CO_2 otherwise, and x_t is defined as the difference between carbon sequestration in year t and 3000 tCO_2, and X_t is

$$X_{2006} = x_{2006}, \tag{4.3}$$

$$X_t = X_{t-1} - V_{t-1}/y + x_t,\ t > 2006. \tag{4.4}$$

By definition, if x_t or X_t is negative, then V_t is zero, which means that there is no payment. Prices obtained for the variable part are updated proportionally to the Powernext price variations.

This was the first experience in Portugal of a contract for carbon sequestration in grasslands, which are our case study. Prices and amounts involved in the contract were negotiated between the two firms, even though they reflect the carbon market price fluctuations. However, Terraprima was given a choice to sub-contract part of the carbon sequestration, if Quinta da França was not enough to fulfil the total quantity of contracted CO_2, which were 7 000 tons per year. Terraprima has chosen to do so. There are now six other sub-contracted farms.

In the case of public contracts, we wish to determine the optimum price and quantity of sequestered carbon based on market considerations that arise from the existence of multiple options and multiple farmers. This is the subject of the next section.

4.4 Why was the Portuguese Carbon Fund interested?

The Portuguese State wants to guarantee that the KP goals are achieved efficiently, in the sense of Antle and Mooney (2002): the government strives to maximize social benefits from carbon sequestration per unit of resource used.

Therefore, considering the state as a buyer of sequestration credits, Portugal will acquire them as long as the final price of each ton is lower than the reference price.

We define the reference price as the lower price between other available domestic projects and the lower possible price for international projects. We assume that, if the grassland carbon price is equal to the reference price, the state will prefer to finance grassland sequestration in Portugal, if the alternative is international carbon, due to its co-benefits and due to the fact that it is a national project.

But the Portuguese State needs to address the permanence issue. Even though the KP does not differentiate sequestration from emissions reduction, each State is not indifferent between both. If a non-permanent option is chosen, and all sequestered carbon is emitted after the KP period, then in a possible future period of a second agreement the country will be obliged to compensate for the carbon lost in some other way. This may be seen, in a negative way, as a double cost for the country. But it may also be seen, in a positive way, as buying time.

In the case of SBPPRL, even though it is highly implausible that all farmers who install them now would change land use after 2012, there is no guarantee otherwise, and so a risk of re-emission in the following periods after the KP has to be adressed. In order to determine the maximum price that the state would be interested in paying to farmers, we considered the worst case scenario, which is that after 5 years of payments, all farmers decide to switch practices in such a way that all sequestered carbon is lost[50]. This would mean that the PCF would only be financing temporary sequestration to borrow time. In a future period after the KP expires, Portugal will either continue to finance farmers to keep SBPPRL, or buy carbon credits, depending on which goals and trade mechanisms are defined for after 2012.

We obtain that price by using the concept of Net Present Value (NPV) (Perman *et al.*, 1996):

$$NPV = \sum_{t=1}^{T} A \cdot (1+r)^{-t}, \tag{4.5}$$

where A is the annuity value (€), r is the discount rate and T is the time horizon (years). The NPV of five years of sequestration, plus the updated value of buying an equivalent quantity of credits, must be the same for the state as obtaining the same quantity of credits in the present, instead of financing sequestration. In our case, the A term in Equation (4.5) is equal to the unit price of carbon (€.t^{-1} CO_2) times the number of tonnes sequestered (t CO_2).

We must now make one of two assumptions. Either the PCF would pay farmers only at the end of the period, or divide the payment throughout the period. We begin by formulating the first option, that farmers will only be paid at the end of the KP accounting period (6[th] year), as

$$p \cdot (1+r)^{-6} \sum_{t=1}^{5} C(t) + p_6 \cdot (1+r)^{-6} \sum_{t=1}^{5} C(t) = p_0 \cdot \sum_{t=1}^{5} C(t), \tag{4.6}$$

[50] We will mention next that the actual PCF Project is only valid between 2009 and 2010. However, the application stage was launched in early 2008, and therefore the programme was being prepared for some time before the regulation was published. Therefore, we assumed that we should replicate in this study the same initial conditions that guided the decision of the establishment of the PCF, and consider the case of a 5-year payment.

where p is the price that the PCF is interested in paying farmers (€.t^{-1} CO_2), p_6 is the estimated carbon price in the sixth year (2013), p_0 is the reference price in 2008, $C(t)$ is the amount sequestered in year t (t CO_2.ha^{-1}), and r is the discount rate (for the PCF). The choice of discount rate is crucial, as we shall see. Therefore, we used four different values for r, namely 1%, 3%, 5% and 7%. Using Equation (4.6), we obtain p, since

$$p = \frac{p_0 - p_6 \cdot (1+r)^{-6}}{(1+r)^{-6}} . \tag{4.7}$$

Note that, under this assumption of payments to farmers in the sixth year, the price is independent of the quantity sequestered. Regarding the reference prices, the carbon prices in the present and future market has mostly been oscilating around 20 €.t^{-1} (according to the Bluenext website, http://www.bluenext.fr/). We used several combinations of three values for p_0 and p_6, namely 15, 20 and 25 €.

Results obtained are shown in Table 56. If both reference prices p_0 and p_6 are equal to 20 €, p varies from 1.23 to 10.01 €.t^{-1} CO_2. If both reference prices are lower (15 €), p decreases, and if reference prices are higher (25 €), then p increases. A change in both prices of one quarter (5 €) implies a change in p which is also close to a quarter. Therefore, the price that the PCF is willing to pay follows closely the reference prices. If the reference prices change independently, this result is even more expressive. If the estimated carbon price in the sixth year is higher than the price in the beginning of the project, than for some interest rates the PCF should not be interested in buying any carbon. But if the estimation is that the price is lower, than even for low discount rates the PCF would pay a high price for carbon sequestered.

Table 56 – Maximum carbon price (paid in the sixth year), depending on the discount rate.

	Scenario	***r***			
		1%	**3%**	**5%**	**7%**
***p* (€.t^{-1})**	p_0 = 20 € p_6 = 20 €	1.23	3.88	6.80	10.01
	p_0 = 15 € p_6 = 15 €	0.92	2.91	5.10	7.51
	p_0 = 25 € p_6 = 25 €	1.54	4.85	8.50	12.52
	p_0 = 15 € p_6 = 20 €	-4.08	-2.09	0.10	2.51
	p_0 = 20 € p_6 = 15 €	6.23	8.88	11.80	15.01

r – discount rate (PCF); p – price paid by the Portuguese Carbon Fund for carbon sequestration in pastures; p_0 – carbon price in the present; p_6 – carbon price at the end of the project.

The second possible assumption is that the PCF would pay farmers in a yearly basis. This is translated by the equivalente to Equation (4.6), which is

$$\sum_{t=1}^{5} p \cdot (1+r)^{-(t+1)} \cdot C(t) + p_6 \cdot (1+r)^{-6} \sum_{t=1}^{5} C(t) = p_0 \cdot \sum_{t=1}^{5} C(t) . \tag{4.8}$$

In this case, we obtain p using the expression

$$p = \frac{\left[p_0 - p_6 \cdot (1+r)^{-6} \right] \cdot \sum_{t=1}^{5} C(t)}{\sum_{t=1}^{5} (1+r)^{-(t+1)} \cdot C(t)} . \tag{4.9}$$

In this case, the price p depends of the total carbon sequestered. For any $r > 0$, $\sum_{t=1}^{5} C(t) > \sum_{t=1}^{5} (1+r)^{-(t+1)} \cdot C(t)$, which means that p increases with the total carbon sequestered. And price p also depends of the distribution of sequestration during the project. For any $r > 0$, the quantity $\sum_{t=1}^{5} (1+r)^{-(t+1)} \cdot C(t)$ is maximum when sequestration occurs soon (since the amount sequestered is subjected to a smaller discount). Therefore, if more carbon is sequestered early than late, the PCF will pay less for each ton of carbon.

Again doing the same analysis, and using for C sequestration the average sequestration potential of 5 t $CO_2.ha^{-1}.yr^{-1}$, we obtain results in Table 57. Overall results are quantitatively similar to those in Table 56, albeit relatively lower. This second approach is also approximately independent of the total carbon sequestered.

Table 57 – Maximum carbon price (paid yearly), depending on the discount rate.

	Scenario	r			
		1%	3%	5%	7%
p (€.t⁻¹)	p_0 = 20 € p_6 = 20 €	1.21	3.66	6.15	8.71
	p_0 = 15 € p_6 = 15 €	0.90	2.74	4.62	6.53
	p_0 = 25 € p_6 = 25 €	1.51	4.57	7.69	10.88
	p_0 = 15 € p_6 = 20 €	-4.00	-1.97	0.09	2.18
	p_0 = 20 € p_6 = 15 €	6.11	8.36	10.68	13.05

r – discount rate (PCF); p – price paid by the Portuguese Carbon Fund for carbon sequestration in pastures; p_0 – carbon price in the present; p_6 – carbon price at the end of the project.

Therefore, since it simplifies calculations and graphical representations, and the values are very similar, we opted for the first assumption (payment in the sixth year). From here on, we also assume constant yearly prices of 20 € for both p_0 and p_6 (since they are intermediate estimates).

Note that considering that all carbon will be lost in 2012 is a very conservative approach. It is highly unlikely that all farmers will till their plots. This is important because if we assume that only a fraction α of the carbon which was accumulated during the project is lost after 2012, then the price p will increase, because in that case

$$p' = \frac{p_0 - \alpha \cdot p_6 \cdot (1+r)^{-6}}{(1+r)^{-6}} . \tag{4.10}$$

Since $\alpha > 0$, $p' < p$. By assuming that $\alpha = 1$ in the calculations above, we are considering the worst case scenario.

In this case, to the PCF, every ton of carbon paid at this price is equally worthwhile, since marginal costs are constant. Therefore, the demand curve[51] would be a horizontal line. Considering that there were no physical constraints, Portugal would be interested in financing sequestration in SBPPRL at the prices stated before until the KP deficit was compensated (3.73 Mt CO_2e). However, this is not the case. Carbon sequestration supported by the PCF has to be additional to carbon sequestration considered in PNAC, which is 0.5 Mt CO_2 for cropland and grassland management. The Portuguese Rural Development Programme predicts a supported area of SBPPRL of 70 000 ha, which we will assume to be the cut-off point, i.e., the point after which all sequestered carbon is additional[52].

However, at the time of this study, we considered that, for a matter of equality and justice, the PCF would predictably have to pay all farmers for the carbon they sequester, regardless of Rural Development support. Such is to say that whichever price is fixed, it will also be paid to the first 70 000 ha, even though their sequestration was already planned and will not be additional.

For simplicity purposes, we consider that those 70 000 ha will always be accounted using a fixed conservative factor of 5 t $CO_2.ha.^{-1}.yr^{-1}$. Therefore, we must adjust the equilibrium point to cope with this fact. Table 58 shows the quantity of carbon that is not additional to PNAC's objectives, depending on the use of a factor or a model.

Table 58 – Maximum price paid by the PCF for carbon sequestration additional to PNAC's objectives.

Average sequestration ($t\ CO_2.ha^{-1}.yr^{-1}$)	Area (ha)	Not additional carbon ($Mt\ CO_2.yr^{-1}$)	Real price to be paid (€.$Mt^{-1}\ CO_2$)
5.0	70 000	0.35	$p^* = p \cdot \frac{C - 0.35}{C}$

r – discount rate (PCF); p – price paid by the Portuguese Carbon Fund for carbon sequestration in pastures; p – price paid by the Portuguese Carbon Fund if all carbon sequestered is additional; p* - real price for additional carbon; C – total carbon sequestered in pastutes.

Since when the PCF pays price p there is a certain quantity of carbon that is not being truly being accounted for Kyoto purposes, a real price must be defined, p*, which is the maximum price (€) per ton of carbon that the PCF is willing to spend. This real price is a function of the model used of the total quantity of sequestered carbon. The expressions in Table 58 show that, as C increases, this effect decreases. Therefore, we may plot a demand curve, as shown in Figure 35. In our demand curve, the PCF will not be interested in paying for carbon sequestration in SBPPRL until the base sequestration is met. From then on, it will be progressively interested in paying more, since the effect of the first 70 000 ha is diluted in the larger area of interested farmers. The upper limit for the real price is the maximum price, indicated in Table 56. But when the total deficit is met, the PCF will stop financing any project.

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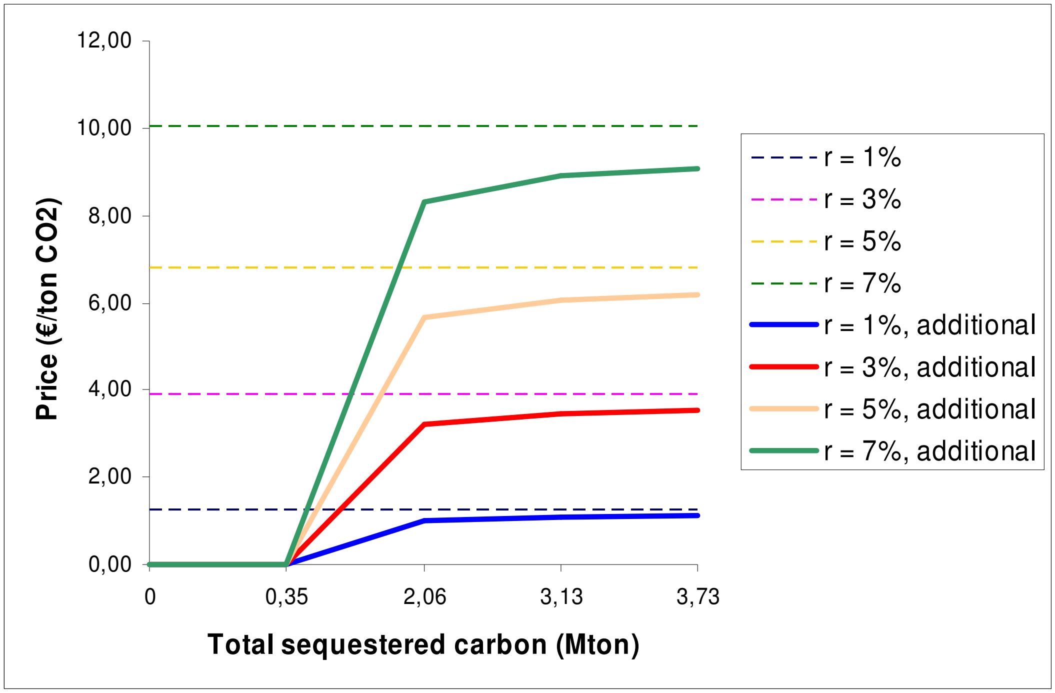
[51] We define supply curve for carbon sequestration in SBPPRL as a representation of the total carbon sequestered as a function of the payment per ton.

[52] Note that in practice it may not be the case, since the target is common for cropland and grassland management. It is the combination of both that must add up to 0.5 Mton, and so even it may not suffice to achieve the target for the implementation of SBPPRL. We will assume that cropland management achieves its part of the goal.

Figure 35 – Demand curve for carbon sequestration in SBPPRL, depending on the discount rate and total carbon sequestered.



r – discount rate (PCF).

4.5 Designing the Terraprima-PCF Project

The PCF launched the application stage in 2008. In October 2008, the firm Terraprima, taking advantage of its experience managing the EDP contract mentioned before, did make a proposal to the PCF for payments for carbon sequestration in SBPPRL, for a Project that would start in 2009.

The proposal, however, was quite different from what was expected at the time this work was carried out. Our work was based on the principle that no farmer with SBPPRL could be excluded, for fairness reasons. However, the PCF included in the regulation a mandatory clause that obliges all projects to start only after the proposal is delievered. Such is to say that only new pastures could be included in the proposal (pastures installed from 2009 on).

The next section shows the calculations which underlined the PCF project that was approved.

4.5.1 Proving adittionality

There are three additionality criteria set by the PCF. The first one is project additionality, which requires that the project does not happen without support from the PCF. The second one is regulation additionality, which implies that the project is not mandatory due to some regulation or legal obligation. The third one is financial additionality. This criterium requires the proponent to show that the level of the payment is enough to significantly influence its realization, but not so much that it completely depends on it to be maintained after the PCF support finishes. We turn our focus to this last criterium.

We said before than one explanation why the installation of new SBPPRL has decreased is that for some farmers (namely those with lower stocking rates and which do not recur to public support) installing pastures never has a positive economic balance. This may be further illustrated by depicting average costs and revenues from SBPPRL. We chose to make this balance for the scenario when the steer is sold after weaning, because it is the most common scenario in Extensity farms, and for consistency with the rest of this thesis. The balance comprises three instances: pasture installation, pasture maintenance, and costs related to livestock. We chose a time span for calculations of 10 years, since it is the approximate average time that well-managed SBPPRL can be sustained without new installation. Costs and revenues will be determined for a functional unit of 1 hectare.

Note that in these calculations we do not consider the future revenue from increases in production, as a consequence of SOM increases and better soil structure and fertility in general. This effect, though being very important, may not be visible to farmers during the first 10 years after pasture installation.

4.5.1.1. Pasture installation

The installation of SBPPRL requires the following field operations:

- First-year tillage, necessary to prepare the field for sowing and fertilizing;
- Liming, when pH(H_2O) is lower than 5.3, in order to correct soil acidity levels;
- Cover fertilization, using most commonly phosphorus, borax[53] and zinc sulphate[54];
- Sowing of the biodiverse seed mix;
- Rolling, after all other operations, to compact the soil.

The cost per hectare of these operations is shown in Table 59. Table 59 was built using field data from the farm Quinta da França for prices and Carneiro *et al.* (2005) for input quantities (Nuno Rodrigues, personal communication). Costs include inputs, transportation, machinery used and labour.

[53] Legumes are highly sensitive to borax concentration in soils. Therefore, borax should be applied in soils with concentrations lower than 0.40 ppm (INIAP-LQARS, 2006).

[54] Grasses are sensitive to zinc concentration in soils. Therefore, zinc should be applied in soils with concentrations lower than 1.4 ppm (INIAP-LQARS, 2006).

Table 59 – Average costs of operations required for the installation of SBPPRL.

Operation	Nr.	Labour cost (€)	Machinery cost (€)			Input costs				Total cost per operation (€)
			Variable cost	Total fixed cost	Ammortization	Input	Quantity	Unit cost (€)	Total cost (€)	
Lime transportation	1	8.20	8.88	7.37	6.80					31.25
Liming	1	1.08	0.90	0.86	0.80	Dolomitic lime (t)	2.0	65.52	131.04	134.67
Tillage	2	28.69	13.22	10.01	6.80					58.72
Rolling	1	14.34	13.22	10.01	6.80					44.37
Fertilizer transportation	1	4.78	2.34	1.95	1.79					10.86
Cover fertilization	1	9.56	7.33	6.16	4.42	Superphosphate 18 % (kg)	200.0	0.41	82.40	109.87
						Borax (kg)	10.0	0.35	3.52	3.52
						Zinc sulphate (kg)	7.0	1.82	12.72	12.72
Sowing	1	15.78	8.08	10.45	7.75	AC 700 Seed mix by Fertiprado[c] (25 kg)	1.0	107.83	107.83	149.89
									Total	555.87

This value must be added to other costs, which according to the GPP crop sheets are the interest of circulating capital (equal to 2.94 €, using an interest rate of capital of 1.5%) and general costs, which amount to 5% of the total costs with inputs (in this case, general costs are 16.88 €). Therefore, the total gross cost of installing a SBPPRL is approximately 575.68 €. We then converted this value into an annualization (10 years, discount rate 5%), which resulted in an average cost of 74.55 €.yr^{-1}.

However, PRODER supports 25% of the installation costs. Considering this support, the total net cost is 431.76 €, or 55.92 €.yr^{-1} (10 years, discount rate 5%).

4.5.1.2. Pasture maintenance

A well-managed pasture will only require one yearly event of fertilization with phosphorus. The costs respecting to this operation are shown in Table 60. Note that to assume that the same quantity of phosphorus is applied during 10 years is an overestimation. Towards the end of the period, if the pasture is well-managed, a cycle within the pasture will be established that recycles phosphorus flowing between the soil, plants and animals, and returning to the soil.

Table 60 – Average costs of operations required during maintenance of SBPPRL.

Operation	Nr.yr^{-1}	Labour cost (€)	Machinery costs (€)			Input costs (€)				Total costs per operation (€)
			Variable cost	Total fixed cost	Ammortizations	Input	Quantity	Unit cost (€)	Total cost (€)	
Fertilizer transportation	1	4.78	2.34	1.95	1.79					10.86
Cover fertilization	1	9.56	7.33	6.16	4.42	Superphosphate (kg)	200.0	0.41	82.40	109.88
									Total	112.74

The cost of phosphorus is particularly important. The reference unit cost in Table 60 for phosphorus is 0.41 €.kg^{-1}. Maintenance costs are 112.74 €.yr^{-1}. But if the price doubles, then maintenance costs become 195.14 €.yr^{-1}. Therefore, doubling the cost of phosphorus will increase 73% the maintenance costs, or around 50% the total costs related to pastures (maintenance and installation – installation also becomes more expensive).

This justifies why the price of an input such as phosphorus may be a crucial factor in the decision of a farmer to install SBPPRL. And it also explains why, in years such as 2008, farmers prices skyrocketed, and so many farmers already with SBPPRL skipped fertilization (Pedro Silveira, personnal communication). In the first years of settlement, this may be a serious threat to the permanence of the pasture.

There are other costs which are not directly for maintenance, but are also yearly costs. These costs are the rent (we assumed a rent of 39.90 €.ha^{-1}.yr^{-1}), circulating capital interest and general costs. Adding all contributions, the yearly costs with pastures are 165.45 €.ha^{-1}.yr^{-1}.

4.5.1.3. Costs related to livestock

We considered that each cow has a steer per year. The steer is weaned and sold at 6 or 7 months old. Several costs are associated with the existence of livestock in the pasture, namely silos for forage and feed storage (to complement cow feeding), establishment of plots (e.g fences), and labour costs with the farm manager and cowherd. The following costs were considered:

- While they are in the pasture, steers are fed only with milk. Cows, however, are fed by the pasture itself during its most productive monts (7 months) and by feed during the rest (5 months). As a simplification, we assumed that the supplementation is entirely provided by silage maize, and that each cow consumes approximately 2.6 t.yr^{-1}. Assuming an indicative price of 35 €.t^{-1} for silage maize, the total cost with feeds is 92 €.yr^{-1}.
- Three silos, which store silage maize for 62 cows, cost 10 000 € and are paid in 12 years.
- Setting a cow plot of about 65 ha, costs about 11 900 €, which are paid in 20 years. The fences for the plot cost about 18 200 €, paid in 10 years.
- A full-time cowherd can manage a plot of around 150 cows. The average salary of a full-time cowherd is 10 545 €, including social expenses.
- The farm manager spends around 10% of its working time managing a farm with 62 cows. The estimated average cost of that ammount of time is 1 400 €.
- Sanitary costs are around 9.3 €.cow^{-1}.yr^{-1}.

All values were annualized using the respective time frame and a discount rate of 5%. Even assuming the average figures shown before, the cost for each farm is still very dependent on the number of cows and total farm size, which is the same to say that it is very dependent on the stocking rate (LU.ha^{-1}). Therefore, we spawned cost scenarios for six possible stocking rates: 0.15; 0.3; 0.5; 0.7; 1.0 and 1.5 LU.ha^{-1}.

4.5.1.4. Revenue from SBPPRL

There are two sources of revenue for farmers. The first one is a direct payment for each breeding cow of 230 €.cow^{-1}.yr^{-1}. The second one comes from selling a steer each year. The price is highly volatile and variable, and therefore we studied three scenarios: 250 €, 375 € and 500 €. Therefore, the revenue depends on the cow stocking rate (more cows equals more steers per hectare) and on the price itself for which each steer is sold.

Note that we did not consider as source of revenue the specific payment for SBPPRL from PRODER. Only organic or integrated production farmers are eligible for this support, and those are not only the minority, but also have different costs and revenues from those shown here. We also did not consider activity maintenance payments[55] since they are equal for SBPPRL and other alternative land uses (natural pastures or cropland).

4.5.1.5. Support from the PCF

Farmers who install SBPPRL in 2009 will receive 50 €.ha^{-1} during the first 3 years after installation. We annualized this value for 10 years with a 5% discount rate, in order to make it comparable with the other costs and revenue. It is then equal to 17.59 €.ha^{-1}.yr^{-1}. As for farmers who install SBPPRL in 2010, they receive 75 €.ha^{-1} during the first 3 years after installation. Again annualizing this value, we obtain 12.01 €.ha^{-1}.yr^{-1}.

4.5.1.6. Final balance

Table 61 reviews and synthesises all the previous information for costs and revenue.

Table 61 – Synthesis of costs and revenue of producing steers in SBPPRL.

	Item	€.ha^{-1}	Item	€.ha^{-1}
Revenue	Payment from the PCF – Sowing in 2009	17.59 €	Steer sale	250 € - 500 €
	Payment from the PCF – Sowing in 2010	12.01 €	Support for breeding cows	230.0 €
Cost	Maintenance costs	165.5 €	Sanitary costs	18.5 €
	Installation costs (without support)	74.6 €	Cow feed	91.4 €
	Installation costs (with support)	55.9 €	Silos for maize silage	18.2 €
	Fences and other plot costs	51.0 €	Cowherd labour	70.3 €
			Manager labour	22.6 €

Table 62 shows why we needed to generate several scenarios throughout the previous exposition of the costs and revenues of SBPPRL. For example, for each class of farm size and number of cows there is a different balance (balances for all scenarios are shown in Appendix V – Economic balances for steer production in SBPPRL). We

[55] "Regime de Pagamento Único", in Portuguese.

now must determine the average scenario for Portuguese farmers with pastures. In order to do so, we make the following assumptions:

- Within each stocking rate class, 80% of farmers receive the installation support from PRODER and 20% do not;
- 25% of farms support a stocking rate of 0.5 $LU.ha^{-1}$, 60% of farms support 0.7 $LU.ha^{-1}$, and 15% of farms support 1.0 $LU.ha^{-1}$;
- 25% of steers are sold for 250 €, 50% for 375 €, and 25% for 500 €.

Under these assumptions, we obtain the global balance in Table 62. On average, installing SBPPRL has a negative economic balance, on a yearly basis and for a 10 year time frame. However, even though support from the PCF is only received during the first 2 or 3 years, it is enough to make the balance positive, both for farmers sowing in 2009 and in 2010.

Table 62 – Final balance between costs and revenue for SBPPRL.

Scenario	Balance ($€.ha^{-1}.yr^{-1}$)
Without support from the PCF	-9.2
With support from the PCF (sowing in 2009)	8.4
With support from the PCF (sowing in 2010)	2.8

It should be noticed that these calculations were made under the assumption that prices will be constant for 10 years. The expected value for each of the costs and revenues considered here has a very high variance, but since the farmer must decide at present whether to install pastures or not, and he has no information available as to how prices will evolve, it seems plausible to use present values and assume them constant throughout the time frame considered.

4.6 How much carbon will be sequestered with the PCF project?

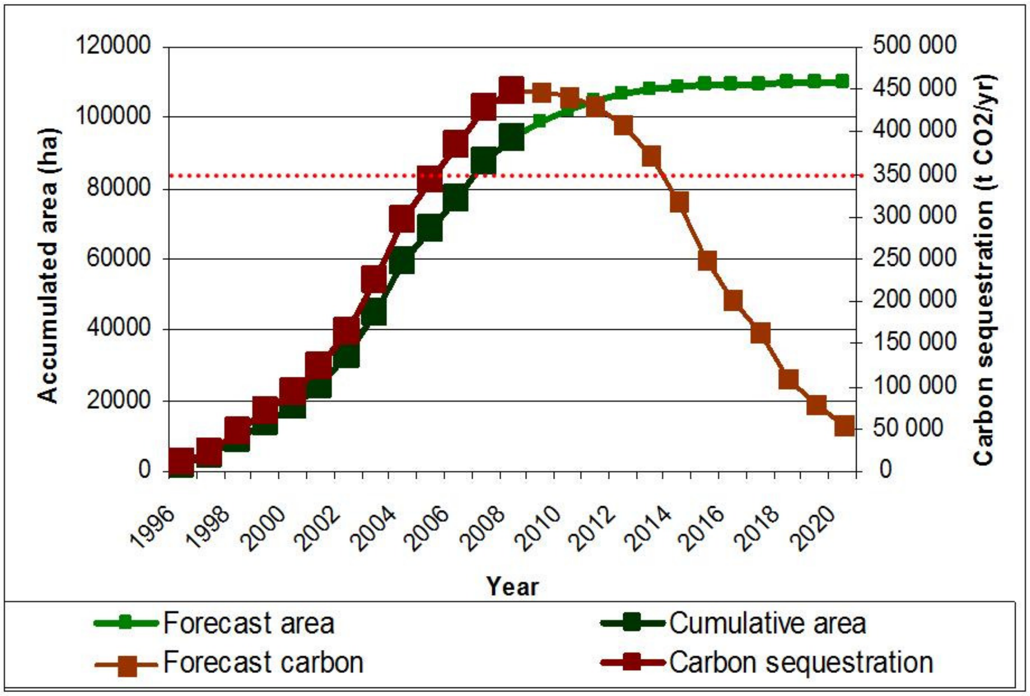
The project estimates the installation of 42.000 hectares, half of which in 2009 and the other half in 2010. 2011 was left off because farmers would only benefit of the support for one year. The area estimate was obtained from inquiring farmers from the Extensity Project and associations of farmers.

Considering the payments refered above (50 and 75 $€.ha^{-1}$, for installations in 2009 and 2010, respectively), farmers will be paid around 5-6 € per ton of CO_2 sequestered, and the project will be responsible for the sequestration of about 0.91 Mt CO_2[56].

To show how this project changes the reference situation, we can use the logistic model established in Section 1.4.4 for the area increase of SBPPRL in the absence of specific payments. This model allows us to forecast how much area would be installed in the future. Multiplying the area of SBPPRL by the specific sequestration factors determined in 3, and summing the yearly contribution of pastures from all ages, we obtain Figure 36. The PNAC target is shown in the dotted line. We can see that, throughout the whole period, it would be complied, but from 2012 on the area would stabilize and carbon sequestration would decrease steeply as the pastures age.

[56] Results for SOM increases in SBPPRL from the model in Teixeira *et al.* (2008), and respective carbon sequestration equivalent determined in Teixeira (2008), were used in these calculations.

Figure 36 – Baseline area and carbon sequestration of SBPPRL.



The PCF project will cause a structure break in this series, as Figure 37 shows. The area installed per year has increased until 2005, and then started decreasing, following a trend that quickly approaches zero. Due to the PCF project, more are will be installed than in any other year in the records. Figure 38 shows the accumulated area of SBPPRL obtained from Figure 37.

Figure 37 – Area of SBPPRL installed yearly, observed, modelled and due to the PCF project.

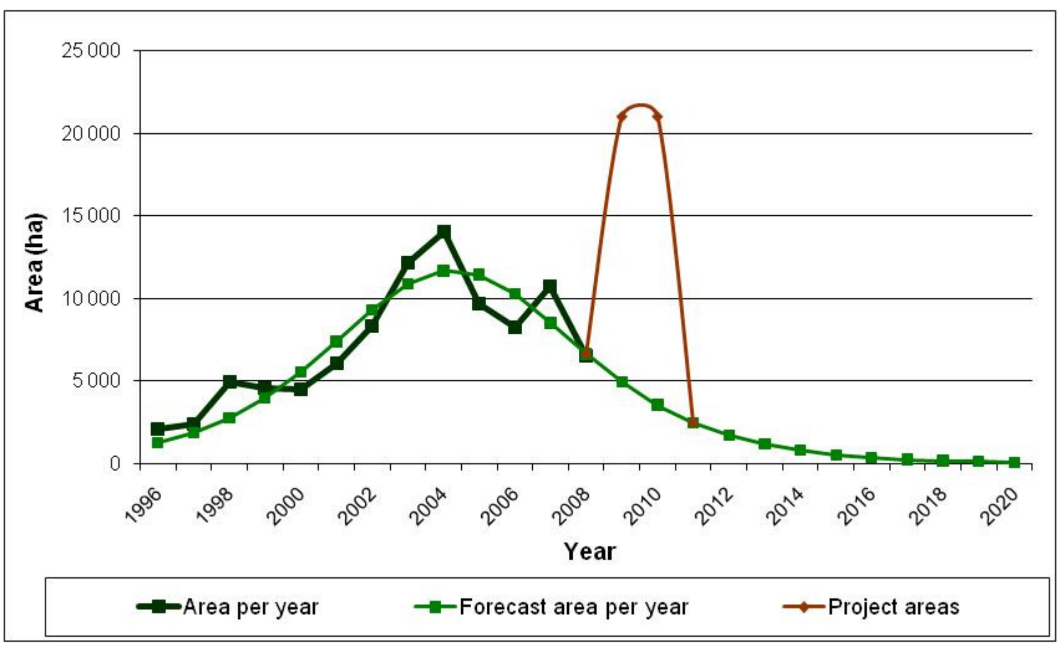
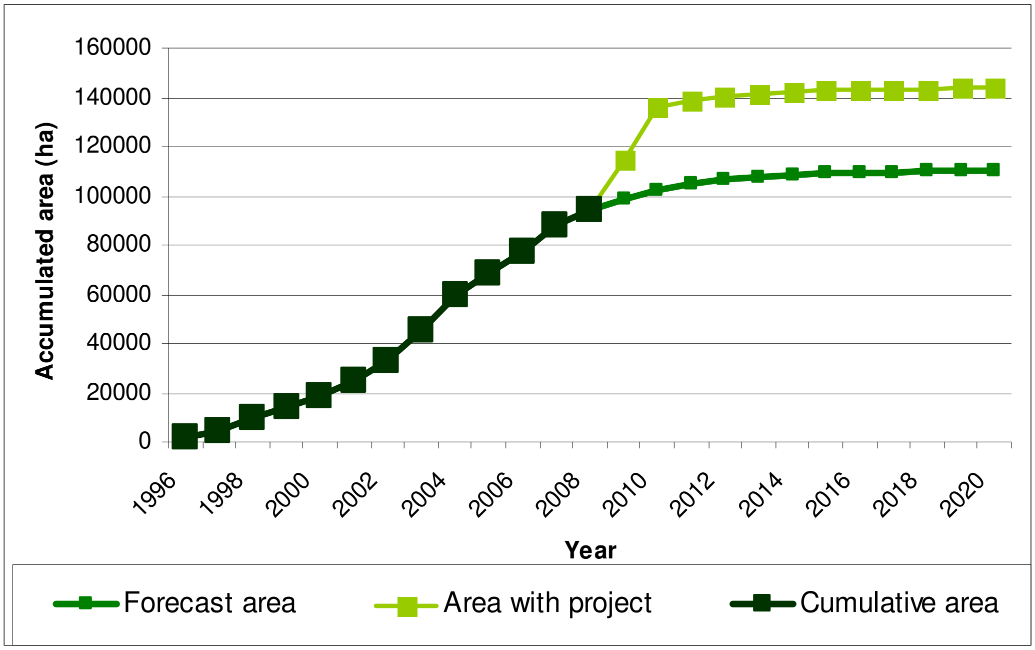


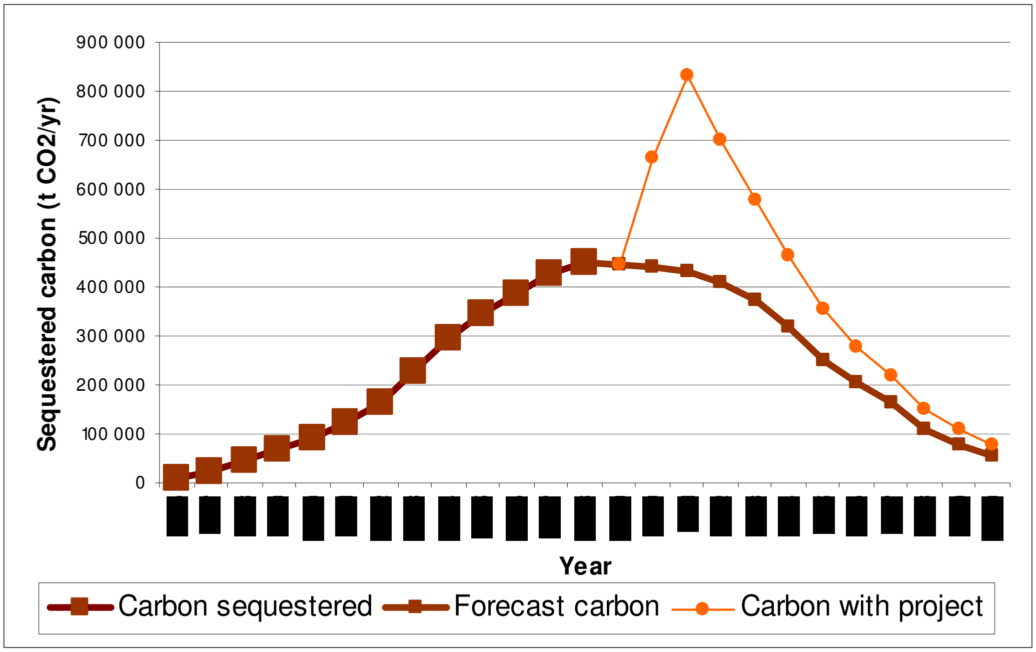
Figure 38 – Accumulated area of SBPPRL, observed, modelled and due to the PCF project.



The result in terms of total carbon sequestered per year is a very high increase from the expected sequestration that would occur in the absence of the project, as shown in Figure 39. This additional amount of carbon sequestered continues after 2012. If we assumed that all farmers kept their pastures, than an extra 0.67 Mt CO_2 would be sequestered between 2013 and 2020. This means that each ton sequestered has a lower price to the Portuguese State than it seems, since there will be free additional carbon remaining after the end of the Kyoto period.

Note that we considered for calculations in Section 4.4 that all area would be lost after 2012. At the time, we argued that it was a very conservative approach, since farmers will hardly change land use after such an investment (note that we studied viability for 10 years). Farmers in the PCF project are techinally supported to better manage their pastures. This technical support guarantees that the output of the pasture is maximized, and transferes knowledge to farmers which they may use for years to come. This fact, together with the financial support, almost nulls the risk of complete or almost complete loss of area.

Figure 39 – Total carbon sequestered per year in the area of SBPPRL, observed, modelled and due to the PCF project.



A final word goes to notice that the PCF will support SBPPRL just due to carbon sequestration, but Portugal will benefit from the other environmental and socio-economic positivie effects for years to come. Other ecosystem services will remain as externalities until new markets are created and new work is done to create options for the valuation of those services.

4.7 Synthesis of results and discussion

In this chapter we began by making a simple economic comparison between SBPPRL and NG. We monetized the environmental impacts obtained in the previous Chapter, and then internalized them in the economic balance. We found that SBPPRL represent an overall lower cost than NG, because they have lower economic costs and also lower environmental costs. It was curious to notice that environmental costs are much lower than the direct economic costs. We were then sure that SBPPRL were a win-win solution.

We then turned to improving the benefits of SBPPRL. We briefly mentioned the valuation that consumers make of meat products produced in SBPPRL, only to switch then our focus to payments for public goods. We studied the issue of resource conservation contracts, namely contracts for carbon sequestration in SBPPRL. We referred the case of the private contract established between EDP and Terraprima, and introduced the possibility of the PCF to incentive farmers to adopt this system.

In all calculations, we used the concept of NPV. Fist, we determined the demand curve, which is a horizontal line that translated the fact that the Portuguese state will finance every ton of sequestered carbon in pastures for the same price, as long as it is lower than the alternative. This result is independent of the contract scenario, but is extremely dependent of the discount rate used, due to the fact that carbon sequestration is only temporary.

This study, however, assumed that all farmers had to be paid. The regulation of PCF imposed that only new SBPPRL could apply. Therefore, the actual project was different from this study. Under this project, 0.91 t CO_2 more will be sequestered in Portugal until 2012.

5. Conclusions

The last Chapter is mainly dedicated to conclusions, but also intends to make a bridge to future studies by stating the main limitations and unanswered research questions. We start by summarizing the main results obtained, in order to draw conclusions from their integration. We state the main conclusions and policy recommendations, before acknowledging the shortcomings of our work, and proposing ways to answer questions that we left unanswered. In the end, we will wrap our work up by listing the contributions of this thesis.

5.1 Summary of results and conclusions

At the present moment, there is an ongoing debate on which are the best strategies to mitigate climate change. Some of those strategies do not take into consideration a full scale sustainability assessment considering all environmental aspects and the economic trade-offs involved in the project. Some projects for emissions reduction or carbon sequestration have negative effects in other environmental themes, which doubtfully compensate the benefit in terms of carbon emissions. The best example of this is biofuel production, a subject we refer on Section 3.3 for the case of bioethanol from maize, and which is now widely regarded as an unsustainable option for climate change mitigation.

Therefore, optimum options for climate change mitigation should have other environmental co-benefits. For emissions reduction, this means that projects have to be economically competitive and not have significant risks (carbon capture and storage seems to be an example of a potentially hazardous and expensive project, while N_2O scrubbers in fertilizer industries a very positive project, which is also being paid by the Portuguese Carbon Fund). In the case of carbon sequestration in soils, the general principle of “green carbon” implies that all projects must provide widespread ecosystem services. Ecosystem services in Portugal are crucially dependent on agricultural sustainability. Sustainable land uses must guarantee quality food and other goods, thus increasing farmer revenues, and also co-produce environmental amenities.

Therefore, in this thesis, we study the ecosystem services provided by grasslands, focusing on carbon sequestration. This choice occurred for two main reasons. First, to study carbon sequestration one must first describe soil organic matter (SOM) accumulation in grasslands, and SOM is the key parameter to most environmental services. Second, SOM improves soil structure and restores fertility, which guarantees sustainable increases in production for farmers. Therefore, systems which accelerate SOM increases will have both environmental and economic positive effects. Since SOM increase is also the mechanism through which carbon is stored in soils, we obtain a bundle of co-effects that occur at the same time as carbon sequestration. If those co-effects are positive, than we have found “green carbon”.

It may still be asked: if there is a bundle of ecosystem services to be studied, why focus on carbon? Most of all, because it is the only environmental amenity with an established economic market. Therefore, chosing carbon sequestration in grasslands as the main object of study allowed us to finish the present thesis with a proposal for a connection with a policy measure, namely payments for carbon sequestered in domestic projects for emissions reduction by the Portuguese Carbon Fund (PCF).

Increasing economic revenue is one of the key aspects to obtain agricultural sustainability, since it breaks recent tendencies in land use change in Portugal, which has been moving towards abandonment and more extensive natural (unmanaged) grassland systems. Those have many problems pointed out in Chapter 0 and confirmed for some particular aspects in the next Chapters.

We studied three types of pastures: Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL), fertilized natural grasslands (FNG) and natural grasslands (NG). SBPPRL are a Portuguese innovation and have been installed in more than 90 000 ha in the country. Because SBPPRL are more productive, they support a higher stocking rate. We show that the increase in SOM is higher in SBPPRL than in the other grassland systems studied. SBPPRL soils are usually tilled before the pasture is installed (even though no-tillage is possible is some situations), and therefore lose SOM in the beginning. But they quickly recover, due to the increased grass root production, which is a result of the complementary effects of plant biodiversity and functional roles of legumes and grasses.

On average, SBPPRL increase SOM by 0.20 percentage points per year during 10 years. Since SOM increase in the mechanism through which athmospheric carbon is stored in soils, we could convert this increase to equivalent carbon, and found that SBPPRL sequester around 5 t CO_2 per hectare and per year. This is a very high potential. We studied carbon accumulation in other land uses, and found the only management practice that topps this potential is the use of no-tillage and mulching in cropland areas.

However, SBPPRL, when compared with NG, may have increased emissions from livestock, legumes and liming for pH increase. SBPPRL allow an increase in sustainable stocking rate. This may mean that more animals can be fed in SBPPRL plots, thus effectively increasing emissions. But it may also mean that farmers will use less concentrated feeds and keep animals in pasture plots longer, thus decreasing life cycle emissions from feed production. Therefore, emissions from livestock are crucially dependent on the scenario. Even the worst case, however, the sum of all increased emissions in SBPPRL is never high enough to null carbon sequestration. The level of sequestration is enough to guarantee that the SBPPRL, as a whole, is a carbon sink.

But what makes SBPPRL a good environmental and socio-economic policy to mitigate climate change is the fact that there is much more than carbon to it.

Due to SOM increases and to the fact that, unlike in NG, no tillage is required during maintenance for shrub control, less soil is lost in SBPPRL, and soil fauna is more abundant. Decreased erosion and improvements in soil structure mean that there is an improved capacity for holding water, which increases water availability for roots and decreases superficial runoff. This fact yields another positive effect. SBPPRL provide not only mitigation of greenhouse gas emissions, but also adaptation to climate change.

Climate change will alledgedly increase atmospheric CO_2 concentration, average temperature and precipitation (Schils *et al.*, 2008). Some authors argue that these effects will promote vegetable growth, hence increasing productivity and soil quality (Niklaus *et al.*, 2001). However, other authors believe that there are other limiting effects (such as the availability of nitrogen), and since temperature and precipitation distribution and variability will become more irregular, more adaptability to extreme circumnstances will be required from plant species. Existing models are insufficient to determine how soil.plant dynamics will evolve under stress or rapid change (Chapin III *et al.*, 2009).

But that is precisely one of the advantages of SBPPRL – their resilience. Biodiversity and grassland soils with a well established and diverse bank of seeds allows production of each species to adapt to climate circumnstances, and take full advantage of positive environmental factors. Soils with increased fertility and capacity to hold water will be more adaptable to a future with larger temperature and precipitation extremes. Furthermore, nitrogen fixation from legumes diminuishes the probability of grass productivity limitation.

Results for some other environmental themes also show a clear trend. Biodiversity clearly benefits from high SOM, even though results for birds are inconclusive. The territory for birds is larger, and the spatial scope in our study is the plot level, and that makes it harder to find a clear preference for any type of grassland. However, the least that may be concluded is that birds do not seem to prefer NG to SBPPRL. Results for life cycle effects are always favorable to the scenario in which SBPPRL are installed to replace commercial feeds, even though they require more inputs than NG. One very important input, however, is phosphate fertilizer, which has important impacts due to its production and transport.

Other effects, which were unaccounted here, may also favour SBPPRL. Due to livestock pressure, herbaceous competition and high SOM, shrubs seldom grow in SBPPRL. This could seem as one less source of biomass entering the system (since shrubs also use photosynthetic carbon to grow, when compared to the reference situation of NG, easily invaded by shrubs. However, in practice, this is not the case, due to fire and SOM mineralization in NG. While shrub growth inhibition in SBPPRL decreases fire risk, NG are highly susceptible to conduct forest fires. For that reason, NG must be intervened to control shrubs. This usually occurs using tillage. Tillage will mineralize most of the SOM pool, which had been improved using shrub biomass. And, even in that case, there are periodic stages (before each tillage) where fuel is maximum and risk is high.

We are now able to provide justification for the seven statements made in Chapter 1:

1. SBPPRL are more productive than natural pastures, both above and belowground. Higher belowground productivity is translated by SOM, which we showed to increase more in SBPPRL in Chapter 1.
2. Increased SOM and soil cover by pasture plants implies an improved soil structure due to SBPPRL, leading to decreased erosion, better capacity to hold water and consequent flood regulation. Less superficial runoff leads to decreased erosion as well, particularly at extreme rainfall events. This soil loss effect was modeled in Chapter 3, using the Universal Soil Loss Equation.
3. Since SBPPRL are more productive, they feed more animals, and so are typically exploited with high sustainable stocking rates. High stocking rates mean that animals control shrubs either by stomping or by using them in their feed, since shrubs are rich in fiber, to complement for excess protein from the consumption of legumes. Besides, higher productivity of grasses and legumes leaves fewer resources available for shrubs. we discuss these effects through the present thesis, and present data that supports them in Chapter 3.

4. Increased stocking rates in SBPPRL mean that livestock in pastures emits more methane, which is a GHG. There are also emissions due to biological fixation of nitrogen by legumes, and emissions from liming, a process required to increase the pH of soils (if they are too acid). However, the global carbon balance is positive, as we show in Chapter 3.
5. Since SBPPRL are more productive enough, there is less need to resort to concentrated feeds. These feeds are composed mainly of cereals and oilseeds, which required fertilizers to be produced, and which are then avoided. The LCA study of this effects was done in Chapter 3.
6. Higher SOM has a positive effect on soil fauna, but the effects for birds are so far inconclusive. We can at least say that SBPPRL are not worst than natural pastures, as we refer in Chapter 3.
7. High SOM increases mean that carbon sequestration occurs, and we account it in Chapter 3. We studied other land uses in the agricultural sector, and from those only the combination of no-tillage and mulching has a higher potential to sequester carbon than SBPPRL.

Therefore, as a whole system, the scenarios that we studied for SBPPRL (rainfed) or SIP (irrigated) increased soils with more SOM, thus sequestering carbon, more than their respective alternatives (NG or maize, respectively). They do so with positive environmental effects in all studied themes. We could not find any environmental impact theme where SBPPRL cause significantly more damage than NG. On the contrary, for most of them (carbon sequestration, soil loss) they provide environmental services. Therefore, as a whole, SBPPRL are a good all-around sustainable land use. Carbon sequestration is not traded-off with other environmental amenities, but rather a win-win effect. In SBPPRL, we obtain many environmental services in the same bundle. This is why we may coin carbon sequestered in SBPPRL as “green carbon”.

So, after showing that SBPPRL may be considered a sustainable land use, we turned to the possibility of obtaining economic incentives for farmers who install SBPPRL. This may be done in two ways: valuing products which along its production cycle were related to SBPPRL, and payments for the environmental services provided by SBPPRL.

Regarding the private goods produced in SBPPRL, namely meat, consumers seem to value the fact that these pastures are used to produce cattle, as they claim to be willing to pay more for it. However, commercial networks still have to be created to value meat products from livestock produced in SBPPRL. On the public goods provided by SBPPRL, carbon sequestration is the most directly valuable. After previous experience with a smaller scale project, an application of new SBPPRL areas was made to the PCF. We showed that, even facing the risk of non-permanence after the end of the project (in 2012), this project is worthwhile for the PCF. This happens because even if in the highly implausible case that all carbon stored in SBPPRL soil is lost in 2013, the economic benefit alone of buying time to acquire permanent reduction credits is already enough to justify paying for that carbon.

Turning to farmers, we find that, in a 10 year time frame, the revenue obtained from SBPPRL is higher or lower than the costs, depending on three crucial factors, which are the stocking rate in the farm, the total area of pastures in the farm, and the price paid for steers. Larger farmers with higher stocking rates will always have higher revenue from SBPPRL, but for smaller farms when lower prices are paid for steers SBPPRL are not economically positive. This fact justifies two observations. First, that the installation of SBPPRL has decreased its pace in the last years. This is because this system has already been installed in places where it is more prone to be profitable. Second, this justifies the need for payments from the PCF. The payment will effectively turn loss into profit by some scales of farmers, significantly improving the probability of adoption of this system.

The PCF will pay for carbon sequestered in 42.000 ha of new SBPPRL installed in 2009 and 2010. This may be a significant push to increase the area of SBPPRL in the future. In this project, farmers will be supported by technicians who will guide them in using the best management for pasture installation and maintenance. This means that, with this project, one last important issue in improving ecosystem serices will be achieved with SBPPRL, which is transfer of knowledge to farmers.

The overall conclusion we obtain is that SBPPRL are, in general, a more sustainable extensive system of meat production than natural pastures. They maximize the bundle of services provided by grasslands, and are an answer to the decline in ecosystem services in Portugal. Results show that SBPPRL verify most responses pointed to negative conditions and trends identified by the ptMA, namely:

- Changes to sustainable land uses – Our work shows that SBPPRL are a more sustainable land use than the alternative (NG). An increase in their area (due to the increase in revenue and policy changes) will yield widespread envrionmental benefits, one of which is carbon sequestration.
- Increased revenue for private goods – Consumers seem to value the principles of SBPPRL, even though no surveys specifically about this subject were done yet.
- Increased revenue for public goods – Projects in the voluntary carbon market are now available to farmers who correctly manage SBPPRL, and the PCF will pay for carbon sequestered in new areas. Besides, there is a differentiation in the public support to installation and maintenance of SBPPRL instead of natural pastures, which is also mainly due to their environmental benefits.
- Technical support and information management – Farmers in the PCF project receive technical support in the field for the installation and maintenance of SBPPRL.

5.2 Future research plan

In the previous section, and also throughout the present thesis, conclusions were presented identifying several caveats. Those were issues outside the scope of this thesis, or issues in which this thesis was inconclusive. We now turn to the proposal of research questions, suggested as future work, which directly tackle some of the most important lacunas in the knowledge on environmental services of grasslands.

5.2.1 Should we sow pastures all over the country?

Throughout this thesis, the spatial scope was always plot level. This was due to the fact that available data was collected at specific plots, and due to it we disregarded two important effects: interactions with other land uses in the zone, and specificity of the location of the plot.

Regarding interactions, landscape planning is needed. When discussing the application of SBPPRL in the country (for example, via the PCF support), one question arises: do SBPPRL remain a sustainable land use when sown over large continuous areas? Different areas could have different sustainable uses. A mosaic between natural and sown pastures may be better, in overall sustainability terms, than an entire zone of either one of them.

As for local specificities, in some very degraded NG zones, sowing pastures may not even be the best technical option. First, soils shoud be recovered (for example, by substituting tillage for shrub controls), and only then SBPPRL can be installed. Besides, there may be differences between opportunity costs. Some pasture areas have to be analysed relating to the alternative of conversion to forest, while for others the most plausible alternative use is conversion to cropland. For each of these areas, the cost-benefit environmental and economic balance will be different.

Therefore, the effects of large continuous areas of SBPPRL, as well as the regionalization of effects, are still to be determined. Field data would have to be gathered: on soil loss, biodiversity, and large scale effects on SOM and productivity from large areas of each grassland system. Those should be compared with specifically determined alternative land uses.

5.2.2 Is it plant diversity or functional group diversity?

As we have mentioned before in section 5.1, according to the literature, composition (number of functional groups) seems to be more important than functional group richness (number of species in each group). It is the composition that mainly determines the stability (resistance to disturbance) of ecosystem properties and ecosystem resilience (Wardle *et al.*, 2000).

However, in principle, the total number of species in SBPPRL should be important, since in heterogeneous Mediterranean soils each species will find better conditions to grow than others. It is always important to have more than one species with a given ecosystem function but with a different response to ecosystem changes (Hooper *et al.*, 2005). This is known as the “portfolio” effect. Besides, we would not even know which grasses or legumes should be sown, if we were to sow less species in SBPPRL.

In order to test this effect, an experimental setting should be devised using a SBPPRL plot and several plots with pasture composed only of one legume and one grass. The plots should then be compared in terms of aboveground productivity and carbon sequestration.

5.2.3 Other scenarios for sustainable stocking rate increase

We mentioned before, right at the beginning of this work, that SBPPRL are more productive than natural grasslands, and it is possible to increase the sustainable stocking rate. This led to the design of several scenarios to analyse, one of which was effective increase in number of animals. If there truly are increases in stocking rate in SBPPRL, there are several co-effects that must be studied. First, if we consider that the Portuguese self-provisioning level of meat does not change, then the higher stocking rates in SBPPRL could mean that we can feed the same number of animals in a smaller area. Therefore, there is some land sparing. The environmental and economic effects of these marginal lands which are abandoned for meat production must also be accounted.

There is another possibility, which is that Portugal substitutes meat imports with meat from the increased stocking rate in SBPPRL. In this case, there will be more emissions from livestock inputed to Portugal, under the current Kyoto framework. But there is an important economic effect that can balance out this environmental burden.

Therefore, an integrated, sectoral economic model is required to study what is most likely to happen after natural pastures are substituted by SBPPRL:

- Do stocking rates increase, or do farmers maintain the same stoking rate but decrease the concentrated feed supplementation (which depends on feed prices and meat prices)?
- If stocking rates increase, will the resulting meat substitute national or international meat (which depends on meat prices and logistic chains)?
- If stocking rates increase and national meat is substituted, what happens to the land which is no longer used for animal production (which depends on the opportunity cost of each field used for grazing)?
- If stocking rates increase and international meat is substituted, what is the overall economic balance and the excess in emissions (which depends on meat prices and provenience of imports)?

5.2.4 Eat meat or go vegan?

The study of extensive production has a curious side effect. Grazing animals are consuming a resource (grass) which does not directly compete with other sources of human feed (like cereals and oilseeds which compose aggregated animal feeds normaly are). Therefore, the commom energy efficiency argument for direct consumption of vegetable commodities is not applicable. It would only be applicable for intensive livestock production. Even though feeds are still required in extensive animal production, and the inputs for pasture installation and maintenance also use energy, so does the production of crop cultures.

It would be interesting, then, to perform the same kind of sustainability analysis, but applying it to compare vegetarian meals with meat produced in SBPPRL. An LCA approach would be particularly important, but direct modeling of the effects in farms would have to be particularly accurate. There are also differences between marginal effects or widespread effects (for example, if 5% of the population become vegetarian, or if the whole population becomes vegetarian). Land that would marginally change from meat to crop production would have different environmental effects of widespread ending of meat production (end of pastures). An economic assessment of trade effects would also show dramatic differences in the agricultural sector balance.

5.2.5 Trees or grasses?

We have refered in section 3.3 carbon sequestration in grasslands and croplands. One land use, however, was left off – forest land.

The comparison seems, at first, unfair, since forests and grasslands are managed with radically different objectives. The main product of the first is wood or woody products, and the second one meat or meat products. But the comparison is not farfetched because both uses are sometimes combined, like in the case of Portuguese *montado*. In this framework, pastures and grasslands are not different or separated, but rather extreme land uses in a possible continuum of combinations of both. Trees in pastures creates shaded areas, increasing available biota, and biodiversity within forests may also have positive effects (Huston and Marland, 2003).

Therefore, the following question could be asked: what is the optimum rate of tree cover in grassland systems? Or, to put the question the other way around, what's the maximum tree cover where the installation of pastures still has positive effects? This is the fair comparison, especially when speaking of carbon balance of land uses.

5.2.6 Reduced forest fire risk

During a forest fire, large scale emission of greenhouse gases occurs. Using SBPPRL has positive effects on fire hazard for two reasons:

- The grazed agro-forestry mosaic landscape interrupts large areas of forest monocultures prone to fire;
- Unlike in natural pastures, increased grazing pressure in improved grasslands stops invasion by shrubs, which are also good fuel for forest fires.

The use of improved pastures to control fires is already foreseen in several agro-forestry management plans. However, accounting for this effect, as well as monitoring it, is extremely complicated. Only inferences on greenhouse gases' emissions based on the direct comparison with similar burnt areas is possible.

A geostatistical model is necessary to determine the exact effect that SBPPRL plots have in zonal fire risk. The decrease in fire risk can only be accounted by employing statistical models, and taking into consideration the spatial location of the pastures.

There is also another possibility worth testing regarding animal control of shrub invasion. It has been suggested (Carlos Aguiar, personal communication) that livestock grazes shrubs that eventually come up in SBPPRL to compensate for excessive protein (due to the high percentage of legumes in their composition). This effect has to be discerned in test fields, accompanying grazing animals and registering their feeding habits.

5.2.7 Natural or artificial regeneration of *montado*?

We've mentioned that SBPPRL are less prone to be invaded by shrubs. This is due to animal stomp and also due to increased grass production. Plant diversity in itself decreases invasion, as noted by Fargione and Tillman (2005). They also show that some grassland C_4 plants strongly limitate nitrogen for invader's biomass growth.

However, this may also be a limitation for natural regeneration of forest areas. In agro-forestry areas, such as the *montado* system in Portugal, "invasive" plants are primary stages of natural succession that leads to new oak trees. By suppressing their development, there is no need for tillage operations to destroy some of their woody biomass, and there is no fire risk (since they are high combustible). Nevertheless, there is no natural regeneration either. Regeneration requires a natural grassland system, with the disadvantages we described throughout this thesis.

Therefore, it would make sense to devise an experimental setting that would study in two plots whether natural or artificial regeneration of *montado* is more efficient. The first one would be a natural pasture plot in a *montado* ecosystem, were a tillage operation would be performed to control shrubs. However, some shrubs would be protected, so that new trees could grow. The second plot would be a SBPPRL, and so there would be limited to none shrub growth. In this second plot, however, there would be artificial introduction of young trees (in the same number of the "natural" trees protected in the first plot), protected from livestock. The test should be maintained for several years, and in the end the number of trees that survived should be determined, and conclusions taken on which one was more successful.

5.2.8 Optimization of phosphate fertilizer use

Fertilization operations were crucial in the development of agriculture. Fertilizer inputs were essential during the green revolution, propelling food production ahead of population growth. But the environmental costs of pollution are also significant. Those impacts include the degradation of downstream water quality, eutrophication of superficial waters, winter and summer smog and also emission of N_2O, a greenhouse gas (Vitousek *et al.*, 2009).

Most of these problems are due to nitrogen fertilizers. In our work, as we have noticed in section 3.5.4, we found that phosphate fertilizers also have high life cycle impacts. Since they are crucial for the management of SBPPRL, in virtue of the dependence of legumes, their optimisation is crucial. This optimization should include a detailed study of transportation distances, types of fertilizer (single or triple superphosphate, for example), other micronutrients provided by each type of fertilizer, yield output and also economic cost-benefit analysis.

5.2.9 CO_2e emissions reduction due to reduced fertilizer use

As referred before, in section 1.4, the use of an extensive animal production system, using SBPPRL, may decrease the need for concentrated feed supplementation for animals. Therefore, the need for industrial processing of concentrate feed, as well as the need for ingredients, such as cereal grains and oilseeds, decreases. Since such crops are very demanding in terms of fertilization (particularly nitrogen), less concentrate feed demand also means less fertilizer use, and less emissions due to its production and application. We did not account this effect, since it depends on feeds used and actual reduction.

Beside this life cycle effect, the reduction in fertilizer use occurs directly, since improved grasslands are rich in legumes, and these capture nitrogen. Therefore, nitrogen fertilization is not required. Cassman *et al.* (2003) note that mineral nitrogen fertilizers are responsible for direct field emissions (N_2O) and indirect emissions due to fossil fuel consumption during their production and application. Improved grasslands may require the use of phosphorus, potassium and, eventually, other nutrient applications, but the reduction in life cycle use for feed ingredients (which also require phosphate fertilizers) approximately compensates the extra use.

5.2.10 There is so much more than carbon

Our study in this thesis was somewhat asymmetrical, since it privileged the carbon sequestration service over other services provided by SBPPRL. There was, however, a reason to this, which is the connection to the PCF done in Chapter 4. There is now a market for carbon, and so it was the best choice to develop this integrated work, from the sustainability analysis to policy options and economic incentive schemes.

But it is possible to establish the same connection for other ecosystem services of SBPPRL. It is possible and desirable to devise schemes for environmental payments for biodiversity, soil protection or water cycle regulation. To such purpose, the analysis done in this thesis for each of those other services would have to be extended. Since there is no established market for them (there is no "biodiversity fund", for example), that would require an innovative framework.

5.2.11 Closing the cycle – expanding the borders of the analysis

We mentioned before, in section 1.8, that the limits of our analysis (at the gate of the animal farm) account for the vast majority of environmental impacts. However, such may not be the case in economic terms. In fact, it must be recognized that meat produced in SBPPRL and natural pastures have different quality and value. As we mentioned in section 4.2, some surveys have already been done. But they did not report consumer choices on the pasture type itself.

More than inquiring consumers on their preferences, it would be important to study the dynamics of the consumption of both types of meat (from natural pastures and from SBPPRL). It would also be important to determine their impact (if any) on the diet of consumers. If one type of meat is richer in any nutrient or parameter (fibre, protein, digestible energy, etc.), then the equivalent quantity of other source must be considered "spared", and the corresponding impacts removed from the overall balance.

5.2.12 Estimation of errors and uncertainties

Finally, we should notice that we did calculate errors and uncertainties for LCA results. Furthermore, we did some sensitivity analysis when we changed our assumptions and checked if results would be maintained. Those processes altogether assure us of our conclusions.

However, the overall uncertainty of the sustainability analysis was not determined. In fact, when combining several methods of environmental analysis, it is unclear how uncertainty in results should be assessed. Therefore, we suggest as a verification of our results the development of analythical calculations of errors and uncertainties in this work.

5.3 Contributions of the thesis

The present thesis had research contributions and policy contributions. All the work presented in the thesis is original and was prepared as part of the PhD. (even though part of it was published in other instances), except when specifically noted.

The most impactful results had to do with the finding of the sequestration factors, namely for SBPPRL. Those factors are used today for Portugal to account the carbon sequestration is the exhisting areas of this system of grasslands. This work has been developed together with the Working Group to account carbon sequestration in Portugal from agriculture, forestry and other land uses, in the framework of the Kyoto Protocol. The Group is part of the National System of Inventory of Emissions by Sources and Removal by Sinks of Atmospheric Pollutants, and is coordinated by the Portuguese Environmental Agency. The sequestration factors obtained here are also used in the private voluntary carbon market projects, and were also used in the PCF project for new SBPPRL areas. The PCF project itself was also an offspring of the present thesis.

That choice of SBPPRL was important for the establishment of a specific agricultural support for SBPPRL in the Portuguese Rural Development Programme (PRODER) 2007-2013. Part of the PhD. work that led to the present thesis was the Strategic Environmental Assessment of PRODER.

Work done in this thesis was also published in several instances. The main publications are:

- 3 chapters in a book with peer review:
    - Rosas, C., **Teixeira, R.**, Mendes, A.C., Valada, T., Sequeira, E., Teixeira, C., Domingos, T. (2009). *Chap. 7: Agricultura*, in Pereira, H., Domingos, T., Vicente, L., Proença, V. (eds.), *Ecossistemas e Bem-Estar Humano: A Avaliação para Portugal do Millenium Ecosystem Assessment*, Escolar Editora, Lisboa, pp. 213-249.
    - Domingos, T., Valada, T., **Teixeira, R.**, Rodrigues, O., Rodrigues, N., Aguiar, C., Belo, C.C. (2009). *Cap. 19: Quinta da França*, in Pereira, H., Domingos, T. (eds.), *Ecossistemas e Bem-Estar Humano: A Avaliação para Portugal do Millenium Ecosystem Assessment*, Escolar Editora, Lisboa, pp. 661-684.
    - Pereira, H., Domingos, T., Marta, C., Proença, V., Rodrigues, P., Ferreira, M., **Teixeira, R.**, Mota, R., Nogal, A. (2009). *Cap. 20: Uma Avaliação dos Serviços dos Ecossistemas em Portugal*, in Pereira, H., Domingos, T. (eds.) (2009), *Ecossistemas e Bem-Estar Humano: A Avaliação para Portugal do Millenium Ecosystem Assessment*, Escolar Editora, Lisboa, pp. 687-716.
- 4 papers published in international journals with peer review:
    - **Teixeira, R.**, Domingos, T., Costa, A.P.S.V., Oliveira, R., Farropas, L., Calouro, F., Barradas, A.M., Carneiro, J.P.B.G. (2008). The dynamics of soil organic matter accumulation in Portuguese grasslands soils. *Options méditerranéennes – Sustainable Mediterranean Grasslands and Their Multi-Functions*, A-79: 41-44.
    - **Teixeira, R.**, Domingos, T., Canaveira, P., Avelar, T., Basch, G., Belo, C.C., Calouro, F., Crespo, D., Ferreira, V.G., Martins, C. (2008). Carbon sequestration in biodiverse sown grasslands. *Options méditerranéennes – Sustainable Mediterranean Grasslands and Their Multi-Functions*, A-79: 123-126.

- Valada, T., **Teixeira, R.**, Domingos, T. (2008). Environmental and energetic assessment of sown irrigated pastures vs. maize. *Options méditerranéennes – Sustainable Mediterranean Grasslands and Their Multi-Functions*, A-79: 131-134.
- **Teixeira, R.**, Domingos, T., Costa, A.P.S.V., Oliveira, R., Farropas, L., Calouro, F., Barradas, A.M., Carneiro, J.P.B.G. (2008). Soil Organic Matter Dynamics in Portuguese Natural and Sown Grasslands. *Ecological Modelling* (accepted, pending revision).

- 3 papers published in Portuguese journals with peer review:
  - **Teixeira, R.**, Domingos, T., Costa, A.P.S.V., Oliveira, R., Farropas, L., Calouro, F., Barradas, A.M., Carneiro, J.P.B.G. (2010). Dinâmica de Acumulação de Matéria Orgânica em Solos de Pastagens. *Revista da Sociedade Portuguesa de Pastagens e Forragens* (in press).
  - **Teixeira, R.**, Domingos, T., Canaveira, P., Avelar, T., Basch, G., Belo, C.C., Calouro, F., Crespo, D., Ferreira, V.G., Martins, C. (2010). Balanço de Carbono em Pastagens Semeadas Biodiversas. *Revista da Sociedade Portuguesa de Pastagens e Forragens* (in press).
  - Valada, T., **Teixeira, R.**, Domingos, T. (2010). Pastagens (sequestro de carbono) versus Milho (produção de bioetanol) – Análise Ambiental e Energética. *Revista da Sociedade Portuguesa de Pastagens e Forragens* (in press).
- 3 papers published in Portuguese journals without peer review:
  - Domingos, T., Marta-Pedroso, C., **Teixeira, R.** (2007). Projecto Extensity – O desafio da sustentabilidade. Revista Mais Ambiente, Nº4, pp.18-21.
  - Domingos, T., **Teixeira, R.** (2008). O Papel da Biodiversidade no Sequestro de Carbono em Pastagens. Revista Im))pactus, Edição n.º 11 - Os desafios da Biodiversidade e dos potenciais serviços ecológicos para as empresas e sector financeiro, pp. 17.
  - Domingos, T., **Teixeira, R.** (2008). The Role of Biodiversity in Pastures' Carbon Sequestration. Im))pactus Magazine, Issue nº 11: The Challenges of Biodiversity and the potential ecological services for companies and financial sector, pp. 17.
- 6 papers published in conference proceedings with peer review:
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Furthermore, the study of feeds and feed ingredients has been used in a thesis in Husbandry at the Évora University. The author of the thesis was Maria Maurícia Caeiro Rosado, and the dissertation was entitled "Contributo para a Integração da Componente Ambiental na Avaliação Económica de Sistemas de Produção Agro-Pecuários" ("Contibution Towards the Integration of an Environmental Component in the Economic Evaluation of Animal Husbandry Systems", in Portuguese).

Work done in this thesis was recognized three times:

- By the Organizing Committee of the XXIX Spring Meeting of the Portuguese Society of Pastures and Forages, which granted the "Progresso dos Pastos" ("Progress of Pastures") award, for a paster consisting on the analysis of SOM dynamics, similar to an earlier version of Chapter 1.
- By the Portuguese Order of the Engineers – Southern Region, which granted me an honorable mention, attributed in the contest for the Young Engineer Innovation Award 2007, for the work entitled "The Contribution of Sown Biodiverse Pastures in the Fight Against Climate Change", consisting of an earlier version of Chapters 1 and 3.
- By the Portuguese Association of Environmental Engineering, which awarded the PCF Project with the Gold Climate Network Award 2010.

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Afterword

So was Mr. Crespo right in his 2006 statement that he had been responsible for the sequestration of enough carbon to compensate the emissions of his family for generations to come?

According to the United Nations Statistics Division[58], average per capita emissions in Portugal from 1990 to 2006 were 5.55 t $CO_2e.yr^{-1}$. Mr. Crespo would only need to maintain slightly more than one hectare of SBPPRL per year to compensate the emissions of each member of his family. Just in his farm (Herdade dos Esquerdos, Monforte, Portalegre), he has 360 ha of them, some of which are more than 30 years old. More than enough area for generations to come.

But of course, as the person who developed the system of SBPPRL, we may give him credit for more than the area of SBPPRL that he owns.

Let us very roughly assume that these are the emissions of a Portuguese citizen during any given year (past, present or future), and are constant throughout his lifetime. Let us also assume an average life expectancy of 75 years. We conclude that the average lifetime emissions of a Portuguese citizen are 412.5 t CO_2e.

In 2006 alone, and considering the estimates in PNAC of 70 000 ha and the average factor of 5 t $CO_2e.ha^{-1}.yr^{-1}$, SBPPRL sequestered 350 000 t CO_2e.

Therefore, and if we give credit to Mr. Crespo for the existence of this system, only due to the pastures sown in 2006 in Portugal, enough carbon was sequestered to compensate the lifetime emissions of nearly one thousand Portuguese citizens.

[58] Available at the official website for the United Nations Millenium Development Goals in: http://mdgs.un.org/unsd/mdg/SeriesDetail.aspx?srid=751&crid=.

Appendix I – Alternative estimations of the SOM model

Finding different approaches

As we have mentioned before in Chapter 1, we used a simple statistical model to try to identify the SOM dynamics. This model was calibrated using available data. The model states that the mass percent balance of SOM is the difference between input and mineralization:

$$\frac{dSOM_{i,t}}{dt} = K_i - \alpha_i SOM_{i,t-1}, \tag{1}$$

where $SOM_{i,t}$ is the SOM concentration (%) in grassland type $i = \{SBPPRL, FNG, NG\}$ at time t, K_i is the SOM input in each parcel and period, and α_i is the organic matter mineralization rate.

There are many ways to estimate the parameters in Equation (1). We have shown several in Section 2.3.3. In the present Appendix, we show several more, namely:

- Linearization of the function vs. obtaining the analytical solution;
- Making the SOM input term depend on site-specific conditions, using precipitation and texture as a proxy (instead of the initial SOM concentration);
- Using results of soil samples of a different project for 2007 and 2008.

Linearized model

Base data used for model calibration were collected between 2001 and 2005. This period comprises the first five years after the beginning of the trials. Therefore, even if the dynamic pattern is a saturating exponential, our results may be indistinguishable from a pure linear trend. Therefore, we tried estimating a model consisting of constant increases. Such is to say that we considered:

$$\frac{d^2SOM_{i,t}}{dt^2} = 0, \tag{2}$$

which is equivalent to:

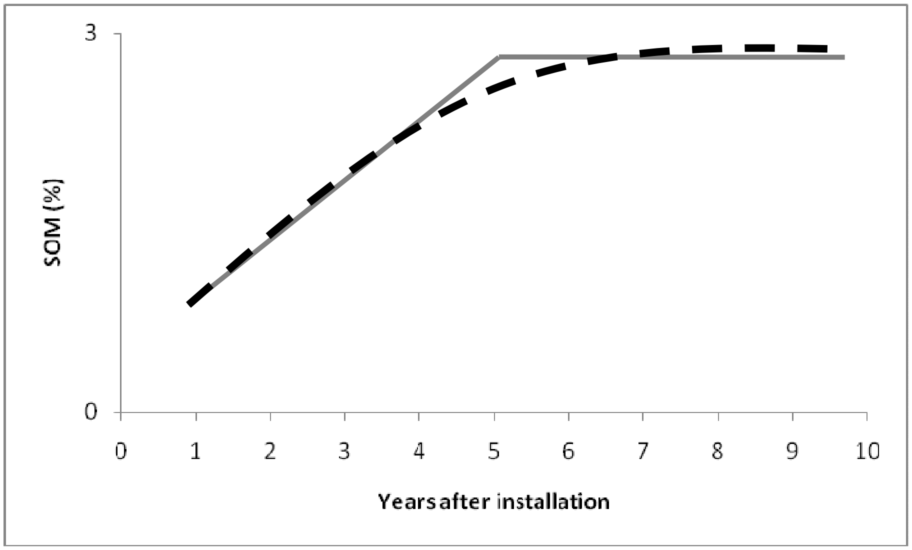
$$\frac{d}{dt}\left(K - \alpha SOM_t\right) = -\alpha \frac{dSOM_t}{dt} = 0. \tag{3}$$

The equality in Equation (3) is satisfied if:

$$\alpha = 0 \wedge \frac{dSOM_t}{dt} = 0. \tag{4}$$

This depicts a linear approximation of the saturating exponential. In the beginning, the mineralization rate is zero, and therefore the change in SOM over time is equal to *K*. From a certain point on, the second condition is true, and we reach the maximum. This approximation is shown in Figure 40.

Figure 40 – Difference between the analytical model and the linear approximation.



In the first year, the best approximation is obtained by considering $\alpha = 0$. From then on, we reach saturation, and the best fit is obtained from $\frac{dSOM_t}{dt} = 0$.

Therefore, for the first years, the model we estimate is

$$\frac{\Delta SOM_{i,t}}{\Delta t} = K_i \ . \tag{5}$$

Since $\Delta t = 1$ (yearly data), we obtain the linear model

$$SOM_{i,t} - SOM_{i,t-1} = K_i \, . \tag{6}$$

The specific input is equal to the average increase per year. Note that the results corresponding to this model are obtained directly by averaging the soil analyses data. Results obtained in this way only respect to the first years, and so cannot be used to extrapolate future gains in SOM.

Then, after the first years, when saturation is obtained, the linearization of the function implies that:

$$\frac{dSOM_t}{dt} = 0 \Rightarrow SOM_{i,t-1} = \frac{K_i}{\alpha_i} . \tag{7}$$

This is the (constant) value of *SOM* for all years when saturation is reached.

Specification of SOM input using other variables

In Section 2.3.3 we used the initial SOM concentration as a proxy for soil natural conditions. Other possibility is that the best proxy for site-specific natural conditions is not the initial SOM concentration, but meteorological and soil data. The first variable we tested is accumulated yearly precipitation (P_t), and the second term is the percentage of sand (S_t), as an indication of soil texture.

The basic Equations are now Equation (5) in the linear model, and Equation (6) in the analytic model:

$$SOM_{i,t} = K_i^{'} + \left(1-\alpha_i\right) SOM_{i,t-1} + pP_t + sS_t \,, \tag{5}$$

$$SOM_{i,t} = \frac{K_i^{'}}{\alpha_i}\left(1-e^{-\alpha_i}\right) + e^{-\alpha_i} SOM_{i,t-1} + \frac{p}{\alpha_i}\left(1-e^{-\alpha_i}\right) P_t + \frac{s}{\alpha_i}\left(1-e^{-\alpha_i}\right) S_t \,. \tag{6}$$

Enlarging the data pool

Project Biopast collected samples in the plots of Project Agro 87 in the years of 2007 ans 2008. Even though some plots were lost and no data exists for the year 2006, the results from Project Biopast may also be included in our analysis.

Procedure for calculations

Besides the missing values for 2002, we now also faced missing values in 2006. In the main text of the present thesis we had filled-in missing data using geometric averages. In this appendix, we used an alternative method, namely logarithmic regression.

In this second approach, we adjusted a logarithmic curve to each location and treatment in MS Excel, since we know, due to the analytical solution of the model, that SOM dynamics follows a saturating path. We attributed to the missing values the result of the logarithmic estimation for the respective year.

Since our regression models compare pairs of points (SOM_t, SOM_{t-1}), by using these methods instead of just using measured values we end up doubling the number of observations in the regression.

Then, using both filled-in and unfilled data tables, we followed the same procedure for calculations as shown before in Section 2.3.3, only this time for a different data set (two more years of sampling) and for the new approaches to the model.

Results of the calibration of the new SOM models

Alternative filling-in of missing values

Results for the filling in of missing values using a logarithmic regression are shown in Table 63. The R^2 of the adjustment is generally high (above 0.70). The main exceptions are two cases in which the fit for NG in Farms #3 and #5. In those cases, since increases are very low (SOM is stable), the logarithmic curve is not well adjusted.

Table 63 – SOM concentration in each type of pasture for experimental sites (0-10 cm) – missing data filled in using a logarithmic regression.

Farm No.	Grassland system	SOM (%)								Logarithmic regression	R^2
		2001	2002	2003	2004	2005	2006	2007	2008		
1	SBPPRL	1.55	2.51	3.05	3.60	3.80	3.32	3.40	2.57	y = 0.7387ln(x) + 1.9953	0.4495
1	FNG	1.30	2.15	2.60	3.40	3.00	3.16	2.90	3.36	y = 0.9213ln(x) + 1.5132	0.8029
2	SBPPRL	1.75	2.54	2.65	2.70	5.40	3.58	3.40	3.10	y = 0.9486ln(x) + 1.883	0.3391
2	FNG	1.95	2.82	3.00	4.50	3.50	-	-	-	y = 1.251ln(x) + 1.957	0.7082
2	NG	1.95	2.75	2.70	4.00	4.00	4.12	5.10	3.70	y = 1.2456ln(x) + 1.8893	0.7217
3	SBPPRL	0.33	0.73	1.20	1.63	1.60	1.70	1.60	1.98	y = 0.769ln(x) + 0.3269	0.8663
3	FNG	1.08	1.25	1.10	1.40	2.00	1.51	1.50	1.34	y = 0.2388ln(x) + 1.0801	0.0834
3	NG	0.28	0.74	1.10	1.20	1.15	1.47	1.60	1.74	y = 0.6648ln(x) + 0.2784	0.8387
4	SBPPRL	3.40	3.08	5.10	4.60	5.60	5.20	4.60	5.80	y = 1.1272ln(x) + 3.1786	0.6352
4	FNG	3.80	4.51	4.70	5.40	5.60	-	-	-	y = 1.114ln(x) + 3.7347	0.9542
4	NG	3.80	4.54	4.70	5.60	5.42	5.59	6.40	5.10	y = 0.9604ln(x) + 3.8696	0.6726
5	SBPPRL	0.65	0.94	1.00	1.28	1.50	-	-	-	y = 0.4914ln(x) + 0.6033	0.9157
5	FNG	0.55	0.87	1.10	1.15	1.25	-	-	-	y = 0.4378ln(x) + 0.5644	0.9865
5	NG	0.55	0.60	0.62	0.75	0.55	-	-	-	y = 0.051ln(x) + 0.5657	0.1483
6	SBPPRL	1.82	2.11	2.40	2.18	2.70	2.37	2.20	2.29	y = 0.2324ln(x) + 1.9491	0.3643
6	FNG	1.75	2.30	2.90	2.70	2.70	-	-	-	y = 0.6462ln(x) + 1.8511	0.7957
6	NG	1.75	2.33	3.10	2.40	2.98	3.11	3.20	3.35	y = 0.7086ln(x) + 1.8374	0.7676
7	SBPPRL	0.55	0.83	1.14	1.60	-	-	-	-	y = 0.7124ln(x) + 0.464	0.9059
7	NG	1.10	1.20	1.20	1.33	-	-	-	-	y = 0.1442ln(x) + 1.0929	0.8456
8	SBPPRL	0.80	1.40	1.54	2.08	-	-	-	-	y = 0.8501ln(x) + 0.7796	0.9441
8	NG	0.84	1.06	1.10	1.45	-	-	-	-	y = 0.3853ln(x) + 0.8064	0.8424

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; in the logarithmic regression, y is SOM (%) and x is year.

Table 64 shows results for the model calibration with the original data set (2001-2005). The unfilled data and geometric average filling-in columns are equal to those already shown in Section 2.3.3, and are here just for comparison with the new columns of logarithmic filling-in. We can see that results of both fillin-in methods do not differ, both in absolute terms and relatively to the results using unfilled data.

Table 64 – Results of the estimation of models (logarithmic filling-in).

Model	Grassland system	Using SOM_0?	Unfilled data						Filled data											
									Logarithmic filling-in						Filling-in using geometric averages					
			R2	K			Alpha	a	R2	K			Alpha	a	R2	K			Alpha	a
				SBPPRL	FNG	NG				SBPPRL	FNG	NG				SBPPRL	FNG	NG		
Pooled	All	No	0.952	0.422	0.171	0.136	-0.017		0.963	0.374	0.252	0.176	-0.027		0.966	0.370	0.248	0.165	-0.034	
		Yes	0.956	0.500	0.303	0.128	0.403	0.630	0.969	0.401	0.276	0.087	0.353	0.556	0.969	0.415	0.289	0.109	0.267	0.430
Specific	SBPPRL	No	0.760	0.413			-0.020		0.781	0.353			-0.036		0.794	0.353			-0.042	
		Yes	0.731	0.531			0.237	0.348	0.784	0.364			0.205	0.358	0.794	0.379			0.151	0.276
	FNG	No	0.810		0.428		0.071		0.870		0.454		0.051		0.886		0.442		0.043	
		Yes	0.841		1.083		1.105	1.434	0.907		0.500		0.538	0.707	0.912		0.508		0.443	0.566
	NG	No	0.899			-0.034	-0.105		0.924			0.032	-0.099		0.935			0.011	-0.113	
		Yes	0.920			-0.282	0.512	1.073	0.939			-0.067	0.328	0.625	0.943			-0.048	0.190	0.432

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

Using the linear model to forecast average SOM increases

According to the linear approximation and the analytical model we chose, we obtain the set of parameters presented in Table 65. The underlying assumption of the linear approximation model is that the SOM increase in each year is equal to K in the first years (unknown, depending on the dynamic pattern for each grassland system), and it is zero after that (saturation).

Table 65 – Statistics for analytical and linear approximation models' parameters for each grassland system.

Time	Grassland system	Parameters in analytical and linear models					
		Analytical solution			Linear approximation		
		K	a	α	K	a	α
First years	SBPPRL	0.415	0.430	0.267	0.28	0	0
	FNG	0.289			0.16		
	NG	0.109			0.03		
Later years	SBPPRL	0.415			$\alpha \bullet SOM_t$		$\frac{K_i}{SOM_t}$
	FNG	0.289					
	NG	0.109					

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

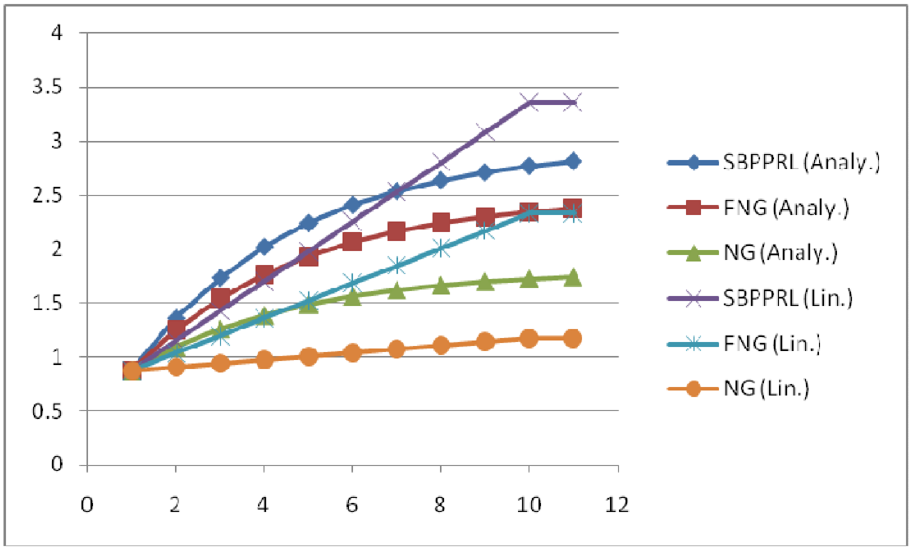
Using dynamic parameters in Table 65, we determined the average SOM increase in 10 years from each grassland type. We also assumed a starting hypothetical SOM concentration of 0.87%. Note that for the case of the analytic model we chose one set from those already obtained in Section 2.3.3, merely for comparison purposes. Results are shown in Table 66 and Figure 43, which depicts the increases in the three grassland studies in the first 10 years. In the case of the linear approximation, we assumed that pasture soils increase their SOM concentration for 10 years, before saturating.

Table 66 – Estimated SOM concentration per year in each model, starting from 0.87% SOM.

Year	Estimated SOM concentration (%)					
	Analytical model			Linear approximation		
	SBPPRL	FNG	NG	SBPPRL	FNG	NG
1	0.87	0.87	0.87	0.87	0.87	0.87
2	1.36	1.25	1.09	1.15	1.03	0.90
3	1.73	1.54	1.26	1.42	1.20	0.94
4	2.02	1.76	1.39	1.70	1.36	0.97
5	2.24	1.93	1.48	1.97	1.52	1.00
6	2.41	2.06	1.56	2.25	1.68	1.04
7	2.54	2.16	1.62	2.53	1.85	1.07
8	2.63	2.24	1.66	2.80	2.01	1.11
9	2.71	2.29	1.70	3.08	2.17	1.14
10	2.77	2.34	1.72	3.35	2.33	1.17
11	2.81	2.37	1.74	3.35	2.33	1.17
Average increase (percent points)	**0.19**	**0.15**	**0.09**	**0.25**	**0.15**	**0.03**

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; in the linear approximation, we assumed that pasture soils increase their SOM concentration for 10 years, before saturating.

Figure 41 – Simulated SOM concentration in each year, as estimated by a pooled data model, using data filled-in using geometric averages, starting from 0.87%.



NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

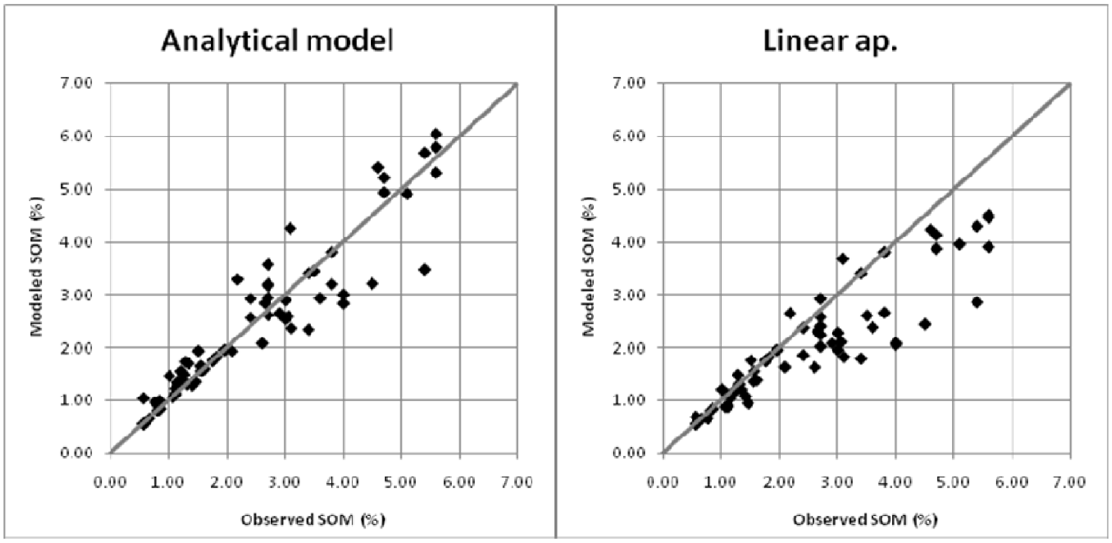
From Table 66 and Figure 43, we can see that the use of a simple average would imply a higher SOM concentration for SBPPRL, and a lower SOM concentration for both natural grasslands. For example, SBPPRL increase their SOM concentration, on average, by 0.19 percent points per year using the model and 0.25 percent points using the linear approximation.

Ultimately, a model is useful if it allows us to accurately depict data. In order to verify the adjustment provided by the model, we applied it to each farm. We considered the models that use the average parameters indicated in Table 65.

Results are shown in Figure 42, which plots all observed and simulated results. The closer that the points are to the 45º line, the better the fit. We can see that, for lower SOM concentrations, the linear model is a good approximation, but it systematically underestimates higher SOM concentration. The analytical model estimation is a good fit for all SOM concentrations (notwithstanding a slight bias to overestimate results for low SOM concentrations). However, for higher SOM concentrations, the differences seem unbiased, but variance is higher.

Our conclusion is, then, that the linear model does not provide a good fit to the data.

Figure 42 – Observed and simulated SOM concentration for all farms and grassland systems, using an analytical model (on the left) and the linear approximation (on the right).



SOM – Soil Organic Matter.

Testing for precipitation and percentage of sand (2001-2008)

In this section, we tested precipitation and percentage of sand as explanatory variables of SOM increases. In this instance, we tried to use the linear model again, this time with the complete data set of all sampling years (from 2001 to 2008). Results are shown in Table 67 for unfilled data, Table 68 for data filled with a logarithmic regression, and Table 69 for data filled-in using geometric averages.

Results show that the statistical fit of the models to base data, measured with the adjusted-R^2, is very high for the estimation of the analytical expressions, but very low for linearized first-differences. This is further evidence that the linear model is not a valid fit to the data.

We can also see that the introduction of the initial SOM concentration as an independent variable in the model increases the level of the parameters K and α. It also increases the R^2 of the estimation. However, the use of precipitation and percentage of sand as independent variables is never statistically significant.

Table 67 – Results of the estimation of models using unfilled data.

Model	Grassland system	Variables	Unfilled data – Analytical solution R^2	K SBPPRL	K FNG	K NG	α	*a*	*p*	*s*	Unfilled data – Linearized differences R^2	K SBPPRL	K FNG	K NG	α	*a*	*p*	*s*
Pooled	All	Only ct. and auto-regress	0.951	0.588	0.431	0.170	0.077				0.118	0.567	0.415	0.164	0.075			
		Ct., auto-regress and SOM(0)	0.960	0.670	0.719	0.183	0.697	0.955			0.286	0.483	0.518	0.132	0.502	0.688		
		Ct., auto-regress and prec+sand	0.962	1.510	1.405	1.114	0.116		0.000	-0.011	0.096	1.425	1.325	1.050	0.110		0.000	-0.010
Specific	SBPPRL	Only ct. and auto-regress	0.755	0.379			-0.006				-0.042	0.382			0.005			
		Ct., auto-regress and SOM(0)	0.747	0.501			0.444	0.649			0.023	0.406			0.359	0.524		
		Ct., auto-regress and prec+sand	0.827	0.699			0.030		0.000	-0.005	-0.213	0.681			0.030		0.000	-0.005
	FNG	Only ct. and auto-regress	0.822		0.339		0.045				-0.072		0.331		0.044			
		Ct., auto-regress and SOM(0)	0.826		1.073		0.942	1.199			0.175		0.695		0.610	0.776		
		Ct., auto-regress and prec+sand	0.933		0.659		-0.020		-0.001	-0.002	-0.116		0.666		0.020		-0.001	-0.002
	NG	Only ct. and auto-regress	0.851			0.399	0.174				0.127			0.366	0.160			
		Ct., auto-regress and SOM(0)	0.893			0.090	1.010	1.496			0.432			0.057	0.636	0.942		
		Ct., auto-regress and prec+sand	0.893			3.409	0.162		-0.001	-0.036	0.160			3.148	0.149		-0.001	-0.033

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

Table 68 – Results of the estimation of models using data filled with a logarithmic regression.

Model	Grassland system	Variables	Logarithmic filling-in															
			Analytical solution								Linearized differences							
			R^2	*K*			α	*a*	*p*	*s*	R^2	*K*			α	*a*	*p*	*s*
				SBPPRL	FNG	NG						SBPPRL	FNG	NG				
Pooled	All	Only ct. and auto-regress	0.963	0.451	0.470	0.417	0.089				0.130	0.433	0.450	0.399	0.085			
		Ct., auto-regress and SOM(0)	0.972	0.533	0.477	0.449	0.529	0.683			0.334	0.415	0.370	0.349	0.411	0.530		
		Ct., auto-regress and prec+sand	0.960	0.882	0.849	0.776	0.107		0.000	-0.006	0.118	0.834	0.803	0.734	0.101		0.000	-0.006
Specific	SBPPRL	Only ct. and auto-regress	0.754	0.648			0.167				0.074	0.599			0.154			
		Ct., auto-regress and SOM(0)	0.809	0.737			0.680	0.805			0.278	0.536			0.494	0.583		
		Ct., auto-regress and prec+sand	0.730	1.469			0.255		0.000	-0.010	0.057	1.295			0.226		0.000	-0.009
	FNG	Only ct. and auto-regress	0.858		0.327		0.033				-0.029		0.322		0.032			
		Ct., auto-regress and SOM(0)	0.893		0.336		0.418	0.596			0.225		0.275		0.342	0.487		
		Ct., auto-regress and prec+sand	0.810		0.820		0.083		0.000	-0.006	-0.121		0.787		0.079		0.000	-0.006
	NG	Only ct. and auto-regress	0.906			0.331	0.057				0.005			0.321	0.055			
		Ct., auto-regress and SOM(0)	0.925			0.363	0.431	0.579			0.205			0.294	0.350	0.471		
		Ct., auto-regress and prec+sand	0.916			-0.031	0.049		0.000	0.001	-0.061			-0.034	0.047		0.000	0.001

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

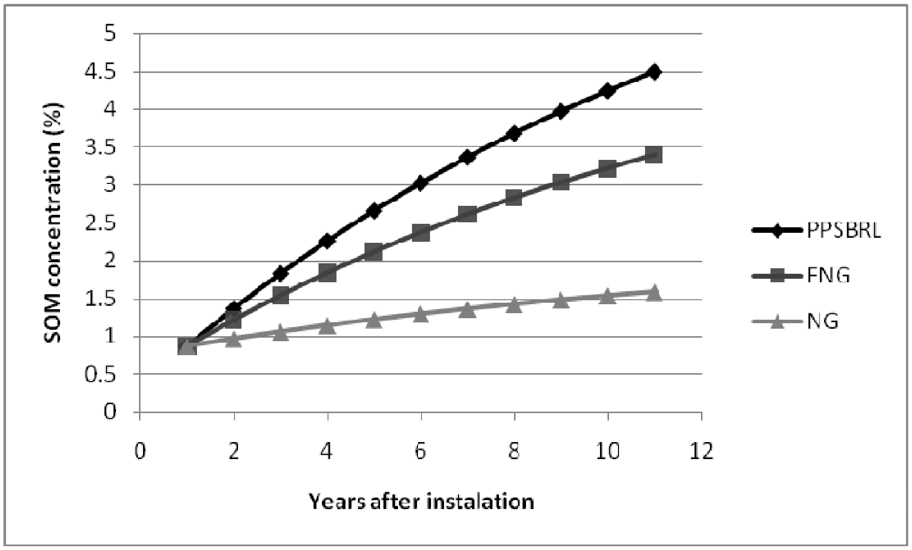
Table 69 – Results of the estimation of models using data filled with geometric averages.

Model	Grassland system	Variables	Geometric averages filling-in															
			Analytical solution								Linearized differences							
			R^2	K			α	***a***	***p***	***s***	R^2	K			α	***a***	***p***	***s***
				SBPPRL	**FNG**	**NG**						**SBPPRL**	**FNG**	**NG**				
Pooled	All	Only ct. and auto-regress	0.967	0.406	0.437	0.357	0.073				0.132	0.393	0.422	0.345	0.070			
		Ct., auto-regress and SOM(0)	0.974	0.407	0.418	0.258	0.440	0.608			0.322	0.330	0.339	0.209	0.356	0.492		
		Ct., auto-regress and prec+sand	0.964	0.909	0.897	0.834	0.084		0.000	-0.007	0.126	0.872	0.861	0.799	0.081		0.000	-0.007
Specific	SBPPRL	Only ct. and auto-regress	0.780	1.533			0.146				0.063	0.555			0.137			
		Ct., auto-regress and SOM(0)	0.821	0.592			0.556	0.680			0.238	0.456			0.426	0.521		
		Ct., auto-regress and prec+sand	0.762	1.350			0.232		0.000	-0.010	0.053	1.204			0.208		0.000	-0.009
	FNG	Only ct. and auto-regress	0.875		0.328		0.028				-0.029		0.324		0.028			
		Ct., auto-regress and SOM(0)	0.910		0.434		0.426	0.575			0.264		0.354		0.348	0.468		
		Ct., auto-regress and prec+sand	0.834		1.055		0.088		0.000	-0.008	-0.098		1.012		0.085		0.000	-0.007
	NG	Only ct. and auto-regress	0.924			0.262	0.037				-0.011			0.257	0.036			
		Ct., auto-regress and SOM(0)	0.938			0.121	0.355	0.557			0.181			0.102	0.299	0.469		
		Ct., auto-regress and prec+sand	0.928			-0.107	0.002		0.000	0.001	-0.111			-0.109	0.002		0.000	0.001

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

To understand what kind of dynamic plot these values correspond to, we use the first case shown (unfilled data, only constant and auto-regressive terms, pooled data). Beginning from an arbitrary 0.87% SOM concentration plot, Figure 43 depicts the increases in the three grassland studies.

Figure 43 – Simulated SOM concentration in each year, as estimated by a model using unfilled data, for an arbitrary situation starting from 0.87%.



NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

We decided not to use the complete data set with results for 2007 and 2008 in the main text of the thesis. The results were obtained in a different project, and therefore they do not guarantee the same standards of sampling. However, the previous analysis shows that results using the complete data set provide much higher increases of SOM. If they are true, then we are being conservative in the potential for carbon sequestration that we obtained in the Conclusions Chapter.

Appendix II – Estimation of CO_2e emissions from livestock

In the base scenario, which for Kyoto will be that of 1990, we assume that bovines are distributed as follows:

- Breeding cows (1 LU = 1 head·ha^{-1}) are fed in natural pastures, with a stocking rate of about 0.5 LU;
- All steers (1 LU = 0.6 head·ha^{-1}) are fed in an intensive system in stables.

We propose a scenario in which natural pastures are converted into sown pastures, and stocking rate is increased by 0.5 LU. We assume new animals will be:

- Steers transferred from stables;
- Cows in such quantity that they exist in the same number as steers. 0.5 LU refers to cows that were already in the pastures, and remaining cows are new.

Therefore, steers are now fed and finished also in an extensive system. Therefore, we must:

- Calculate emissons of cows installed in pastures and steers transferred;
- Subtract to that value the emissions avoided of steers in stables.

Therefore, we find the number of animals of each type considering that we intend to have the same number of each (x) in the end, and a total stocking rate of 1 CU. This translates into equation $x.1LU + x.0.6LU = 1LU$, and so $x = \frac{1}{1+0.6} = 0.625$.

Therefore, in the end, there are 0.625 steers·ha^{-1}, and in the beginning there were none. Initially there are 0.5 cows·ha^{-1}, and in the end 0.625 cows·ha^{-1}; variation is 0.625 - 0.5 = 0.125 cows·ha^{-1}.

Emissions from breeding cows in pastures:

We calculated the CH_4 and N_2O emissions from enteric fermentation and faeces of 0.125 cows·ha^{-1}, as shown in Table 70.

Table 70 – Emissions from breeding cows in pastures.

Gas	Emission factors		Stocking rate	Emissions	
	Enteric fermentation	Faeces		Enteric fermentation	Faeces
	kg·head^{-1}·year^{-1}	kg·head^{-1}·year^{-1}	head·ha^{-1}	kg·ha^{-1}·year^{-1}	kg·ha^{-1}·year^{-1}
CH_4	73[59]	2.156[60]	0.125	9.125	0.270
N_2O	0[61]	1.927[62]	0.125	0	0.241
CO_2e				191.625	80.331

CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

[59] Enteric fermentation CH_4 emission factor for breeding cows in grasslands, in 2004.

[60] Faeces CH_4 emission factor for breeding cows in grasslands, in 2004.

[61] IPCC (who establishes the Kyoto accounting method) does not consider N_2O emissions from enteric fermentation.

[62] Faeces N_2O emission factor for breeding cows in grasslands, in 2004.

Emissions from steers in pastures

We calculated the CH_4 and N_2O emissions from enteric fermentation and faeces of 0.625 steers·ha^{-1}, as shown in Table 71. Note that each steer only emits 0.6 of an adult.

Table 71 – Emissions from steers in pastures.

Gas	Emission factors		Stocking rate	Emissions	
	Enteric fermentation	Faeces		Enteric fermentation	Faeces
	kg·head^{-1}·year^{-1}	kg·head^{-1}·year^{-1}	head·ha^{-1}	kg·ha^{-1}·year^{-1}	kg·ha^{-1}·year^{-1}
CH_4	50.2[63]	0.679[64]	0.625	31.375	0.424
N_2O	0	0.659[65]	0.625	0	0.412
CO_2eq				658.875	136.593

CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

Emissions from steers in stables

We calculated the CH_4 and N_2O emissions from enteric fermentation and manure of 0.625 steers·ha^{-1}, as shown in Table 72. Again, note that each steer only emits 0.6 of an adult.

Table 72 - Emissions from steers in stables.

Gas	Emission factors		Stocking rate	Emissions	
	Enteric fermentation	Manure		Enteric fermentation	Manure
	kg·head^{-1}·year^{-1}	kg·head^{-1}·year^{-1}	head·ha^{-1}	kg·ha^{-1}·year^{-1}	kg·ha^{-1}·year^{-1}
CH_4	50.2	1.156[66]	0.625	31.375	0.723
N_2O	0	1.122[67]	0.625	0	0.701
CO_2eq				658.875	232.560

CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

Balance of emissions

Results are summarized in Table 29. Emissions' balance is obtained by subtracting avoided stable emissions from total pasture emissions. Therefore, there is an increase in emissions of $191.625+80.331+658.875+136.593-(658.875+232.560)=175.989$ kg CO_2-eq. This means that the avoided emissions from transferring steers from an

[63] The emission factor ranges from 60 for female steers (more than 2 years old) to 87.5 for male steers (more than 2 years old), in 2004. Emissions from animals between 1 and 2 years old range from 55.1 to 70.5. Therefore, the weighed average is 50.2.

[64] Emisson factor for steers from 1 to 2 years old and finishing beef calves (>2) in grasslands, in 2004.

[65] Emisson factor for steers in grasslands, in 2004

[66] Emisson factor for steers from 1 to 2 years old and finishing beef calves (>2) in stables, in 2004.

[67] Average steer emission factor from several manure management systems (from stables), in 2004.

intensive to an extensive system almost compensate the increased animal stocking rate that the system implies.

We then conclude that carbon balance is almost not affected by increasing animal stocking rate, as long as some animals are withdrawn from intensive breeding.

Appendix III – Environmental analysis of maize production

Introduction

International trade of agricultural commodities is common. Market globalization allowed trade rates between countries to increase, such that production may be transferred to the economically most appropriate sites in the World. Transportation thus becomes a relevant factor.

A question arises from this global fragmentation of agricultural systems: is it better, in economic and ecological terms, to transfer production to the places where it is most adequate, and then transport it to where it shall be consumed? Or, instead, is it better to reduce transportation to a minimum by producing nearby where it is mostly consumed? The answer to this question has deep implications in each country's economic structures, and can be posed in many forms and regarding all sorts of products.

To answer this question, a global life cycle approach is required, in order to comprehend not only the production and processing steps, but also the influence of both transportation steps, and finally determine the relative impacts of production and transportation. This method was strongly stimulated by the Communication on Integrated Product Policy (COM(2003) 302 final) of the European Commission. This policy intends to reduce the environmental impacts of products and services during their life cycle. The determination of the product groups with the greatest impacts was done by Project EIPRO – Environmental Impact of PROducts (Tukker *et al.*, 2006). They conclude that those groups are food and drink, housing and private transport. Minimizing the impacts of grain crops has an effect on food.

In the European Union (EU), grain crops cover over 40% of the cultivated area and are present in every member state (European Commission, http://europa.eu.int/comm/agriculture, 2005). From the beginning of the Common Agricultural Policy (CAP), their production was greatly promoted, in order to achieve self-sufficiency in Europe, as well as competitive food prices, through the intensification and specialization of agriculture (Avillez *et al.,* 2004). These policies were implemented with increasing mechanization, large irrigation and drainage systems, mass conversion of grassland to cropland, production in large monocultures, and intensive use of fertilizers and pesticides (EEA, 2003). Commodity prices were also kept high, as a guarantee of farmers' revenue.

In time, the environmental burden of both productivity and production increases became evident. Environmental impacts were considerable. Soil quality decreased, fossil fuel consumption grew, emissions to air and water caused relevant problems of acidification and eutrophication, large biodiversity loss was registered and many ecosystem values were degraded (Ventura-Lucas *et al.,* 2002). But there were also unwanted economic impacts. European prices increased in relation to world prices, and this was a large restriction to trade. Therefore, the EU, in the 1992 CAP reform, tried to end the unnecessary production surplus, thus bringing European crop prices close to those current in the rest of the World (Avillez *et al.,* 2004). Nowadays the choice for crop buying industries ranges from using local raw materials to importing from remote places where production is cheaper.

As a result, incentives to production for several crops began to decrease. One example of such was maize (*Zea mays* L.), which is an especially important food crop. In the EU, about 75% of all grain maize is used for animal feed, depending on price fluctuations, which in turn depends on the demand for human consumption. Virtually all commercial feeds for all types of animals contain maize or industrially processed maize by-products (INE, http://www.ine.pt, 2007). Maize is poor in proteins, but is extremely rich in starch. (EC, 2000). Therefore, it has high energetic content, and some farmers also feed their animals exclusively on grain maize that may be produced in the farm or bought elsewhere.

Maize is targeted by several agri-environmental measures in Portugal, namely integrated production and protection, and soil improvement and erosion prevention (direct sowing and minimum mobilization), and by the Nitrate Directive. This happens because the main environmental impacts of maize production are nitrate leaching and soil loss, especially when the lack of cover crops leaves the soil unprotected during winter. Rain may then damage the top layers of soil, and carry nutrients to lower layers of earth, and from there contaminate underground water resources and ultimately rivers. If irrigation is excessive, nitrate leaching also happens during the maize growing phase as a side effect. Fertilizers, as well as pesticides, are also responsible for emissions leading to acidification and for changes in the biological structure of the soil (EC, 2000).

Today's energy policy has turned its attention to maize as a possibility for biofuel production. This started whole new market possibilities for maize trade. Even though we will consider the optimization from the viewpoint of an animal farmer, we could as well consider it from the standpoint of an investor in a bioethanol factory.

In this paper we study the impact of grain maize production using a Life Cycle Assessment (LCA) tool, SimaPro 6.0. We characterize and select from several options regarding maize production, in terms of production zone, techniques and inputs. To assess the effect of transportation in the global impact of consumption, we consider a case study located in Beira Interior, which is a farm called Quinta da França. We chose this particular location since there is plenty of information available on how maize is produced there.

In the next section, we describe the method and data used. We also refer the specific changes to LCA considered. We then present the results obtained and analyse their uncertainty.

Method

LCA Tool

SimaPro 6.0 was developed by the National Reuse of Waste Research Programme and Pré Consultants of the Netherlands, and is widely used in assessing environmental performance. It consists of a data base of inputs and outputs from several processes and production of materials.

Therefore, the assessment of environmental impacts consists in the sum of impacts from each step of its life cycle. Impacts are then added by environmental themes. The total impact in each theme is then normalized and aggregated into a single impact indicator, usually using one of two methods: "Ecoindicator 95" and "Ecoindicator 99". Both methods aggregate impacts into a subjective and abstract unit called "Ecoindicator

Point", or Pt. Even though conclusions drawn are often similar (Luo 2001), it is important to use both, as they present different themes and a different conception.

"Ecoindicator 95" classifies, characterizes and normalizes the environmental impacts based on their contribution to several themes (Luo, 2001). The environmental aspects related to a given product are first aggregated into a number of effects caused, and those are then characterized according to the degree of damage inflicted on the environment; finally, these results are normalized into a single score, based on subjective evaluation (Goedkoop, 1998).

"Ecoindicator 99" is an update and extension of Ecoindicator 95, which emphasises its damage-oriented methodology by considering three areas of environmental damage: human health (measured in DALY – Disability Adjusted Life Years), ecosystem quality (expressed as PAF – Potentially Affected Fraction and PDF - Potentially Disappeared Fraction) and resource depletion (expressed as $MJ.kg^{-1}$) (Luo, 2001; Goedkoop and Spriensma, 2000).

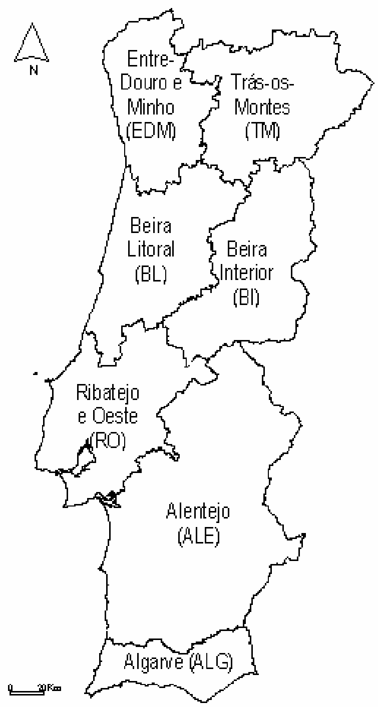
The main advantages of using LCA software such as SimaPro are the speed of assessment and the fact that its data base is very wide-ranging, since all existing inputs for any given activity are considered. However, many processes are country-specific, and the fact that it uses a foreign database is a strong limitation. Therefore, we decided to use SimaPro as the basis, but we incorporated national information whenever it was available or the impact resulting from the process was significant. Those methodological changes are described in the next sub-sections.

Analyzed zones and case study description

Maize production in Portugal was about 5.1×10^5 t in 2005, of which 2.3×10^4 t were exported. Portugal imports 1.2×10^6 t of grain maize, of which more than a third is from Argentina (FAOSTAT, http://faostat.fao.org, 2007). According to this data, the country's grain maize self-sufficiency level is about 30%.

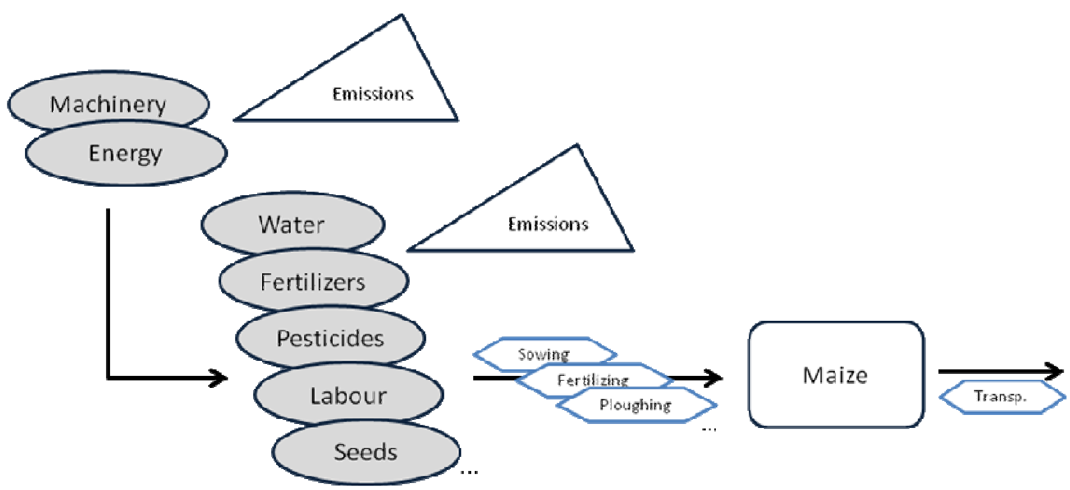
In Portugal, Entre-Douro e Minho (EDM), Beira Litoral (BL) and Ribatejo e Oeste (RO) (locations shown in Figure 44) are the zones with the largest area of cultivated grain maize. The zones with the highest productivity, in $t.ha^{-1}$, and/or total production, are Beira Interior (BI), BL and RO (INE, http://www.ine.pt, 2007).

Figure 44 – Agricultural regions in Portugal



We chose the most productive zones and characterized the production systems using the technical coefficients from crop fact sheets in GPP (2001). Then, every product entry (fertilizers, chemicals, …) and every action (sowing, ploughing, …) considered in the fact sheets were simulated using SimaPro 6.0. The system's frontiers are shown in Figure 45.

Figure 45 – System studied for maize production



We included a case study – Quinta da França (QF), in BI. We studied whether it would be an environmentally better option for QF to import maize (i.e. buy outside the farm) or to produce it. We also considered that Quinta da França could choose to produce maize with conventional tillage or using no-tillage techniques. Information regarding maize production in Quinta da França was collected at the farm.

Maize produced in Portugal is transported by road to QF. From BI itself, we assumed a distance of 50 km; from BL, we estimated the distance as being 100 km, and from RO 250 km. We also studied the impact of transportation by train. We assumed that transportation distance by railway is the same as by road. But there is additional transportation by road from the railway station (located in Covilhã) to QF. The distance between the Covilhã station and QF is 20 km. Since the other locations are generic regions, we assumed as a simplification that the farms are close to the station.

Since most imported maize comes from Argentina, we determined the average impacts of Argentinean production. We used information from Argentina, and maize production in the Pampas region. Information was obtained in an Argentinean article collection page, at http://www.elsitioagricola.com/maiz/actual/maiz.asp. We assumed that maize was transported from this region to the closest sea port by road. We considered that all ports in the area are within an average range of 200 km (Bahía Blanca, La Plata and Buenos Aires). We also considered sea transportation of 9500 km and road transportation of 200 km from the port in Portugal to QF.

The Argentinean information already corresponds to an optimized production with no-tillage. Not only are Argentinian soils and climate better to produce maize, but we are also comparing national options with maize produced using an environmental friendly technique. Therefore, we chose a good international option, in order to guarantee a fair comparison.

It should be noticed that all distances indicated in this section are only rough approximations, since we are dealing with generic regions and not specific locations, except the final consumption place. This choice is justified because we wish to draw generic conclusions regarding the impacts of transportation.

Base information

Relevant information for the cases studied is shown in Table 73. Comparing the productivity considered by each fact sheet (second row) with the average for the corresponding zone (third row), only the result for RO is coincident. The sheets are, still, a good approximation if we consider that they are overestimating productivity in the regions that, as results will show, are already the ones with a greater environmental impact (BI and BL). Table 73 also shows that the most expensive place to produce maize is BI (at the farm gate). Maize cannot be produced anywhere in Portugal with a positive revenue (without subsidies).

Since no sheet was available for no-tillage in Quinta da França, we built it by removing tillage operations from the conventional tillage sheet. This is an approximation, but since specific machinery for conventional and no-tillage is not present in SimaPro's database. We also assumed that productivity is the same for both methods (which is an empirical observation in QF).

We considered that no-tillage has an additional effect on soil carbon sequestration, by increasing soil organic matter. This effect is not permanent, but it is still a positive environmental impact. The potential for carbon sequestration by no-tillage is estimated as 0-3.0 $tonCO_2 \cdot ha^{-1} \cdot year^{-1}$ (ECCP, 2003). Calculations based on Carvalho and Basch (1995) for Portugal point to a sequestration potential of 1.8 to 2.5 t $CO_2eq.ha^{-1}.year^{-1}$. Therefore, we used the lower bound of the interval.

Table 73 – Maize production technical coefficients, for each production site and method studied

Sheet	mil8	mil14	mil15	QF	QF-NT	ARG
Production zone	Beira Interior [1]	Ribatejo e Oeste	Beira Litoral	Beira Interior	Beira Interior	Argentina
Fact sheet productivity ($t.ha^{-1}$)	6.5	10.0	10.0	6.5	6.5	8.5
Region average productivity in 2004 ($t.ha^{-1}$)	3.2	10.2	4.3	3.2	-	-
Space occupation ($ha.t^{-1}$)	0.154	0.100	0.100	0.154	0.154	0.117
Market value (€.kg^{-1})	0.14	0.14	0.14	0.13	0.13	0.04
N fertilizer ($kg.t^{-1}$)	10.6	21.9	16.5	37.8	37.8	8.1
P_2O_5 fertilizer ($kg.t^{-1}$)	8.6	10.5	7	26.6	26.6	1.1
K_2O fertilizer ($kg.t^{-1}$)	8.6	10.5	7	40.3	40.3	2.7
Irrigation method [2]	Sprinkling	Furrows, gravity	Sprinkling	Furrows, gravity	Sprinkling	Gravity
Water consumption ($m^3.ha^{-1}$) [3]	7 200	6 480	8 850	9 000	7 200	2 500
Water consumption ($m^3.t^{-1}$)	1 108	648	885	1 385	1 108	294
Number of activity months [4]	8	4	4	8	8	-
Total cost (€.t^{-1})	224.8	157.8	150.5	167.3	-	98.0
Revenue [5] (€.t^{-1})	140.7	140.7	140.7	130	-	42.5
Gross margin (€.t^{-1})	-3.1	54.5	79.8	20.2	-	26.6
Net margin [6] (€.t^{-1})	-84.2	-17.2	-9.9	-37.4	-	-

[1] Production in Beira Interior requires an additional 20 $t.ha^{-1}$ of manure
[2] For more on irrigation methods, and their environmental and energetic evaluation, see, for example, Esteves et al. (1995)
[3] Water needs for national locations were considered accoding to IDRHa (http://www.idrha.min-agricultura.pt/hidrologia/necessidades/inecini.htm, 2007). Irrigation water was obtained by multiplying water needs by a factor of 1.5 for gravity irrigation, or 1.2 for sprinkling
[4] The number of activity months is the number of months during which production operations occur. This period is longer than the plant's growth cycle
[5] Revenue does not account for subsidies
[6] Net margin refers to gross margin minus the fixed costs of machinery and land. For the case of Argentina, no information could be gathered

The work was repeated iteratively. The first runs with SimaPro were used to pinpoint the most striking facts. We found that the impact in the eutrophication theme in Quinta da França is surprisingly high, and in Argentina is lower than expected. The impact was traced back to fertilizers used, and therefore the issue was addressed with additional care.

Fertilization

The values recommended by INIAP (2006) for integrated production practices in a soil with average fertility are shown in Table 74.

Table 74 – Fertilization for integrated production practices in a soil with average fertility. Values were used as the corrected fertilization for Argentina and QF.

Plant	Product	($kg.t^{-1}$)	N	P_2O_5	K_2O
Grain maize	Grain	Fertilizer applied	22	10	11

N – Nitrogen; P_2O_5 – Phosphorus oxide; K_2O – potassium oxide.

Comparing fertilization levels shown in Table 73 for QF and Argentina with recommended fertilization, it is clear that in QF an excess of fertilizers is used, and in Argentina an extremely small part of the required nutrients are being replaced. This indicates that Argentinean soils are more productive, but they are also being drained of their nutrients by unsustainable production rates, and therefore maize production has an extremely high burden on soil production capacity.

Observing Table 73 and Table 74 it may also be noticed that RO uses slightly more fertilizers than needed, while BL uses slightly less. In BI less N-fertilizer is used due to the application of manure. Therefore, since fertilization levels in QF and Argentina are extremely different from those needed, the impact was corrected taking into account a generic fertilizer applied to Argentina in order to return to the soil the nutrients that the production withdraws. It was also considered a reduction in fertilization in QF. In our scenario, only the strictly necessary quantity of each nutrient is replaced with an inorganic fertilizer.

Since, in our case study, the optimization is done from the viewpoint of the farmer in QF, it is plausible that he collects soil samples and obtains an analysis of required fertilization. The same cannot be said about Argentina, since an importer in Portugal has no control on production methods. Still, using these values provides a fairer comparison.

Emissions

Fertilization is responsible for emissions leading to eutrophication (process through which groundwater becomes too rich in nutrients, leading to the overgrowth in algae populations, hence oxygen depletion), acidification (process through which rain's pH decreases) and climate change, as shown for the Netherlands by Pluimers *et al.* (2000). Therefore, it is one of the most damaging steps of production, and so we corrected values used by SimaPro. Nutrient leaching, run-off and emissions were considered to be as shown in Table 75, based on van der Werf *et al.* (2005).

Table 75 – Emission values and uncertainty intervals used

Process	Unit	Value used	Uncertainty interval	Probability distribution used [1]
NO^-_3 leaching	kg NO_3 .ha^{-1}	40	15 - 70	Triangle
NH_3 emitted from ammonium nitrate fertilizer	NH_3 .kg^{-1} N applied	0.02	-	-
N_2O emission due to N fertilizer use and biological N-fixation	kg N_2O.kg^{-1} N applied	0.0125	0.0025 – 0.0225	Triangle
N_2O emission due to atmospheric deposition of NH_3	kg N_2O.kg^{-1} N applied	0.01	0.002 – 0.02	Triangle
N_2O emission due to leaching and run-off of NO_3	kg N_2O.kg^{-1} NO_3	0.025	0.002 – 0.12	Triangle
PO_4 runoff to surface water	kg PO_4.kg^{-1} P applied	0.01	-	-

[1] Refers to this paper. In a triangle distribution the probability of the given value is the highest, and decreases linearly in the uncertainty interval until it reaches zero in the extreme values. For more on probability distributions used by SimaPro, see Grant (2005)

NO^-_3 leaching values vary with crop, production method, soil characteristics and content in nutrients and precipitation, and therefore a local analysis should clearly study these parameters and their relation to leaching. We did not find any data directly referring to the zones studied. Therefore, in order to assess whether the value indicated above is adequate for this paper's conditions, we used two strategies. First, we consulted several Spanish studies. Spain is especially relevant, given its comparability with Portugal in terms of soil and climate characteristics. Then, we calculated the overall nitrogen mass balance.

Regarding Spanish studies, Moreno *et al.* (1996) studied nitrate leaching under irrigation in Spain and reached values of 150 and 43 kg $NO_3^-.ha^{-1}$ leached, corresponding respectively to 500 and 170 kg $N.ha^{-1}$ applied. Villar-Mir *et al.* (2002), for an N application of 250 to 340 kg $N.ha^{-1}$, measured 60 kg $NO_3^-.ha^{-1}$ leached. Diez *et al.* (1997) studied the effect of various fertilization and irrigation choices, emphasising that with convenient irrigation it is possible to diminish leaching in more than 90%[68]. These results confirm that a value of 40 kg $NO_3^-.ha^{-1}$ is plausible as a first approach.

The nitrogen mass balance provided further justification for the use of van der Werf *et al.*'s (2005) values. In steady state, fertilization (Table 74) should be equal to the plant nutrient export plus losses in Table 75. This means that there is neither nutrient mining nor over-fertilization. For example, in QF 6.5 tons of maize are produced. We considered that the maize plant exports from the soil 15 kg of N for each ton produced (Feil *et al.*, 2005). Subtracting this value, as well as quantities in Table 75, from nitrogen fertilization in Table 74, we obtain a value of -1.6 kg $N.ha^{-1}$, about 1% of all nitrogen applied. This means that the law of conservation of mass is verified for nitrogen. For other locations, the balance is also correspondent to less than 10% of nitrogen applied, and so we conclude that values are consistent.

These analyses justify the use in this paper of the generic value indicated above. Still, we carried out an uncertainty analysis, in order to verify whether results would change significantly from the variation of input parameters.

The impact of transportation

This paper considered inland transportation to occur by road, the predominant form in Portugal. Still there are different options regarding the capacity of the truck. The impact of each t.km in Ecoindicators 95 and 99 varies as illustrated by Figure 46. We also considered that transportation occurs in a 16 t truck, since smaller capacities are not used, and larger ones have almost no change in impact.

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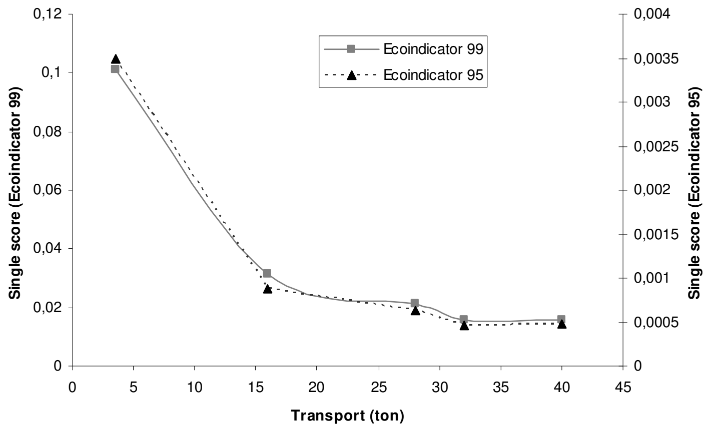
[68] The study was made for several possible fertilizers, and the reduction in leaching occurs for all of them.

Figure 46 – Impact of transportation. The functional unit is 1 t.km of transport (Ecoindicators 95 and 99).



We also considered railway transport as an alternative to road vehicles. According to SimaPro's database, this would decrease the transportation impact of each t.km in 20%.

Transportation from Argentina occurs by sea. The impact of overseas transportation in a transoceanic freight ship is 4.1E-05 $Pt.t^{-1}.km^{-1}$ in Ecoindicator 95 and 1.3E-03 $Pt.t^{-1}.km^{-1}$ in Ecoindicator 99.

Results

The impact of maize production

The main results provided by SimaPro's LCA using Ecoindicators 95 and 99 are shown in Table 76. Only the most significant categories are depicted. Values for QF and Argentina are shown with corrected fertilization, since these are the more realistic. We also included a column for no-tillage production in QF (QF-NT).

Regarding only average national regions, maize grown in RO has the lowest environmental impact. It is also the best in every theme except acidification (where the best is BI). However, maize produced with no-tillage techniques in QF has a lower environmental impact than that produced in all other national regions. No-tillage has an impact about 20% lower than conventional tillage for QF. In all cases, the themes with higher single score impacts are: heavy metals, acidification and eutrophication..

Table 76 also shows that maize produced in Argentina has an impact similar to no-tillage maize in QF for Ecoindicator 95. In Ecoindicator 99 the impact is lower for Argentina. This is coherent with the method of production, since we used an Argentinean no-tillage fact sheet. A higher productivity is not the reason for this result, since 8.5 $t.ha^{-1}$ (in Argentina) is lower than in BI and RO.

More land is used in BI and QF to produce maize, which confirms the results obtained by observing direct occupation of space. RO is the zone where the least area is needed for production (high productivity). However, Argentina has the smallest land use, but not the lowest direct occupation of space, which shows that the latter may not always be a correct indicator of area needed for production.

As for water consumption, in Argentina maize requires less water than anywhere in Portugal. This is mainly due to the climatic difference between both countries. As for national production zones, the most productive ones (BL and RO) have lower consumption per unit produced. However, optimized production in QF-NT, by changing from gravity irrigation to sprinkling, may correspond to a 20% decrease in water use.

All other themes share the same relative impact order with single score results. The only significant difference is that eutrophication in QF is smaller than in RO, which is a consequence of correct fertilization.

Table 76 – Contributions in the most important environmental categories (at the farm gate).

Assessment method	Impact category	Unit	BI	BL	RO	QF	QF - NT	ARG
Ecoindicator 95	Greenhouse	kg CO_2	353	457	343	421	106	423
		% Pt	4.8	6.1	5.4	6.2	1.9	7.2
	Acidification	kg SO_2	4.01	5.26	4.16	2.99	2.69	5.13
		% Pt	25.4	32.4	30.3	20.4	22.5	40.4
	Eutrophication	kg PO_4	1.43	1.38	1.24	1.09	1.04	1.47
		% Pt	13.4	12.6	13.4	10.9	12.8	17.1
	Heavy metals	kg Pb	0.00737	0.00644	0.0056	0.0075	0.00626	0.00377
		% Pt	48.4	41.2	42.2	53.1	54.3	30.7
	Energy resources	MJ LHV	3810	4130	3400	5100	4570	2400
		% Pt	-	-	-	-	-	-
	Total	**Pt**	**1.4**	**1.4**	**1.2**	**1.3**	**1.1**	**1.1**
Ecoindicator 99	Resp. organics	DALY	0.00038	0.00042	0.0004	0.0003	0.00028	0.00034
		% Pt	30.7	31.3	32.5	23.2	22.0	35.2
	Climate change	DALY	8.3E-05	0.00011	8E-05	1E-04	3.2E-05	0.0001
		% Pt	6.7	8.3	7.0	6.6	2.5	10.8
	Acidification/ Eutrophication	PDF.m^2.yr	28.8	38.9	30.3	20.5	18.5	39.2
		% Pt	7.0	8.8	8.0	4.1	4.4	12.3
	Land use	PDF.m^2.yr	38.8	27.2	23	39.1	38	21.8
		% Pt	9.4	6.1	6.0	7.9	9.0	6.9
	Fossil fuels	MJ surplus	394	456	405	571	505	289
		% Pt	29.1	31.6	32.4	35.1	36.5	27.7
	Total	**Pt**	**32.2**	**34.5**	**29.8**	**38.8**	**32.9**	**24.8**
	Water consumption	$m^3.t^{-1}$	1 108	648	885	1 385	1 108	294
	Direct occupation of space	$ha.t^{-1}$	0.154	0.1	0.1	0.154	0.154	0.117
	Revenue	€.t^{-1}	-8.42	-0.99	-1.72	-	-	-

BI – Beira Interior; BL – Beira Litoral; RO – Ribatejo e Oeste; QF – Quinta da França; NT – No-Tillage; ARG – Argentina. Functional unit: 1 t of maize (Ecoindicators 95 and 99).

Operations' impact

As for the impact of specific steps in production, taking RO as an example, and for the impact of specific production operations, fertilization is by far the critical component, accounting for 71% of the total impact.

Fertilization contributes to eutrophication in almost 90% of the total theme's contributions. The impact is mainly due to nitrogen leaching caused by application of fertilizers. 80% of the impact in the theme is the fertilizer itself (application and subsequent loss), and only 20% the fertilizing operation (in terms of machinery use).

In Ecoindicator 99 the most relevant parameter is fossil fuels. The operations responsible for such impact are transportation, machinery use and fertilizer production (56% for QF).

Engström *et al.* (2007) indicate that the most important environmental themes for Swedish agriculture are eutrophication, global warming and resource use. These are also the themes generally referred in the literature. Our analysis confirms that these impacts are important, but indicates some others of interest, like acidification. The importance of heavy metals is striking. SimaPro considers this parameter to be of extreme relevance when it weights its categories to produce a single score. It allocates the impact of building the machinery used to the production in which it intervenes.

Heavy metals are not usually considered an important theme when analysing agricultural life cycles. However, agriculture is an overcapitalized industry. Unlike other types of machinery (industrial, private transportation vehicles), agricultural machinery is used for a relatively small time frame, and only in a very specific time of year. Therefore, costs and inputs of machinery building and use must always be considered, since its impact is comparable to that of maintenance and fuel consumption. Furthermore, in Portugal, recycling or reuse is not necessarily the final destination of materials, and emissions may be aggravated by lack of adequate final destination. For example, in the case of irrigation, machinery needed stands for 46% of its heavy metal emissions, while electricity consumption stands for 38%.

The impact of heavy metals may also be explained by fertilizer use. Fertilizers are currently the main sources of cadmium emissions to the soil, which is an important problem in The Netherlands, where the method was developed. The average European value is 3.8-6.8 $g.ha^{-1}$ in crop land, whereas in The Netherlands values are as high as 7.5-8.5 $g.ha^{-1}$ (Ferrão, 1998). For example, in BI (where the impact on heavy metals is the highest), we found that over 35% of the impact comes from fertilizing. Irrigation also has a very significant part (over 25%).

One simplification we used in this paper was to assume all agricultural machinery similar. In SimaPro's database there is only one choice for each type of machinery. However, depending on weight and power of the machinery, its composition, heavy metal content and energy consumption will differ. In order to confirm if this simplification, as well as all others, could have an influence on final results, we performed an uncertainty analysis.

Uncertainty analysis

Table 77 shows the results of the uncertainty analysis for single score results in Ecoindicator 95. SimaPro uses a Monte Carlo analysis, where random numbers are generated to determine the parameters from the uncertainty domain of each input used in the analysis.

Table 77 – Uncertainty analysis' results, indicated as a percentage of the number of times impact of A > impact of B (Ecoindicator 95).

A \ B	BI	BL	RO	QF	QF - NT	ARG
BI	-	55%	1%	11%	0%	0%
BL	45%	-	0%	17%	0%	0%
RO	99%	100%	-	70%	0%	20%
QF	89%	83%	30%	-	0%	10%
QF – NT	100%	100%	100%	100%	-	82%
ARG	100%	100%	80%	90%	18%	-

BI – Beira Interior; BL – Beira Litoral; RO – Ribatejo e Oeste; QF – Quinta da França; NT – No-Tillage; ARG – Argentina.

It is thus shown that QF-NT is the best option, regardless of parameter variation, except in direct comparison with Argentina 18% of times. These results provide some confidence in the conclusions taken, since they show small variation with changes in parameters. The only exception is BI and BL, since both have very similar results, and it is not clear which one is better.

The uncertainty of the results for each studied location is considerable, as shown for the case of QF in Figure 47. Furthermore, the variation is not symmetric. In fact, the average value obtained with the uncertainty analysis is not the value obtained in calculations with fixed parameters. This is a direct consequence of the fact that the final result is a non-linear function of the uncertain parameters. SimaPro uses error distribution of the lognormal form in almost every entry used, but those values are combined between them and with those introduced. The distribution used for those was triangular. The result is that the final distribution of probability is not itself linear, and therefore the average of the distribution of impacts is not the average impact.

Figure 47 – Overall results for QF, with the error bar indicated (95% confidence interval, standard deviation of 0.395 and standard error of mean of 0.0229).

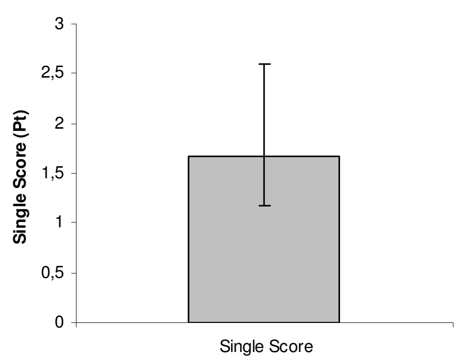
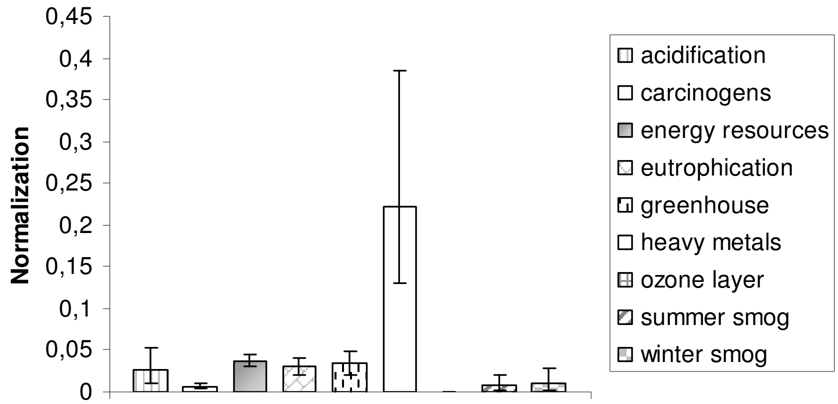


Figure 48 represents single score values and their uncertainty in each theme in Ecoindicator 95 for QF. The most significant themes are those for which we presented the results, and it may be noticed that they are also the most uncertain ones. This analysis was also the criteria for results' precision. Since uncertainty is proportionally high for all themes, and particularly for the aggregated single score as shown in Figure 47, only two significant digits were used.

Figure 48 – Single Score values with correspondent uncertainty in each theme of Ecoindicator 95, for QF maize.



It should be noticed that crop fact sheets do not have any uncertainty data. However, the variation in productivity is of crucial importance in overall impact. It depends on inputs like fertilizers and irrigation, as noted in Megyes *et al.* (2003), Kiniry *et al.* (2004) and Rátonyi *et al.* (2003). But its values may be very different even with the same amount of inputs, depending on regions and meteorological conditions of each year. These determine not only nutrient and water availability for plant uptake, but also its physiological conditions (Bert *et al.*, 2006; Park *et al.*, 2004). For example, Novák and Vidovic (2003) determined a linear relationship between maize nutrient uptake (and therefore plant growth) and the rate of transpiration.

We incorporated the variability in productivity in the results above by observing time series of average productivity values for each region (INE, http://www.ine.pt, 2007). Since the values in the fact sheets are not necessarily the average values, the intervals of variation could not be precisely defined. Still, values of 10% above the normal for a good year and 30% below for a bad one seem plausible from the data observed (maintaining all inputs constant). These were introduced in the analysis with a triangle distribution of probability.

Case study – importation options for Quinta da França

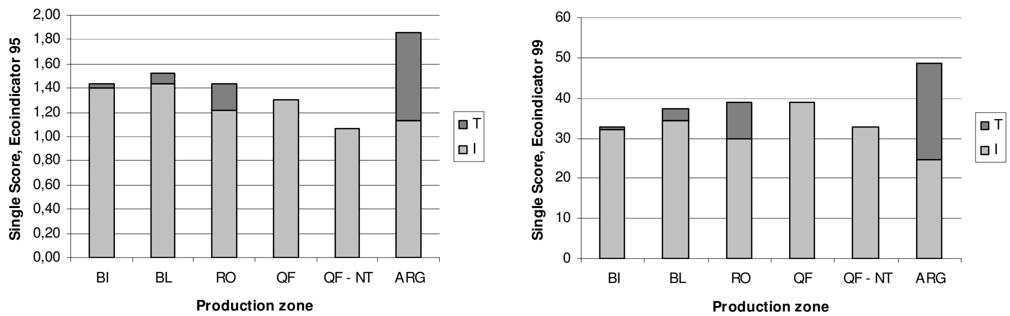
The farmer in QF may choose to produce maize or to import it from elsewhere. We want to study which is his best environmental option. Figure 49 shows that he should produce it himself.

In Ecoindicator 95, QF has the lowest environmental impact even with conventional tillage, but in Ecoindicator 99 production in QF is only the best option if no-tillage is used. Considering the uncertainty level, all national options (using conventional tillage) have very similar results. In both Ecoindicators' analyses maize from Argentina becomes the worst environmental option. Transportation from such a distance makes its net impact clearly the highest.

One explanation for the difference between this evaluation and the one provided by Ecoindicator 95 is that in the latter the impact difference to the other regions at the farm gate was relatively smaller. In Ecoindicator 99, since QF was by far the worst place to produce maize, transportation is not enough to compensate for the difference.

Therefore, the farmer's ideal choice would be to produce maize in QF, where he can control the production conditions, use the best practices (like correct fertilization and no-tillage) and careful irrigation. But even if he does not, the impact of transportation may be enough to justify production in QF.

Figure 49 – Impact of maize production in each region with corrected fertilization and considering transportation to QF by road.



Right: Ecoindicator 95. Left: Ecoindicator 99. Functional unit: 1 t of maize. T – transportation impact; I – ingredient production impact

If transportation by road is substituted by railway, then transportation impacts would decrease significantly. QF with no-tillage would remain the optimal choice, but all other zones would be a better option then conventional tillage in QF, except Argentina. The global impact of importation from Argentina would decrease by 15%, but that is still not enough to compensate the impact of sea transportation.

Conclusions

In this work we studied the impacts of maize production and transportation. Maize is a very important tradable good, since it is highly used for animal and human food, and increasingly used as a biofuel.

We used an LCA tool, SimaPro, to determine the environmental impact of grain maize production in some regions of Portugal and in Argentina. We also studied alternative production methods and the corresponding impacts. We used Quinta da França (QF) as a case study, in order to determine how transportation impacts influence the net result of a management decision to produce maize locally or to import it from elsewhere.

Results obtained show that the distance of transportation is a relevant parameter in the Life Cycle Assessment method, since it may change the results enough to alter the conclusions. For example, the best national zone for production of maize is RO, but since that zone is further distanced from QF, it is environmentally better to produce maize in QF even if production is not optimized. Railway is a valid alternative to road transportation, but the corresponding impact decrease is not enough to change global results.

Importing from a foreign country, Argentina in the case, proved at first to be a valid option, since that is a suitable place for production, even correcting the fertilization practices. But after transportation, Argentina becomes the worst environmental option. Argentina's only advantage is the fact that it requires less water consumption. Water consumption by itself may be an impact whenever it is a scarce resource. It is also a very important theme in agriculture that is not always considered in Life Cycle Assessments, even though irrigation itself may be contemplated via energy consumption.

But QF is not always the best option regardless of its management. Maize production impact is only lower if best environmental techniques are used, such as correct fertilization, correct irrigation and no-tillage. It may be argued that such is also the case in all other places. But, for a given farmer, it is easier to ensure that production is correct if crops are nearby, since the quality of imported crops can seldom be guaranteed. For example, the owner of QF may very easily collect soil samples and find advice on correct fertilization, but it would be impossible for him to individually oblige Argentinean producers to correct their own.

The most important impacts from maize production are greenhouse gas emissions, acidification, eutrophication, energy use and heavy metals. The contribution of heavy metals in Ecoindicator 95 is an unexpected result, since it is not one of the usual relevant themes in agricultural production.

We suggest as further work an economic analysis of alternatives that would widen the scope of the analysis. When optimizing technical solutions and options, like deciding between producing locally or importing from elsewhere, not only direct impacts should be considered but also alternative occupations of space. Because of price regulation, if farmers decide to intensify in one environmentally damaging zone maize production, then former import sources would have to produce less. If the occupation that would substitute maize production there brought a global positive environmental impact, then that would be the better solution. This means opportunity costs must be addressed, as noted by Jungk *et al.* (2002) and Manbiot (2004). However, the LCA method used in this paper assume that choices do not change the whole agricultural sector enough for such effect to be relevant. LCA results have a very low spatial and temporal resolution, and do not regard social and economical aspects, as noted by Owens (1997) and Udo de Haes (2004).

Still, Life Cycle Assessment software such as SimaPro is an important first step in evaluating environmental impacts. We used in this paper a method which included iteratively refined data for the most important aspects of the evaluation. We consider that this method makes SimaPro an easy tool that provides quick but reliable estimations for multiple different indicators, and is an important evaluation option for policy-making.

Appendix IV – Environmental analysis of concentrated feeds

Introduction

Nowadays, a question arises from the global fragmentation of agricultural systems: is it better, in economic and ecological terms, to transfer production to the places where it is most adequate, and then transport it to where it shall be consumed, or, instead, reduce transportation to a minimum by producing nearby where it is mostly consumed? The answer to this question is dominant in any conception of the distribution of space dedicated to agriculture, as well as relevant in the urban/rural segregation, and can be posed in many forms and in respect to all sorts of products. A good example of this is animal feed, since in its composition may be found crop products with different production systems, depending on the place of cultivation, and all those ingredients have to be processed industrially before being given to the animals. Therefore, the optimization of their life cycle has to comprehend not only the production and processing steps, but also the influence of both transportation steps, in order to determine the optimal composition and transportation.

Since the 1950's, consumption of meat products has increased steadily. It is considered that 1 kg of beef requires 7 kg of high-protein feedstuffs (Brown *et al.,* 1999), 1 kg of pork requires 4 kg of grain (CIWF, 1999) and 1 kg of poultry requires 2 kg of feed (CIWF, 1999). Therefore, a higher meat production corresponds to a higher ingredient demand. Today, 95% of the world's soybean production and a third of commercial fish catches are used for animal rather than human feed (Millstone and Lang, 2003). The area needed to produce feeds for the increasing number of animals is, then, high; 75% of all agricultural land in the United States is used to produce ingredients in livestock feed (Millstone and Lang, 2003). This contributes significantly to the impacts of crop production, aggravated by the impact of the animals themselves, namely regarding soil loss and desertification – 85% of all topsoil loss in the United States is attributable to livestock ranching (Millstone and Lang, 2003). Therefore, an increasing number of studies have tried to assess sustainability in agriculture (Lewandowski *et al.*, 1999), particularly researching the environmental impact of feeds (van der Werf *et al.,* 2005; Cederberg and Mattsson, 2000).

In Portugal, the intensification trend as also followed. The animal feed industry is the third most important in the agricultural sector, representing 10.5% of total business volume in 2002 (IACA, 2004). The total production of feeds in the same year has been estimated in almost 3.5 million tons (IACA, 2004). Its relevance derives from the fact that in both intensive (always) and extensive (during the least productive seasons) production systems of meat it is necessary to provide feeding to the animals elsewhere than the pasture. Therefore, it is of great relevance to determine the best composition possible that the feed should have. This work focuses mainly on commercial processed feeds.

Examples of studies on agricultural and meat production are Pereira *et al.* (2004a), Serra *et al.* (1996), Serrano *et al.* (2003) and Simões *et al.* (2003). The determination of the best feed composition, based on a least cost nutritional need, and how it influences the physical quality of the pellet, has already been published (Thomas *et al.,* 2000). Studies have also been made in what respects to feed utilization and its relation to seasonal distribution, as well as improvements in animal production

(Zemmelink *et al.,* 2002; Coleman and Moore, 2003). So far, the consideration of environmental aspects has only been done seldom and as a complement (Castrodeza *et al.,* 2004), and even more rarely using Life Cycle Analysis; to our knowledge, the only examples are pig feed in Bretagne (van der Werf *et al.,* 2005) and dairy cattle feed for milk production (Cederberg and Mattsson, 2000).

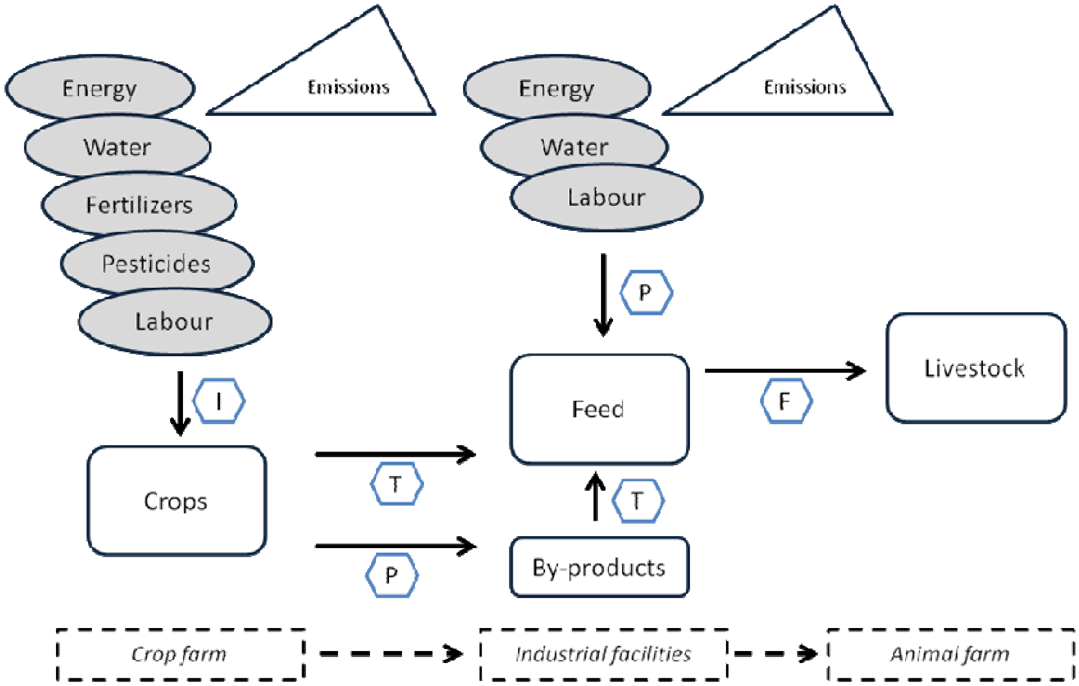
The main goal of this paper is to obtain an optimized feed in the case of beef calves fed in Portugal, regarding its environmental impact. That feed must have the best combination of ingredients (those with the least environmental impact), and the ingredients themselves must be produced in the most appropriate zones through the most appropriate production processes, when possible. That feed also has to fulfil nutritional requisites for healthy animal growth.

Method

Life cycle studied and system boundaries

The typical commercial feed is composed basically of two major groups of ingredients: cereals and oilseeds (protein sources) (IACA, 2004). In both cases, Portugal is a major importer, and so the cost of the raw materials is almost 70% of the total cost of production (IACA, 2004). The life cycle of feed, shown in Figure 25, starts with the production of its constituents, is continued by the transport (national or international) to the factory, where the feed is processed, and from where it is transported again, this time to the farm where it is given to the animals. So, it is important to characterize the very ingredients that compose the feed, because it is during that phase that much of the environmental impact of the final product occurs. That closes the feed's life cycle, since we consider that impacts after the moment the feed is given to the animal are no longer allocated to the animal feed sector, but to the animal production sector.

Figure 50 – Feed's life cycle scheme.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Impact calculation

Therefore, the environmental impact ($\boldsymbol{I_{feed}}$) of the feed consists of the sum of the impact of each of the $\boldsymbol{n}$ ingredients ($\boldsymbol{I_i}$), the impact of transportation of those to the factory ($\boldsymbol{T_i}$), the impact of the production of the feed in the factory ($\boldsymbol{P}$), and, finally, the impact of transportation of the feed to the farm ($\boldsymbol{F}$):

$$I_{feed} = \sum_{1}^{n} I_i + \sum_{1}^{n} T_i + P + F \qquad (1)$$

The impact I_i is not only that of the ingredient production at the farm, but also, in cases where it exists, of its industrial processing prior to that of the feed. This means that there is transportation from crop farms to industrial units considered in impact I_i.

Data gathering

In this paper the ingredients used in the considered feeds were characterized, in terms of its place of origin and ways in which it is produced and quantities used. Then, we determined how transportation involved in the life cycle occurs, and also how the feed is processed. From all the data, different scenarios were considered, in order to determine the optimal configuration of the feed, using the indicators eutrophication, acidification, greenhouse gas emissions, heavy metals, energy and land use.

Alentejo is the region where more reproductive cows are fed, as shown in Table 78 (INE, http://www.ine.pt, 2005). However, only half of the calves are fed in Alentejo (considering each cow has one offspring per year), since they are usually transported elsewhere to be fed for some months before slaughtering. In this paper, we assume an additional optimization that calves are finished in Alentejo.

Table 78 – Number of animals in each region in Portugal

	Beef Calves				Cows	
	Age < 1 year		Age from 1 to 2 years		Others	
Agricultural region	**Heads x 10^4**	**%**	**Heads x 10^4**	**%**	**Heads x 10^4**	**%**
Entre-Douro e Minho (EDM)	9.4	28.4	2.2	31.5	4.1	11.9
Trás-os-Montes (TM)	2.3	7.1	0.1	1.6	2.4	7.0
Beira Litoral (BL)	4.3	12.8	1.0	14.2	1.1	3.4
Beira Interior (BI)	1.3	3.9	0.3	4.3	1.4	4.1
Ribatejo e Oeste (RO)	4.7	14.0	1.7	25.2	2.3	6.7
Alentejo (ALE)	11.0	32.9	1.5	21.7	20.0	65.8
Algarve (ALG)	0.3	0.9	0.1	1.4	0.4	1.2
Total:	**33.3**	**100**	**6.9**	**100**	**31.7**	**100**

As for particular aspects, indicator-based methods used to assess environmental impact at farm level have already been compared (van der Werf and Petit, 2001; Payraudeau and van der Werf, 2004), in order to determine their differences and scope. Virtual water flows (the water needed to produce a commodity which is then sold) in respect to crop trades has also already been analysed (Hoekstra *et al.,* 2004), as well as the best indicators related to land use to provide a good picture of long-term soil fertility and biodiversity (Mattsson *et al.,* 1999). This paper intends to use all of the above in order to produce the most correct evaluation of the life cycle of both feed and feed ingredients.

Feed composition

The European Union regulates which ingredients are safe to use in animal feeds, and that list is eventually incorporated in the Portuguese legislation. A full list of national and Communitarian applicable legislation to the animal feed sector may be found at IACA (2004). Generally, the ingredients can be grouped in the following categories:

- Plant origin products and by-products, as well as transformation products and by-products (cereal grains, oilseeds, legumes, tubers, roots…);
- Animal origin products and by-products (milk, fish, eggs…);
- Mineral origin products;
- Conservants.

The national availability of the most relevant of these ingredients for animal commercial feeds (excluding those given directly to animals) is presented in Table 79 (IACA, 2004).

Table 79 – National availability of feed ingredients

Product	Internal Production (10^3 t)	Gross import (10^3 t)	Gross Availability (10^3 t)
Plant origin products and by-products	934	2 126	3 060
Cereals	845	1 766	2 611
Broken rice	2	-	2
Dry leguminous	1	5	6
Potato	3	2	5
Plant fats and oils	-	20	20
Dehydrated green fodder	-	55	55
Manioc	-	175	175
Other plant origin products and by-products	83	103	186
Transformation products and by-products	925	784	1 709
Grinding and hulling by-products	264	12	276
Malt and beer industry by-products	93	73	166
Distillation products and by-products	32	10	42
Starch extraction industry products and by-products	-	487	487
Of which: Corn gluten feed	-	482	482
Sugar industry products and by-products	63	30	93
Of which: molasses	21	31	52
Oilseeds	401	172	573
Other animal origin products and by-products	72	-	72
Animal origin products and by-products	22	14	36

The table shows that, as a whole, cereals, corn gluten feed and oilseeds comprise 75% of total availability, which makes these components the most relevant to analyse. Corn[69] gluten feed is a by-product of the starch extraction industry obtained by corn wet milling after most starch has been removed. The process starts when shelled corn is cleaned and washed; it is then grinded and screened. The resulting bran, after being dried, is corn gluten feed (Shroeder, 2004). It is composed of a moderately high proportion of protein (20-25%), starch (20%), many digestible fibres and a small amount of oils (Blasi *et al.*, 2001). Portugal mainly imports corn gluten feed from the United States of America (IACA, 2004).

One should, however, notice that the exact ingredient used, in each of the categories shown in Table 79, depends on the species of animal, its current age and phase of development, since its nutritional needs also vary (IACA, 2004). Therefore, the composition of the feed itself is different from animal to animal, as well as the quantity sold and the price.

Nevertheless, there are some common characteristics among feeds. Maize is normally the most used cereal. In general, the quantity of cereals in feeds in 2003 is shown in Table 80 (IACA, 2003).

[69] Since corn gluten feed is fully imported from the United States of America, the American designation of maize is used when referring to this product.

Table 80 – Main cereals' quantity used in feeds.

Product	Quantity (t)
Oats	901
Barley	150 088
Maize	989 935
Sorghum	2 292
Wheat	355 742
Triticale	29
Others	5 586

Scenarios studied

Knowing which ingredients are generally used in feeds, we chose three cases of feeds for the same type of animal and age, in this case finishing calves, to analyse:

4. A grain maize-based feed (Feed 1), which will reflect optimized choices in production zones and methods;
5. A silage maize-based feed (Feed 2), also reflecting optimized choices;
6. The average national feed, which will reflect the current situation in terms of composition and ingredients' origins.

Table 41 gives the composition of each feed. The first type of feed is a conventional maize grain-based feed. It is given to the animal in a quantity of 2.5% of its live weight per day, plus a constant amount of 1.5 kg of wheat straw (Alfredo Sendim; personal communication). The second type of feed is maize silage-based. The animals are fed with 3% of their live weight of this composition per day (Alfredo Sendim; personal communication). Since many feeds are used for each type of animal and age, it is impossible to use a given feed composition as representative (Fernando Anjos; personal communication). Therefore, the average national feed for finishing calves was obtained from the fodder balance (IACA, 2004; INE, http://www.ine.pt, 2005), and so there is a large uncertainty in its composition. This feed should only be considered as a benchmark for the other two.

Table 81 – Base composition of the feeds studied.

Ingredient	% (kg ingredient / 100 kg feed)		
	Feed 1	Feed 2	Average feed
Maize (silage)		58.6	
Maize (grain)	20.0	12.5	13.7
Corn Gluten Feed	20.0	6.6	27.0
Wheat (grain)	19.0	6.6	4.9
Barley	10.0		2.1
Soy meal (44% protein)	6.0	9.1	16.9
Manioc	5.0		
DPG	5.0		
Palm kernel cake	5.0		
Sunflower	4.8		
Carbonate	2.1		
Fats	0.9		
Bicarbonate	0.8		
Salt	0.6		
Premix	0.5	0.8	
Urea	0.3		
Others			36.3[70]
Wheat (straw)	(1.5 kg.day^{-1})[71]	5.8	

[70] The "others" are undisclosed cereals and by-products. Due to lack of information, it was not possible to determine what they are.

The animals are fed from the age of 6 to 8 months (180 to 200 kg live weight) to the age of 12 to 14 months (360 to 400 kg live weight), and considering the average value in each of those, it may be shown that the first type of feed, which provides a fixed quantity of straw and 2.5% of the animal's live weight, is given to the animal in larger amounts in the beginning, and smaller in the end, than the second type of feed, which varies the quantity of straw within the base feed (Alfredo Sendim; personal communication).

Here, we considered two intervals of 2.4 months: (1) from 7.2 to 9.6 months, when the animals are fed in 60% by commercial feed and 40% from pasture; (2) from 9.6 to 12 months, animals are confined to stables and completely fed with the commercial feed. Feed 1 and the average feed allow the animals to grow more rapidly, at the rate of 1.5 $kg.day^{-1}$, whereas Feed 2 provides a slower growth rate of 1.2 $kg.day^{-1}$. Therefore, animals fed with Feed 2 end the second period with less weight than those fed in the other cases (Table 42) (Alfredo Sendim; personal communication).

Table 82 – Animal weights when fed with each feed

Age (months)	Weight 1 and average (kg)	Weight 2 (kg)
7.2	190	190
9.6	298	276.4
12	406	362.8

Ingredient origin

In Portugal, productivity of each cereal depends on geographical location, as shown in Table 83 for 2002 (INE, http://www.ine.pt, 2005).

Table 83 – Cereal production by agricultural zone

Agricultural zones	Area (ha)			Production (t)			Productivity ($t.ha^{-1}$)		
	Wheat	Maize	Barley	Wheat	Maize	Barley	Wheat	Maize	Barley
Entre-Douro e Minho	80	34 645	10	65	129 581	6	0.8	3.7	0.6
Trás-os-Montes	10 493	8 415	586	16 652	14 359	475	1.6	1.7	0.8
Beira Litoral	1 215	30 605	127	2 282	123 575	136	1.9	4	1.1
Beira Interior	1 835	11 366	122	2 012	33 477	131	1.1	2.9	1.1
Ribatejo e Oeste	13 607	33 817	1 538	33 890	333 077	3 028	2.5	9.8	2
Alentejo	200 562	19 428	7 486	354 481	153 356	15 043	1.8	7.9	2
Algarve	2 832	1 239	1 328	3 589	7 191	1 195	1.3	5.8	0.9

The imported cereals come mainly from France (38.4%), Germany (12.6%), Ukraine (12.4%), Argentina (11.1%), Spain (6.8%) and the United Kingdom (6.7%) (INE, http://www.ine.pt, 2005). The most imported cereal is wheat (45% of all cereals), mainly imported from France (40.4% of all wheat), followed by maize (38% of all

[71] Straw is given to the animals in a fixed quantity, which does not depend on the quantity of feed also given. It could not be determined whether the average feed contains straw, but since its fibre content is equal to that of Feed 1 (as shown next) it is plausible to assume that it does not (straw is mainly used for fibre).

cereals), from Argentina (42% of all maize) (FAOSTAT, http://faostat.fao.org, 2005; INE, http://www.ine.pt, 2005; IACA, 2004).

Soybeans and its derivates is the most imported oilseed (78% of all oilseeds), followed by sunflower (10%) and palm kernel cake (9%) (INE, http://www.ine.pt, 2005; IACA, 2004). Soybeans and soybean meal are mainly imported from Brazil (51.1% of all imports) and soy oil from Argentina (81.8%). However, while soy is fully imported, sunflower is, to some extent, produced within the country. Palm kernel cake is mainly imported from Malaysia and Indonesia. Corn gluten feed is fully imported from the United States of America (IACA, 2004).

Imports are mainly transported by road, for intra-UE importation, or by sea (otherwise, mainly America). Transport by railroad or airplane is not significant (INE, http://www.ine.pt, 2005).

Industrial processing

Licencing by the IPPC European directive is mandatory for all facilities dedicated to the processing of animal feeds with a finished product production capacity superior to 300 tons per day. The directive establishes the correct environmental practices that the industry should follow. This regulation also exists in Portuguese legislation, through DL 194/2000. In 2002, there were 22 licensed companies with 27 industrial facilities, 13 of which in Lisbon and Tagus Valley[72], 10 in the centre of the country, 3 in the north and 1 in Alentejo. Total production was 3 478 890 tons, and the business volume was 6.4E08 € (FIPA[73]; APA[74]).

According to FIPA and APA, the most common production scheme starts with the reception, discharge and storing of the raw materials (solid and liquid). Next, these are mixed and crushed, and then, in granulation, the feed is assembled and given its form. It is then cooled and bagged. In all of these operations, the main impacts are the high electrical energy consumption (materials transport indoors), the products resultant from the gas combustion in the boiler (used to form water vapour, which aggregates the ingredients during the granulation stage), other gas emissions (for example, from fossil fuel burning) and dust emissions (from the cereals). Since most materials arrive in bulk, the formation of solid waste from packaging is small. Therefore, most solid waste, as well as wastewater, produced within the facility are of domestic nature (from bathrooms, eventually canteens).

The net emissions considered were based on available environmental licences, regarding two factories: Racentro, in Beira Litoral, and SAPJU, in Alentejo. The data in each case is presented in Table 84.

[72] Lisbon and Tagus Valley corresponds to agricultural region Ribatejo e Oeste.

[73] Federação das Indústrias Portuguesas Agro-Alimentares. http://www.fipa.pt, 2005.

[74] Agência Portuguesa do Ambiente (Portuguese Environmental Agency). http://www.apambiente.pt, 2005.

Table 84 – Animal feeds sector data

		Racentro	SAPJU
Available capacity ($t.day^{-1}$)		500	590
Electric energy ($kWh.yr^{-1}$)		1 802 625	22 700
Propane gas ($t.yr^{-1}$)		148.64	-
Gas emissions (maximum values) ($mg.Nm^{-3}$)	CO	1000	1000
	Organic volatile compounds	50	50
	NO_x	1500	1500
Water spent ($m^3.yr^{-1}$)		-	140
Waste water	pH	6.5 – 8.5	-
	Hydrocarbons ($mg.L^{-1}$)	10	0

Based on these values, an average first estimation of the process was determined, in order to assess the impacts of such methods in the life cycle of the feed. In SAPJU the water effluent has domestic characteristics, as in Racentro, but in the latter the effluent presents additionally a certain level of hydrocarbons, from the maintenance shop. The difference in electrical energy use is striking, but no justification could be found. Therefore, we used the highest value in all calculations, to assure the worst case scenario is considered.

After industrial processing, the feed is transported to regional storehouses or directly to farms themselves, usually by road (INE, http://www.ine.pt, 2005).

Software and inventory used

Life cycle assessment was performed using SimaPro 6.0, developed by the National Reuse of Waste Research Programme and Pré Consultants of the Netherlands, which is widely used in assessing environmental performance in all areas. Two aggregated weighing methods were used: "Ecoindicator 95" and "Ecoindicator 99". Even though conclusions drawn are often similar (Luo *et al.*, 2001), it is important to use both, as they consider different themes and a different conception. Based on studies like Pluimers *et al.* (2000), we considered that the most important themes from Ecoindicator 95 in the case of agriculture were eutrophication, acidification, greenhouse gas emissions, heavy metals and energy use. The themes chosen from Ecoindicator 99 were climate change, acidification/eutrophication and land use. As results will show, these are also the themes to which a highest percentage of global impact is attributable. The aggregated single score values of both Ecoindicators consider not only these themes, but also all those available.

In all crops, nutrient leaching was considered as presented in the last Appendix (for maize) in Table 75, based on van der Werf *et al.* (2005).

NO^-_3 leaching values vary with crop, production method, soil characteristics and content in nutrients and rain occurrence, and therefore a local analysis should clearly study these parameters and their relation to leaching. We did not find any data directly referring to the zones studied. Therefore, in order to assess whether the value indicated above is adequate for this paper's conditions, several Spanish studies relative to maize production were consulted. Spain is especially relevant, given its similarity with Portugal in terms of soil and climate characteristics.

Moreno *et al.* (1996) studied nitrate leaching under irrigation in Spain and reached values of 150 and 43 kg $N.ha^{-1}$ leached, corresponding respectively to 500 and 170 kg $N.ha^{-1}$ applied. Villar-Mir *et al.* (2002), for an N application of 250 to 340 kg $N.ha^{-1}$, measured 60 kg $N.ha^{-1}$ leached. Diez *et al.* (1997) studied the effect of various choices in fertilizer and irrigation, emphasising that with convenient irrigation it is possible to diminish leaching in more than 90% for each fertilizer used. These results confirm that a value of 40 kg $NO_3^-.ha^{-1}$ is plausible as a first approach.

As for wheat, EC (1997) indicates a reference leaching of 35 kg $N.ha^{-1}$ for loamy soils, and 45 kg $N.ha^{-1}$ for sandy soils, under a recommended fertilization level of 180 kg $N.ha^{-1}$.

In this paper, since the main objective is to obtain a comparison between average zones for each ingredient, the generic value was used as indicated above, but an uncertainty analysis was made in order to verify whether results would change significantly from the variation in input parameters.

Water consumption was also analysed, by determining the water used to produce the amount of each cereal that is used in each feed. The direct occupation of space was also calculated, equal to the area needed to produce 1 t of each cereal. The latter is an approximation of Ecoindicator 99's theme land use. However, water use is not usually considered in life cycle analysis, but is of great relevance in agriculture, and so we integrated it in direct occupation of space, and from this in land use. Since land use is measured in a specific Ecoindicator unit ($PDF.m^2.year$) to which no relation exists, water use could not be integrated directly. In order to do so, we used the annual average precipitation in Portugal, estimated as 8890 $m^3.ha^{-1}.yr^{-1}$ (EEA, http://reports.eea.eu.int/92-9167-056-1/en/page003.html, 2005), which allowed us to transform water volume in area and vice-versa. For other countries, we used the world surface average precipitation, estimated as 10 500 $m^3.ha^{-1}.yr^{-1}$ [75]. Then, we converted direct occupation of space to Ecoindicator 99's theme land use, and those to aggregated single score values. This method assumes water use has the same Ecoindicator wheighting as land use. Since crops use both irrigation (except rainfed crops) and rain water, and we only considered in water use the first one, this analysis also allowed us to consider rain water in the global water used by the crop. Such method is, however, a mere estimate, since rain water is not necessarily correspondent, in its temporal and spatial distribution, to the water used for irrigation. Furthermore, infiltration and evaporation would have to be considered in a more accurate estimate, and this approach is not valid when two cultures are sown in the same year.

Data sources

In order to find the environmentally best feed, the first step is to optimize the choices of ingredients. Generically three types of ingredients are the most used - cereals, oilseeds and by-products. Therefore, from these groups, the ones that constitute the feeds were analysed. It is then important to determine the best option in terms of production zones, since they present different methods, inputs and productivities. The data was collected in crop fact sheets provided by the GPP (2001) model for the whole country, and the chosen zones were those with higher productivity and/or production for each ingredient. The sheets analysed and the generic information gathered within them is presented in Table 89. Comparing productivity values therein with the

[75] Value from http://www.physicalgeography.net/fundamentals/8g.html.

averages in Table 83, it may be seen that sheets are not always representative of the corresponding region. In every case of wheat the sheets overestimate the productivity, and therefore results may still be compared, as this paper intends. The same happens for maize, except in Ribatejo e Oeste. Barley's sheets have productivities closer to the average for the regions considered.

The data in the crop fact sheets was the used to determine the production impact of each cereal. All values for maize were used according to Teixeira *et al.* (2007). The environmental impact of each type of barley (common and malts) and wheat products (grain and straw) was allocated according to its economic value.

As for oilseeds, the most used and imported one is soy, and it comes mainly from Argentina as soy oil and Brazil as soy meal. Therefore, it is important to characterize its production in its country of origin. Since no information could be found about Brazil, it was considered that all soybeans are imported from Argentina, which is also a massive exporter of soy, since there it already occupies more land than all other crops taken together (Dros, 2004). There are basically two types of soy produced: type I, which is grown as a monoculture in rotation (bi-annual) with maize, and type II, which is grown together (at the same time) with wheat.

The production of soy in Argentina, commonly in the northeast of the country (Dros, 2004), usually does not include any fertilization, because traditionally it is considered by farmers that the crop's production is not responsive to fertilizers. According to MADRP[76], soybeans also capture some atmospheric nitrogen (Table 85), supplying about 75% of its needs (Table 86).

Table 85 – Quantity of N fixed by soy.

Plant	Fixed N ($kg.ha^{-1}.yr^{-1}$)	
	Common interval	Typical value
Soy (*Glycine soja*)	65 - 179	112

Table 86 – Quantity of N assimilated by soy.

Plant	Product	Production ($t.ha^{-1}$)	Assimilation ($kg.ha^{-1}$)		
			N	P_2O_5	K_2O
Soy (*Glycine soja*)	Grain	2	150	35	60

N – Nitrogen; P_2O_5 – Phosphorus oxide; K_2O – potassium oxide.

Therefore, with the depletion of the soil's nutrient fertility, it should be expected that the situation changes (Gutierrez-Boem *et al.,* 1999). So, we created a scenario in which we considered the level of fertilization needed, according to Galarza *et al.* (2001). Those quantities are shown in Table 87.

[76] Ministério da Agricultura, do Desenvolvimento Rural e das Pescas (Portuguese Ministry of Agriculture), http://www.min-agricultura.pt.

Table 87 – Nutrient intake by soybeans' plant

Nutrient	Quantity (kg.t^{-1} soy)
N	54,0
P	5,4
K	15,7
Ca	2,3
Mg	2,3
S	3,4
B	0,026
Cu	0,012
Fe	0,131
Mn	0,025
Mo	0,005
Zn	0,039

Since Mn, Mo and Zn are not considered in SimaPro's database, and their quantity is very small, these elements were not considered. Soy transportation distances from Argentina to Portugal were taken to be the same as for maize.

It should also be noticed that type II has some supplementation granted by the fertilization of wheat, and therefore the extra nutritional needs are not the same, but, for simplification, they were considered to be so in this paper.

Allocation

The use of by-products, like corn gluten feed, as well as some cereals and oilseeds that have to be processed industrially, raises the allocation problem. Values for mass and price allocation between by-products were considered according to Cederberg and Mattsson (2000), (Table 88). Allocation may as well consider energy, as in Kim and Dale (2002). In this paper price allocation was used, but the effect of such choice was studied by also calculating some results for mass allocation.

Table 88 – Mass and price allocation for ingredients used

Crop	Products	Mass allocation (%)	Price allocation (%)
Soybean	Oil	20	31
	Meal	80	69
Corn for starch production	Starch	63	78
	Corn gluten feed	20	8
	Corn gluten meal	5	10
	German meal	7	4
Palm oil	Crude palm oil	77	83
	Crude palm kernel oil	10	14
	Kernel expel	13	3
Sunflower	Oil	31	63
	Meal	68	37

Table 88 shows that the by-product "kernel expel" is also referred to as "palm kernel cake", used in Feed 1. Following Cederberg and Mattsson (2000), since the market value allocation of this product is only 3%, only the impact of transportation was considered. As the country of origin is Malaysia, transportation by sea and land was considered. DPG, a by-product of barley malt from the beer production process, is fully imported by Portugal from the United States of America, but due to lack of information from the process of barley production in the country of origin, an alternative fully produced in Portugal was considered.

Functional unit

Given the situation described above, the functional unit for the evaluation of each feed had to be relative to the weight variation of the animals, and so all results regarding feeds will be compared with the unit impact.kg^{-1} gained by the animal.

Expected results

The analysis on this paper was structured according to the following steps:

- Determine the impact of each crop's production through life cycle assessment;
- Determine the impact of transportation and the feeds' industrial processing;
- Integrate the latter information to determine each feed's impact;
- Study optimization possibilities for given ingredients;
- Study the relevance of transportation in global impact;
- Verify if impact allocation of by-products influences the final result;
- Study and integrate water use in life cycle analysis results;
- Perform an uncertainty analysis;
- Determine the nutritional composition of the feeds;
- Study if any ingredient with a relevant impact can be substituted.

Table 89 – Production zones and methods analysed

Cereal	Sheet	Production zone	Quantity Produced ($kg.ha^{-1}$) *	Space occupation ($ha.t^{-1}$)	Market value (€.kg^{-1})	Irrigation method**	N fertilizer ($kg.ha^{-1}$)	P_2O_5 fertilizer ($kg.ha^{-1}$)	K_2O fertilizer ($kg.ha^{-1}$)	Water quantity ($m^3.ha^{-1}$)	Water per t ($m^3.t^{-1}$)	Number of activity months	Total cost (€.ha^{-1})	Production value (€.ha^{-1})	Revenue (€.ha^{-1})
Barley ***	cev1	Alentejo	1 400 + 1 500	0.714 + 0.667	0.13 + 0.07	Rainfed	38	24	10	0	0	7	505.9	297.28	-208.62
	cev2	Alentejo	1 800 + 1 800	0.556 + 0.556	0.13 + 0.07	Rainfed	38	24	10	0	0	7	505.9	372.6	-133.30
	cev3	Ribatejo e Oeste	2 500 + 2 000	0.400 + 0.500	0.13 + 0.07	Rainfed	75	72	30	0	0	10	473.09	473.86	0.77
Maize	QF	Beira Interior	6 500	0.154	0.13	Gravity, Pumping	246	172	262	9 000	1 385	8	1 087.76	845.00	-242.76
	mil8	Beira Interior****	6 500	0.154	0.14	Sprinkling	69	56	56	1 200	185	8	1 461.48	914.3	-547.18
	mil14	Ribatejo e Oeste	10 000	0.100	0.14	Furrows, gravity	219	105	105	6 000	600	4	1 578.12	1 406.61	-171.51
	mil15	Beira Litoral	10 000	0.100	0.14	Sprinkling	165	70	70	2 400	240	4	1 505.43	1 406.61	-98.82
	ARG	Argentina	8 540	0.117	0.04	Gravity	69	9	23	2 470	290	-	135.44	362.72	227.28
	Silage	Alentejo	42 000	0.023	0.03	Gravity, Furrows	240	168	168	4000	95	-	35.98	30.00	-5.98
Wheat	tri1	Trás-os-Montes	1 500 + 1 500	0.667 + 0.667	0.13 + 0.07	Rainfed	55	42	42	0	0	12	436.26	306.76	-129.50
	tri5	Ribatejo e Oeste, Alentejo	3 000 + 2 500	0.333 + 0.400	0.13 + 0.07	Rainfed	69	60	25	0	0	7	558.88	568.63	9.75
	tri7	Ribatejo e Oeste	4 500 + 1 500	0.222 + 0.667	0.14 + 0.07	Rainfed	145	108	45	0	0	10	566.11	729.49	163.38
	tri10	Ribatejo e Oeste, Alentejo	5 000 + 3 500	0.200 + 0.286	0.13 + 0.07	Pivot	134	84	35	1 500	175	7	713.09	897.84	184.75
Sunflower	gir1	Beira Interior, Alentejo	2 500	0.400	0.2	Sprinkling	90	63	63	2 400	960	7	1 011.15	505.03	-506.12
	gir2	Ribatejo e Oeste	2 500	0.400	0.21	Rolling	69	63	63	1 500	600	7	711.6	517.5	-194.10
Soybeans	I	Argentina	3 250	0.308	0.09	-	0	0	0	-	-	-	63.24	295.63	232.39
	II	Argentina	2 310	0.433	0.09	-	-	-	-	-	-	-	86.48	143.85	57.36

*** When two values are shown in the same column the first refers to wheat and the second to straw. ** For more on irrigation methods, and its environmental and energetic evaluation, see Esteves et al. (1995) and Vieira et al. (2005). *** First sheet is common barley and others barley malts. **** Maize production in Beira Interior additionally requires 20 000 $kg.ha^{-1}$ of manure.**

Results

Crop production

Maize

Results for maize were considered according to Appendix III – Environmental analysis of maize production. Silage maize is produced in Alentejo (where the feed is given to the animals), while grain maize is transported from Ribatejo e Oeste (the environmentally best option for production).

Wheat

Wheat (*Triticum aestivum*) production results in two separate parts: grain and straw (in variable proportions), both of which are used in animal feeds. It may, then, be shown in Table 90 that irrigated grain wheat has the lower impact in Ecoindicator 95, but rainfed straw in RO and ALE wheat has a lower impact in that case. This happens because the rainfed crop in that produces more grain wheat in proportion to straw, and so a smaller percentage of impact is attributed to straw.

Table 90 – Relative contributions in each environmental category, allocated to each product by economic value (Ecoindicator 95).

Production zone	Single score ($Pt.ha^{-1}$)	Product type	t	$€.t^{-1}$	€	%	Single score ($Pt.ha^{-1}$)	Single score ($Pt.t^{-1}$)
Irrigated Wheat (RO, ALE)	9.7	Grain	5.0	130	195	65	6.3	1.3
		Straw	3.5	70	105	35	3.4	1.0
Rainfed Wheat (RO)	14.4	Grain	4.5	130	390	69	9.9	2.2
		Straw	1.5	70	175	31	4.5	3.0
Rainfed Wheat (RO, ALE)	8.9	Grain	3.0	130	585	85	7.6	2.5
		Straw	2.5	70	105	15	1.3	0.5
Rainfed Wheat (TM)	6.4	Grain	1.5	130	650	73	4.7	3.1
		Straw	1.5	70	245	27	1.7	1.2

RO – Ribatejo e Oeste; ALE – Alentejo; TM – Trás-os-Montes.

Barley

As for wheat, each of the types of barley (*Hordeum vulgare* L.) has two products. Common barley is used directly in animal feeds (principal product), while from barley malts, which is used to produce beer, only the secondary product may be used in animal feeds. The impact again had to be distributed according to economic value, and major results are shown in Table 91.

Table 91 – Relative contributions in each environmental category, allocated to each product by economic value (Ecoindicator 95).

Production zone	Single score ($Pt.ha^{-1}$)	Product type	t	$€.t^{-1}$	€	%	Single score ($Pt.ha^{-1}$)	Single score ($Pt.t^{-1}$)
Common Barley (ALE)	5.0	Principal	1.4	130	182	63	3.2	2.3
		Secondary	1.5	70	105	37	1.9	1.2
Barley Malts (ALE)	5.0	Principal	1.8	130	234	65	3.2	1.8
		Secondary	1.8	70	126	35	1.7	1.0
Barley Malts (RO)	6.4	Principal	2.5	130	325	70	4.5	1.8
		Secondary	2.0	70	140	30	1.9	1.0

ALE – Alentejo; RO – Ribatejo e Oeste.

<u>Sunflower</u>

Sunflower (*Helianthus annum L.*), mainly produced in Portugal, is the second most used oilseed in feeds (10%). The environmentally best option in this case is grown in Beira Interior or Alentejo, with an Ecoindicator 95 single score result of 1.85. The other option studied was Ribatejo e Oeste, but the single score value is 1.95.

<u>National ingredients' best production zones</u>

Performing the same analysis ("Ecoindicator 95") for the other cereals and sunflower, all of which are produced in Portugal, the environmentally best national options were found and are indicated in Table 92. Therefore, not regarding transportation:

Table 92 – Best production zones for each cereal produced in Portugal

Product	Best production zone
Grain maize	Ribatejo e Oeste
Wheat (grain)	Ribatejo e Oeste, Alentejo (irrigated)
Wheat (straw)	Ribatejo e Oeste, Alentejo (rainfed)
Barley	Alentejo
Sunflower	Beira Interior, Alentejo

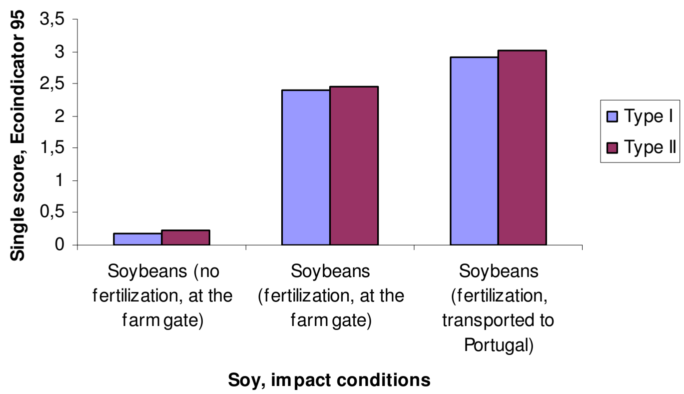
Considering that the factory where the feed shall be produced is located in Lisbon and Tagus Valley (where most are), the impact of transportation from each zone of each ingredient was considered as for maize in Teixeira *et al.* (2007), and average distances between regions were assumed to be 75 km for Ribatejo e Oeste and 150 km for Alentejo. It was then concluded that transportation does not change the results, and the best production zones in Portugal remain the same referred above.

<u>Soy</u>

In the case of soybeans (*Glycine max* L.), produced in and imported from Argentina, the total impact was determined when there is no fertilization and no transportation (growing as is), when there is fertilization and no transportation (growing as it will be) and when both occur. The results in both ecoindicators show that the environmental impact of growing soybeans of both types is, in great part, provided by fertilization. The impact of fertilization is even greater than that of transportation to Portugal. Soybeans of type II are less adequate than type I when compared in the same scenario, but since type II is produced together with wheat it is expected that fertilization will not have to be so high, and so it may be an even better option. Since no information could be gathered about such type of soy, this option was not used in this paper.

Further results may be explained by the fact that, even though they fix a large quantity of N, soybeans still require fertilization. Results for Ecoindicator 95 are shown in Figure 51.

Figure 51 – The impact of cultivating soy in Argentina, with and without fertilization, and with and without considering transportation (Ecoindicator 95)



Corn Gluten Feed

In order to determine the impact of corn gluten feed, data on impacts of the process of U.S.A. corn production and subsequent starch extraction from the SimaPro 6.0 database was directly used, and then allocation according to economic value was done. Ecoindicator 95 single score value, not regarding transportation, is 1.72.

Industrial processing and transportation to animal farm

The impacts of processing the feed (***P***) were based on average maximum production capacity and emission limits, and therefore the worst case scenario was considered, and typical specific emissions were taken to be those of Racentro (Table 84). However, as most facilities are located in Lisbon and Tagus Valley, we considered the processing factory was there. Ecoindicator 95 single score value is 1.8E-02. Since the animal farm was considered to be in Alentejo, additional transportation of the final feed was considered (***F***).

Aggregated feeds

Feed 1

Given the values shown before, we then calculated the impact of each feed. In the case of the grain maize-based feed, overall impact is obtained by using Equation 2. Information for 89.7% of the feed was used.

$$I_{\text{Feed 1}} = \sum_{\substack{\text{i = Grain maize,} \\ \text{Grain wheat,} \\ \text{Barley,} \\ \text{Sunflower,} \\ \text{Soy,} \\ \text{Corn Gluten Feed,} \\ \text{Straw}}} (I_i + T_i) + P_{\text{Feed 1}} + F_{\text{Feed 1}} \tag{2}$$

Considering the total amount of each ingredient used[77], we determined the global impact of production and transportation multiplying each quantity and the unitary impact (Table 93). Finally, the value obtained for the total impact using SimaPro's Ecoindicator 95 is divided by the variation of weight of the animal during the period (Table 96).

Table 93 – Ingredients' impacts over the period of time analysed (I and T) for Feed 1 (Ecoindicator 95).

Ingredient	Quantity (kg ingredient.t^{-1} feed)	Quantity used in 4.8 months (kg)	Unitary productio n impact (Pt.t^{-1})	Impact I (Pt)	Unitary transporaio n impact (Pt.t^{-1})	Impact T (Pt)
Maize (grain)	200	200.03	1.2	0.244	0.066	0.0133
Corn Gluten Feed	200	200.03	0.1	0.028	0.412	0.0825
Wheat (grain)	189	189.03	1.3	0.238	0.066	0.0125
Barley	100	100.01	2.5	0.252	0.133	0.0133
Soy (40% protein)	60	60.01	1.7	0.100	0.514	0.0309
DPG	50	50.01	0.3	0.016	0.022	0.0011
Palm kernel cake	50	50.01	0	0.000	0.514	0.0257
Sunflower	48	48.01	1.1	0.052	0.133	0.0064
Straw	1.5 kg.day^{-1}	7.20	0.3	0.002	0.066	0.0005
Total:	897	904.32		0.932		0.186

Feed 2

In the case of the silage maize-based feed, considering 99% of its composition for which information was obtained, overall impact is determined using Equation 3.

$$I_{\text{Feed 2}} = \sum_{\substack{\text{i = Grain maize,} \\ \text{Silage maize,} \\ \text{Grain wheat,} \\ \text{Soy,} \\ \text{Corn Gluten Feed,} \\ \text{Straw}}} (I_i + T_i) + P_{\text{Feed 2}} + F_{\text{Feed 2}} \tag{3}$$

Silage maize is not processed nor transported, since it is usually grown in the same place where it is consumed. So, in each ton of Feed 2, only the other ingredients are considered in those steps of the life cycle. Once, again the total quantity of each ingredient used was determined (Table 94), and then the aggregated impact (Table 96).

[77] According to the amount per day in "Composition of the feeds".

Table 94 – Ingredients' impacts over the period of time analysed (I and T) for Feed 2 (Ecoindicator 95).

Ingredient	Quantity (kg ingredient.t^{-1} feed)	Quantity used in 4.8 months (kg)	Unitary production impact ($Pt.t^{-1}$)	Impact I (Pt)	Unitary transporaion impact ($Pt.t^{-1}$)	Impact T (Pt)
Maize (silage)	586	635.94	0.5	0.293	0	0
Maize (grain)	125	135.65	1.2	0.165	0.066	0.0090
Soy (44% protein)	91	98.76	1.7	0.164	0.514	0.0508
Wheat (grain)	66	71.62	1.3	0.090	0.066	0.0047
Corn Gluten Feed	66	71.62	0.1	0.010	0.412	0.0295
Wheat (straw)	58	62.94	0.3	0.018	0.066	0.0042
Total:	992	1076.54		0.741		0.098

<u>Feed 3</u>

The average national feed's impact is determined according to Equation 4. Oly 65% is accounted for.

$$I_{\text{Average Feed}} = \sum_{\substack{\text{i = Grain maize,}\\ \text{Grain wheat,}\\ \text{Common barley,}\\ \text{Soy,}\\ \text{Corn Gluten Feed}}} (I_i + T_i) + P_{\text{Average Feed}} + F_{\text{Average Feed}} \qquad (4)$$

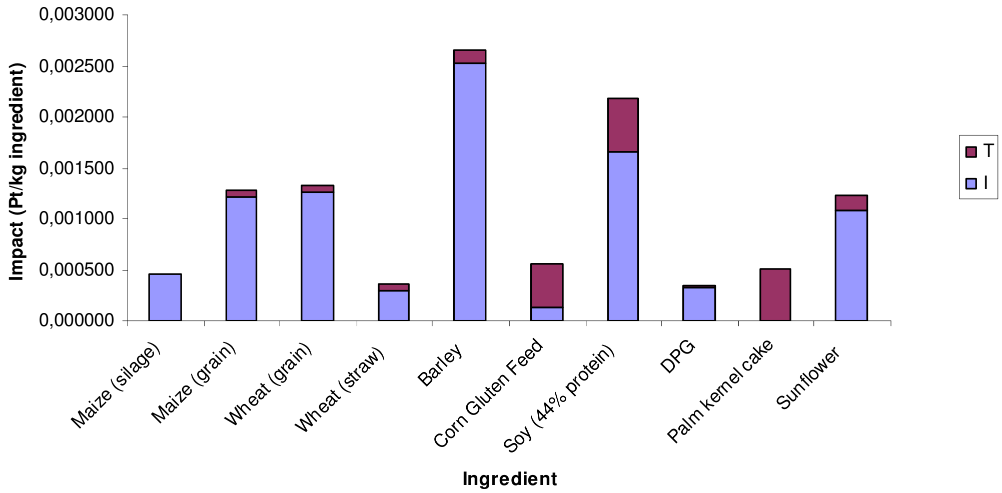
In this case, it was considered that the feed is given to the animals in 2.5% of their weight during the 4.8 months period, in order to calculate the aggregated impact (Table 95 and Table 96).

Table 95 – Ingredients' impacts over the period of time analysed (I and T) for the average national feed (Ecoindicator 95).

Ingredient	Quantity (kg ingredient.t^{-1} feed)	Quantity used in 4.8 months (kg)	Unitary production impact ($Pt.t^{-1}$)	Impact I (Pt)	Unitary transporaion impact ($Pt.t^{-1}$)	Impact T (Pt)
Barley	23.9	21.44	2.5	0.054	0.1326	0.0028
Maize (grain)	158.2	141.90	1.2	0.173	0.0663	0.0094
Wheat (grain)	56.9	51.02	1.3	0.064	0.0663	0.0034
Corn Gluten Feed	467.5	419.42	0.1	0.058	0.4122	0.1729
Soy	293.6	263.35	1.7	0.438	0.5142	0.1354
Total:	1000.0	897.12		0.787		0.324

In order to observe the relation between impact and feed composition, we first determined the impact of each unit (in weight) of the ingredients. Results are presented in Figure 26. Barley and soy are the ingredients with the highest unit impact. Silage maize and wheat straw have the lowest unit impact, along with corn gluten feed, DPG and palm kernel cake, which are by-products.

Figure 52 – Unit impact of each of the ingredients (Ecoindicator 95).



Summary of results

Results in each step of the life cycle are summarized in Table 96. As for Feed 1, the impact obtained in Ecoindicator 95 is mostly due to the production of the ingredients, which is the critical parameter. Transportation has a smaller impact on Feed 2. This happens because silage maize is not incorporated industrially in the feed. Usually, silage is grown close to the animal farm and then given to the animals together with the feed. The average feed is the one where transportation is more important, because the ingredients that have to be transported from a greater distance, namely corn gluten feed and soy, are in a larger proportion than in Feeds 1 and 2. Industrial processing has a very small relevance in all cases.

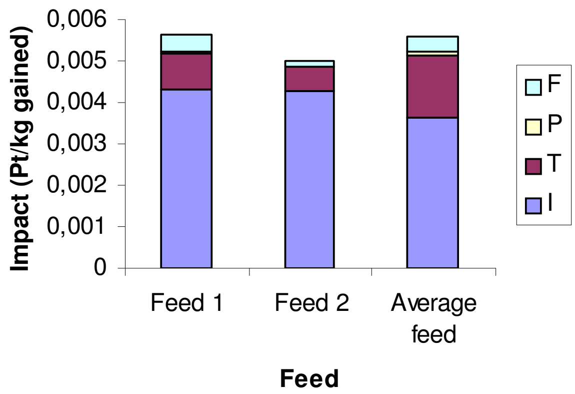
Table 96 – Impact summary for each feed, by step in the life cycle.

	Feed 1		Feed 2		Average Feed	
	Impact (Pt.kg^{-1} gained)	%	Impact (Pt.kg^{-1} gained)	%	Impact (Pt.kg^{-1} gained)	%
I	4.3×10^{-3}	77	4.3×10^{-3}	85	3.6×10^{-3}	65
T	8.6×10^{-4}	15	5.7×10^{-4}	11	1.5×10^{-3}	27
P	7.0×10^{-5}	1	3.0×10^{-5}	1	7.0×10^{-5}	1
F	3.7×10^{-4}	7	1.3×10^{-4}	3	3.7×10^{-4}	7
Total	5.6×10^{-3}	100	5.0×10^{-3}	100	5.6×10^{-3}	100

I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Figure 27 shows that Feed 2 has a 10% lower environmental impact than Feed 1, which is a slightly better option than the national average feed. However, all results concerning the average feed have to be interpreted carefully, since the full composition is not known.

Figure 53 – Overall impact of each feed (Pt.kg^{-1} gained), Ecoindicator 95



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Table 97 shows the impact accountable for each ingredient in the analysed feeds. In Feed 1, maize and barley are the ingredients with the most impact, which is proportionally greater than its weight in the feed. Barley is very demanding, since for input levels equivalent to other cereals its productivity is very small. As for Feed 2, silage maize has the bigger contribution, even though its unit impact is the smallest (Figure 26), since it is used in larger amounts (58.6% of the feed in mass). Maize is responsible for a smaller impact than soy for the reasons stated in the case of the first feed. In the average feed, soy has the bigger contribution, not only because of its unit impact (Figure 26) but also because it is used in large amounts.

Table 97 – Impact of the amount of each ingredient used in each feed (production and transport).

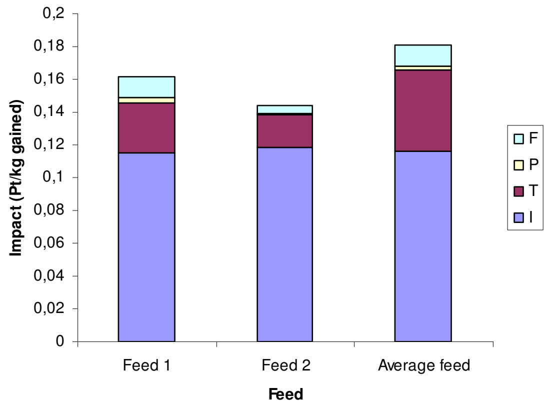
Ingredient	Feed 1		Feed 2		Average feed	
	I + T (Pt.kg^{-1} gained)	%	I + T (Pt.kg^{-1} gained)	%	I + T (Pt.kg^{-1} gained)	%
Maize (silage)	-	-	1.7×10^{-3}	34.9	-	-
Maize (grain)	1.2×10^{-3}	23.0	1.0×10^{-3}	20.8	8.5×10^{-4}	16.4
Wheat (grain)	1.2×10^{-3}	22.4	5.5×10^{-4}	11.3	3.1×10^{-4}	6.1
Wheat (straw)	1.2×10^{-5}	0.2	1.3×10^{-3}	2.7	-	-
Barley	1.2×10^{-3}	23.7	-	-	2.6×10^{-4}	5.1
Corn Gluten Feed	5.1×10^{-4}	9.8	2.3×10^{-4}	4.7	1.1×10^{-3}	20.8
Soy (44% protein)	6.0×10^{-4}	11.7	1.2×10^{-3}	25.6	2.7×10^{-3}	51.6
DPG	8.0×10^{-5}	1.5	-	-	-	-
Palm kernel cake	1.2×10^{-4}	2.3	-	-	-	-
Sunflower	2.7×10^{-4}	5.2	-	-	-	-
Total:	5.2×10^{-3}	100	4.9×10^{-3}	100	5.1×10^{-3}	100

I – impact of the production of ingredients; T – impact of the transportation of ingredients.

Using Ecoindicator 99, results point out Feed 2 as the best option and the average feed as the worst, this time by a large margin (20%), as shown in Figure 54. This indicates that Ecoindicator 99 may be more sensitive to the impact of transportation, since in this case the impact of the ingredients' production (I) is smaller for the average feed, as in

Ecoindicator 95 (although in this case the difference was larger), but in the remaining life cycle the impact is much bigger, since more imported products are used, and therefore that difference is more significant in this case. Still, Feed 2 is better than Feed 1 in the same margin, about 10%.

Figure 54 – Overall impact of each feed ($Pt.kg^{-1}$ gained), Ecoindicator 99

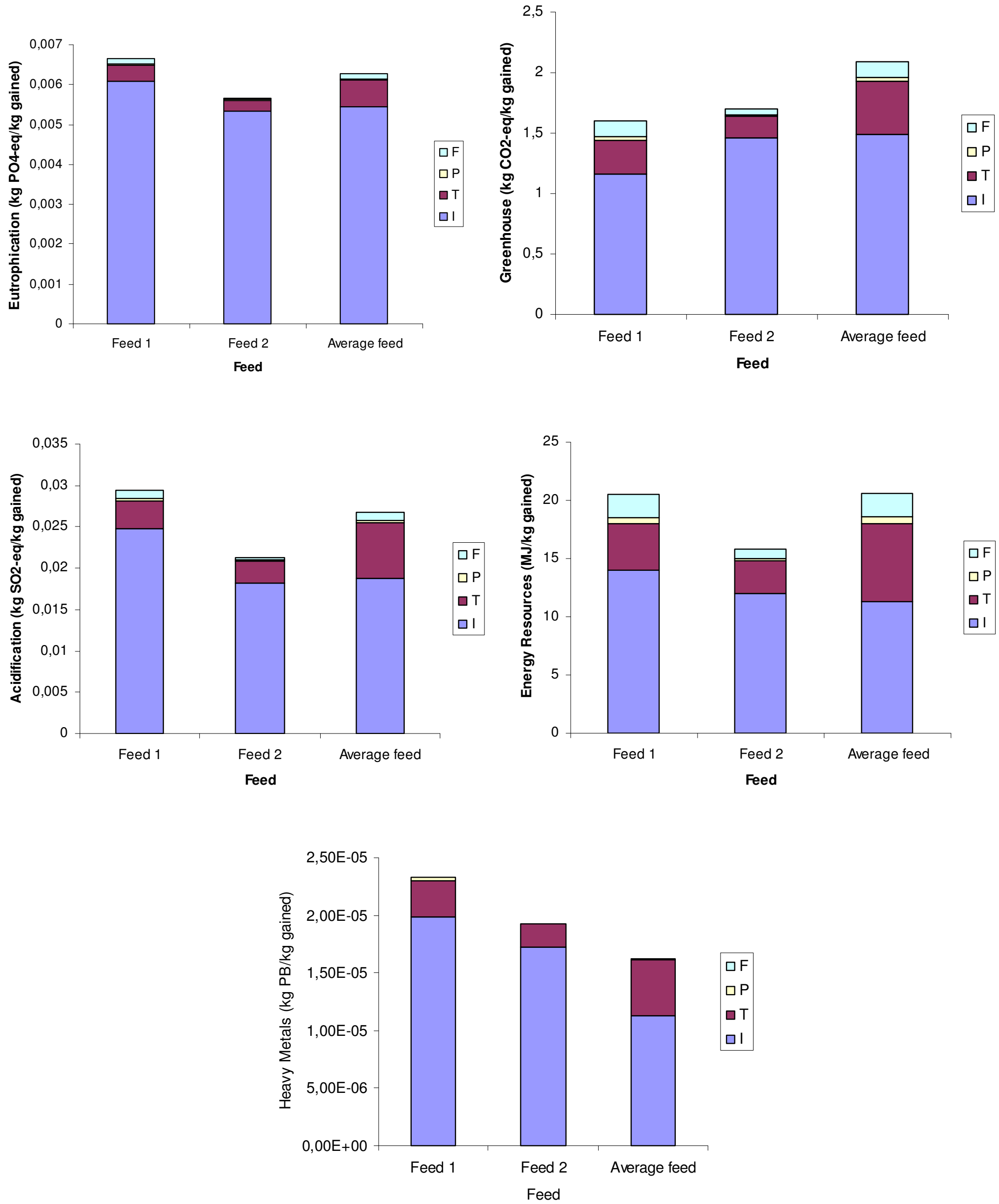


I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Results per environmental theme

As for particular categories of life cycle analysis contributing to Ecoindicator 95, it may be shown that Feed 2 has a lower impact in eutrophication, acidification and energy resources (especially in the latter, since it requires less transportation). In those themes, Feed 1 is always the worst option. However, in greenhouse gas emissions, Feed 1 has the lower impact, and the average feed is the worst. This may be explained by the fact that such feed uses the smaller amount of soy, and that is the ingredient with the biggest contribution. That also explains why greenhouse gas emissions and energy use results are not equivalent. In fact, results are similar except in what respects ingredients' impact (I) for Feed 1. Therefore, a higher energy use implies more emissions, but in the ingredient case there are other sources that change the results. The average feed is better in terms of heavy metals, since impact comes mainly from ingredient production. In this theme, however, Feed 2 is still better than Feed 1. All these results are shown in Figure 55 to Figure 55.

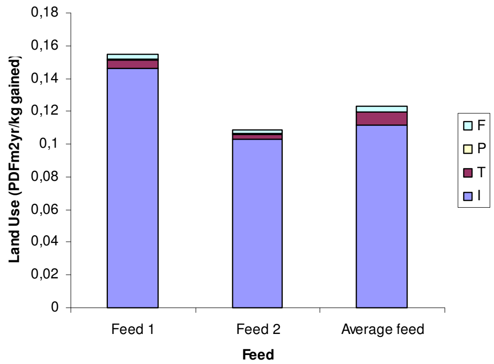
Figure 55, Figure 56, Figure 57, Figure 58 and Figure 59 – Impact per theme of each feed per kg of weight gained by the animal.

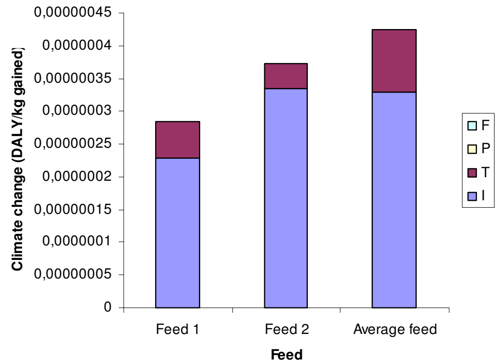


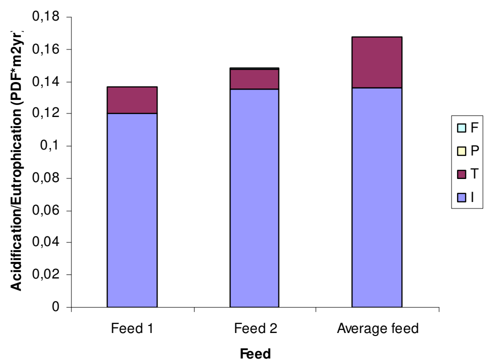
Life cycle assessment themes – eutrophication (top left), greenhouse gases' emissions (top right), acidification (middle left), energy resources spent (middle right) and heavy metals (bottom). I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Themes contributing for Ecoindicator 99 were also analysed, and it was determined that Feed 1 uses more land than Feed 2 (since silage maize requires little space), but contributes less to climate change, acidification and eutrophication. The average feed is in every case the worst option, as shown in Figure 60 to Figure 60.

Figure 60, Figure 61 and Figure 62 – Impact per theme of each feed per kg of weight gained by the animal.





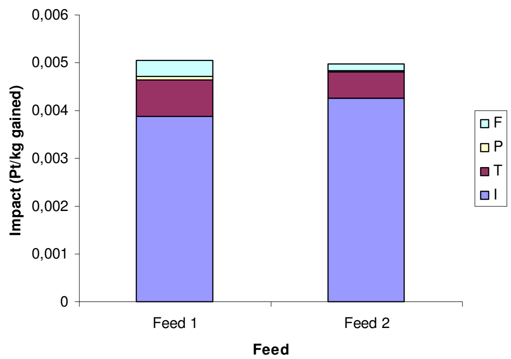


Life cycle assessment themes – land use (on the top left), climate change (on the top right) and acidification/eutrophication (on the bottom).
I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Of the above, land use is of special relevance in agriculture. It may be seen that Feed 1 clearly requires more land, while Feed 2 is the one that requires the less. If we only consider the part due to ingredient production, the average feed has almost the same land use as Feed 2. The results are due to the fact that large productions of silage maize require small amounts of land, even though inputs are small for the productivity, and therefore silage is usually related to extensive systems. This double advantage makes silage maize a very sustainable option.

Therefore, it may be concluded that Feed 2 is a better option for feeds. However, it should be noticed that, in Feed 1, only 90% of the ingredients are considered, and the results are normalized for 100%. Since the other 20% are mainly by-products, Feed 1 may have its impact over-estimated, since it is not to expect that those by-products weigh as much as products to which all of the impact is allocated. If no normalization is considered, and therefore only 90% of the impact is considered, the results are shown in Figure 63.

Figure 63 – Overall impact of Feeds 1 and 2 ($Pt.kg^{-1}$ gained), Ecoindicator 95, only 80% of Feed 1 considered.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

This means that both feeds are probably more equal in impact than it seems if the known composition of Feed 1 is considered the whole composition; however, the fact is that, even considering only 90% of its composition, Feed 1 is already worse than Feed 2. The same process cannot be used for the average feed, since it is not known if the 35% left out are by-products or not.

Uncertainty analysis

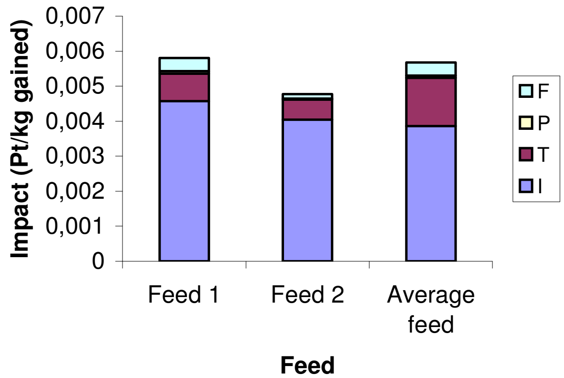
The possibility of silage maize transportation

Silage maize is usually not transported, but these results raise the question of, if transported, how far is still better to use a silage-based feed. Considering that transportation occurs by road, and given the single score differences in Ecoindicator 95, one may note that Feed 2 has the same environmental performance as Feed 1 if silage maize is produced at a distance of 190 km of the animal farm. To be just the same as the average feed, silage may be transported from 200 km. Therefore, even though results from Ecoindicator 95's impact assessment seem very similar, the fact is that the consideration of the transportation of silage maize shows that its use when produced in the same farm where animals grow is the best option. In Ecoindicator 99, for Feed 2 to have the same impact as Feed 1, silage would have to be transported 150 km, and in relation to the average it would still pay off to transport silage from 320 km.

Possibility of silage maize production enhancement

Another important analysis is, since silage maize is usually produced locally or nearby, if the production was controlled so that it could have an environmental performance that, for example, would make nitrate leaching, in time, null, through optimised irrigation practices and direct seeding. In that case, results would be as shown in Figure 64.

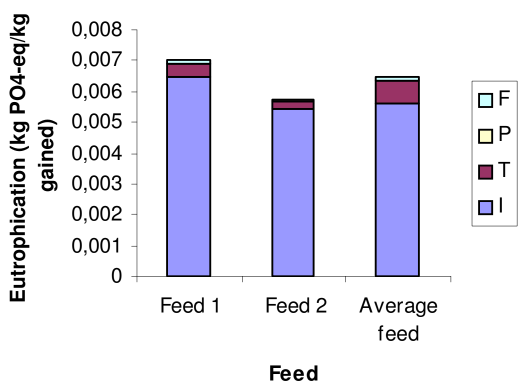
Figure 64 – Overall impact of each feed ($Pt.kg^{-1}$ gained), Ecoindicator 95, no leaching in silage and no-tillage.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

However, in the case of eutrophication, differences would be very significant, as shown in Figure 65.

Figure 65 – Eutrophication contribute of each feed ($Pt.kg^{-1}$ gained), Ecoindicator 95, no leaching in silage.

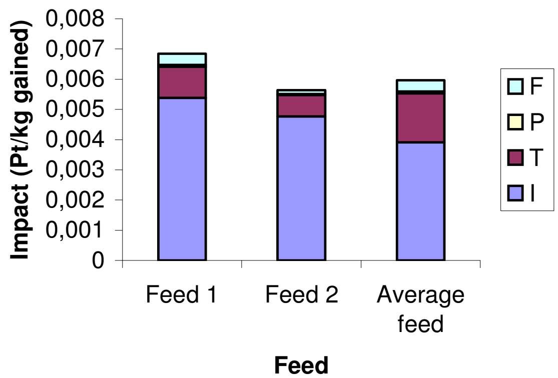


I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

The impact of non-optimized feeds

It is also important to notice that, in this paper, the most appropriate places for each ingredient to be produced (when choice was available) were considered for calculations, but this scenario is very unlikely, since no beef cattle producer can possibly assure that the commercial feeds he buys are composed by the best possible environmental options, and therefore it is not possible to legislate that agriculturalist's activity. The only ingredient that is not processed is silage maize, and therefore that is the only one where some control may exist. Given that, a new analysis was made (with normalized values), this time considering that ingredients are produced in the most common regions, which means that part of grain maize was considered to come from Argentina, and wheat was considered not to be irrigated (since usually is not). However, to guarantee a fair comparison, fertilization corrections in maize and soy were kept. New results are shown in Figure 66.

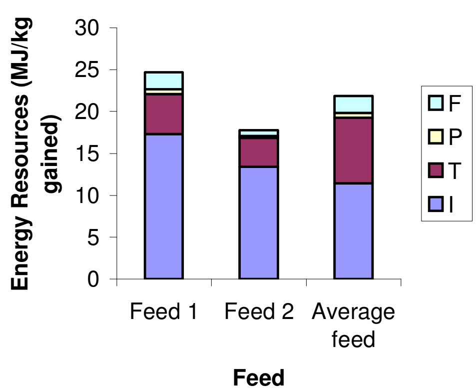
Figure 66 – Overall impact of all feeds (Pt.kg^{-1} gained), Ecoindicator 95.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

These results translate the fact that the impact of the feed with the highest content in maize and wheat (Feed 1) is the most altered by the zone optimization, and therefore it further increases the distance between them. Feed 1 is now almost 20% worst than Feed 2, and more than 10% worst than the average. Feed 2 is the best option, but only by 5% from the average. This confirms that the average feed, since it is mainly composed with by-products, is a good environmental option. Still, since in all the results a market value allocation method was used, this feed is very dependent on prices fluctuations. Furthermore, in particular categories, like those which directly reflect transportation, as energy resources, the average feed is much worse than Feed 2, as shown in Figure 67.

Figure 67 – Overall impact of all feeds in energy resources (MJ.kg^{-1} gained).



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

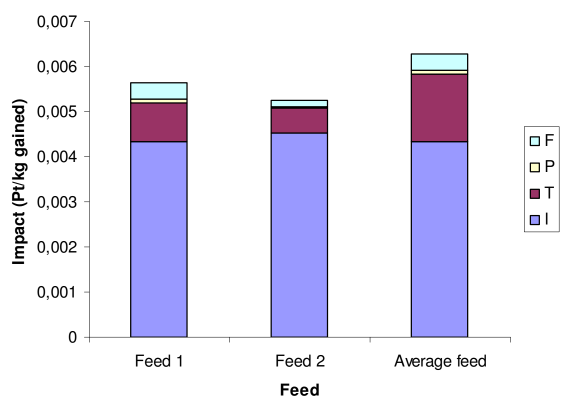
The impact of transportation

Another important analysis is whether the choice of truck for transportation is relevant. If transportation in a 3.5 t van (instead of a 16 t truck) was considered, then the overall impact of Feed 1 rise 11%, for Feed 2 6% and for the average feed 4%, thus aggravating the impact of transportation by far, especially in the case of Feed 1, where all ingredients are transported and most of them by road. Then, it is important that transportation occurs in the largest capacity truck possible.

The impact of allocation choice

Finally, since feeds contain a high percentage of by-products (INE, http://www.ine.pt, 2005), it is important to determine how impact should be allocated. Sometimes, allocation changes results more than any other parameter, as shown by Kim and Dale (2002). In this paper we used economic value, but to determine if that choice influences results an analysis using mass allocation was done, and results are shown for Ecoindicator 95 in Figure 68. The main difference is that mass impact allocation is greater than economic value allocation for soybeans and corn gluten feed (Table 88). Since the feed that used them the most is the average, it increases its overall impact the most. Notice that the impact of ingredients becomes about the same in every case, and therefore Feed 2 has the advantage because it requires less transportation.

Figure 68 - Overall impact of all feeds ($Pt.kg^{-1}$ gained), Ecoindicator 95.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Water use

Since water use is not considered in SimaPro's life cycle analysis, we determined how much water each crop, and therefore each feed, spends. Water uses for industrial processing of both crop and aggregated feeds have also been considered.

Feed 1 and Feed 2 require more total water than the average, as shown in Figure 69. The average feed uses much soybean meal and corn gluten feed, and to each of those only a given percentage may be attributed. Figure 70 shows the unit consumptions of each ingredient.

Figure 69 – Water consumption ($m^3.day^{-1}$) in the composition of each feed.

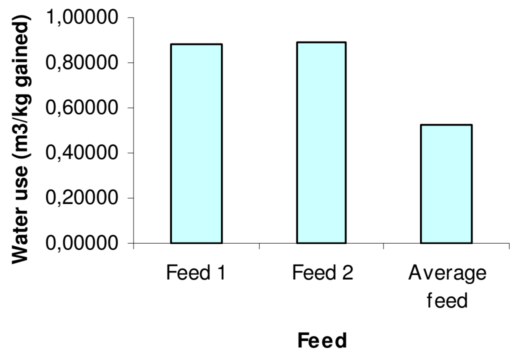
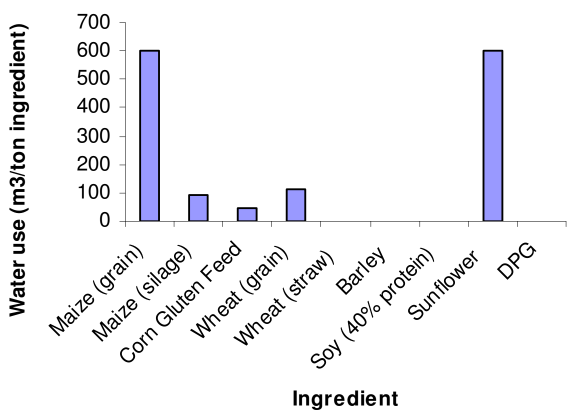


Figure 70 – Water used to produce 1 t of each ingredient.



If irrigated wheat wasn't used, the environmental impact of the feeds would be greater, but less water would be used. In fact, large uses of water when applied incorrectly may be very inefficient, since they favour the drench of soils and consequent tendency for aggravated leaching.

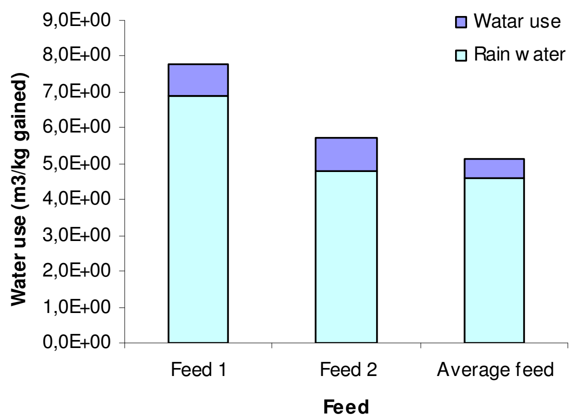
It should be noticed that this analysis is incomplete, since no irrigation data for soybeans was found. If that had been considered, Feed 2 would have a higher water consumption than Feed 1, since it uses more soy. The average feed would surely show a water consumption equivalent to that of the other feeds. Silage irrigation is also uncertain, since the value used in the original grain maize sheet was used. However, it is known that silage maize requires less water and is irrigated fewer times, since it is harvested earlier than grain.

To analyse the importance of both uncertainties, it was considered that soy requires the same level of fertilization that sunflower. Results shown that silage maize would have to be irrigated 50% less than grain for total water consumption to be the same as in Feed 1.

However, according to IDRHa[78], silage maize only requires less 10% irrigation water than grain maize, and therefore Feed 2 surely requires more water than Feed 1.

In order to determine the rain water that the crop receives, we converted land use into water, considering the average precipitation. This does not significantly change results, as shown in Figure 71. Even though Feed 2 requires more water, the difference is not sufficient to compensate the excess of space occupation by the ingredients in Feed 1, since Feed 2 requires less water, which means ingredients receive less rain water.

Figure 71 – Water use due to irrigation and estimated rain water.



To incorporate this global water use in life cycle analysis, we converted irrigation water to direct occupation of space. Then, we converted the new value to land use units, and from there to Ecoindicator 99 single score value. The conversion factors were obtained by dividing the corresponding SimaPro results. Table 98 shows that the aggregated value only changes in about 1%. Feed 2 is still the one with the lower environmental impact, even though more water is spent in producing its ingredients.

Table 98 – Ecoindicator 99 single score value considering water use.

		Feed 1	**Feed 2**	**Average feed**
Water use	$m^3.kg^{-1}$	8.83E-01	8.88E-01	5.14E-01
Occupation of space equivalent to water use	$ha.kg^{-1}$	9.90E-05	1.00E-04	5.60E-05
1 $PDF.m^2.year$	Ha	4.92E-03	4.76E-03	3.95E-03
Land use equivalent to water use	$PDF.m^2.year.kg^{-1}$	2.00E-02	2.09E-02	1.42E-02
1 $PDF.m^2.year$	Pt	7.80E-02	7.80E-02	7.80E-02
Single score equivalent to water use	$Pt.kg^{-1}$	1.56E-03	1.63E-03	1.11E-03
Single Score (without irrigation water)	$Pt.kg^{-1}$	1.61E-01	1.44E-01	1.83E-01
Single Score (with irrigation water)	$Pt.kg^{-1}$	1.63E-01	1.46E-01	1.84E-01

Uncertainty Analysis

Performing an uncertainty analysis for single score results in Ecoindicator 95, results were compared between feeds, and are as shown in Table 77. Results are obtained

[78] http://www.idrha.min-agricultura.pt/hidrologia/necessidades/inec.htm

through a Monte Carlo analysis with SimaPro, where random numbers are generated to determine the parameters in the uncertainty dominium of all values in the database used.

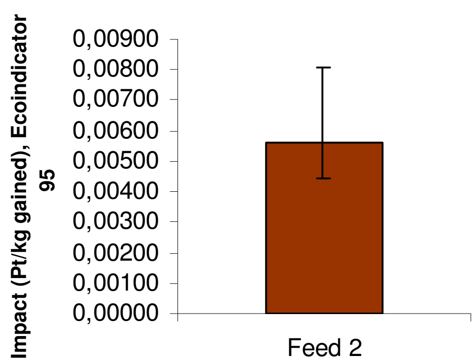
Table 99 – Sensibility analysis' results for all feeds; indicated as a percentage are the number of times impact of A > impact of B (Ecoindicator 95).

B \ A	Feed 1	Feed 2	Average feed
Feed 1		0%	81%
Feed 2	100%		100%
Average feed	19%	0%	

It was then determined that Feed 2 has, in fact, always a lower impact than Feed 1. The same happens in particular categories, except greenhouse (99% less impact for Feed 1) and winter smog (99% less impact for Feed 2). The average feed is relatively worse than Feed 2 in 100% of the cases, and worse than Feed 1 in 78% of the cases.

The variability within each value of feed's impact is illustrated in Figure 47, in the case of Feed 2. It is visible that the error associated is not symmetric in relation to the average, and the average value does not coincide with the value obtained through calculations with the given parameters (Teixeira *et al.,* 2005).

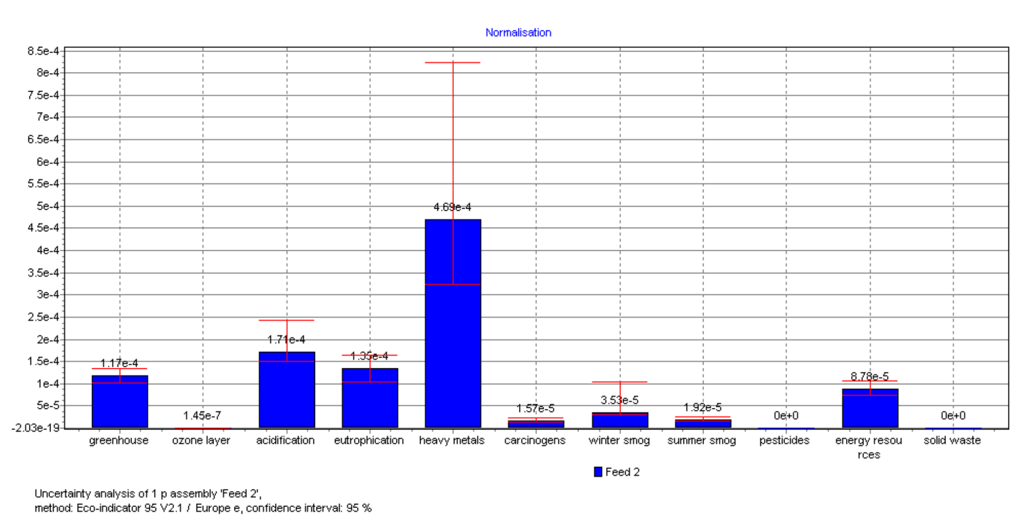
Figure 72 – Overall results, with the error bars indicated.



It should be noticed that crop fact sheets do not have any uncertainty data. Thus, all inputs therein have not been considered to vary. One, however, of particular importance is productivity, which may be very different between regions, in the same zone or even depending on the climatic characteristics of the year. We incorporated that variability in the results above by observing time series of average productivity values for each region (INE, http://www.ine.pt, 2005). Since the values in the fact sheets are not necessarily the average values, the intervals of variation could not be thoroughly defined. Still, values of 10% above the normal for a good year and 30% below for a bad one seem plausible from data observed (maintaining all inputs constant). These were introduced in the analysis with a triangle distribution of probability. Only ingredients produced in Portugal were considered.

In this analysis, heavy metals and winter smog contribute the most to the uncertainty of the final result, as may be seen in Figure 73. As for heavy metals, the biggest contributions to emissions come from fertilizers (uncertainty not considered) and agricultural machinery. Therefore, the most uncertain parameters are the parcel of the field cultivated and the portion of the machine spent in the work hours. The first varies as referred above, with a triangle distribution of probability. For the second one the SimaPro lognormal distribution was used, with a standard deviation square (σ^2) of 1.11. Correspondent emissions of cadmium, lead and zinc also present a lognormal distribution, σ^2 of 1.52, as others like nickel, copper or cadmium, with σ^2 of 5.05. As for winter smog, the largest contributions come from fertilizers and transoceanic transportation. The latter presents a lognormal distribution of probability with σ^2 of 1.5.

Figure 73 – Uncertainty in each theme of Ecoindicator 95 analysis after Single Score normalization, for Feed 2.



This analysis was also the criteria for results' precision. Since uncertainty is proportionally high for all themes, and particularly for the aggregated single score as shown in Figure 47, only two significant algorisms were used.

The choice of functional unit

In order to assess the nutritional equivalence between feeds, the aggregated crude protein content and digestible energy of each were determined. The composition of each ingredient is as shown in Table 100 (Stanton, 2004).

Table 100 – Dry matter, crude protein, digestible energy, net energy for growth and maintenance and crude fibre of each ingredient.

Ingredient	DM (%)	CP (%)	DE (Mcal.kg^{-1} DM)	NE$_m$ (Mcal.kg^{-1} DM)	NE$_g$ (Mcal.kg^{-1} DM)	CF (%)
Maize (silage)	26	8	2.73	1.50	0.88	26
Maize (grain)	92	10	3.92	2.16	1.48	3
Corn Gluten Feed	90	26	3.62	1.94	1.30	9
Wheat (grain)	89	13	3.92	2.16	1.48	3
Barley	89	12	3.66	1.96	1.32	6
Soy (40% protein)	89	50	3.70	2.01	1.34	6
Sunflower	93	50	2.87			
Maize (silage)	26	8	2.73	1.43	0.82	12
Wheat (straw)	88	4	1.94	0.95	0.02	42

DM – Dry Matter; CP - Crude Protein; DE - Digestible Energy; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth; CF - Crude Fibre.

Regarding each feed's composition, their characteristics are shown in Table 101.

Table 101 – Dry matter, crude protein, digestible energy, net energy for growth and maintenance and crude fibre of each feed.

Feed	Mass (kg/4.8 months)	DM (%)	CP (%)	DE ($Mcal.kg^{-1}$ DM)	NE_m ($Mcal.kg^{-1}$ DM)	NE_g ($Mcal.kg^{-1}$ DM)	CF (%)
Feed 1	904.32	84.2	20.4	3.72	2.01	1.35	6.0
Feed 2	1076.54	51.8	17.3	3.30	1.79	1.11	14.9
Average feed	897.12	89.4	27.5	3.73	2.02	1.36	6.4

DM – Dry Matter; CP - Crude Protein; DE - Digestible Energy; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth; CF - Crude Fibre.

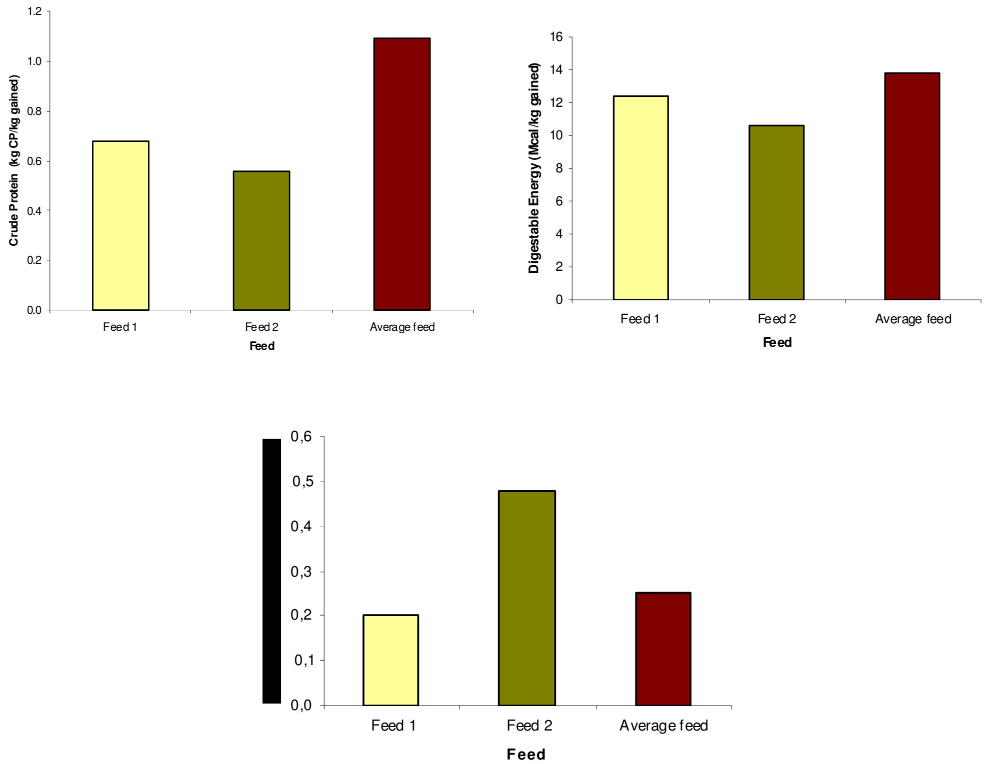
However, as for impacts, values should be compared in relation to the weight gained by the animals during the 4.8 months period. Therefore, values for comparison are shown in Table 102 and illustrated in Figures 74 to Figures 74.

Table 102 – Crude protein, digestible energy, net energy for growth and maintenance and crude fibre of each feed in each kg of weight gained by the animal.

Feed	CP (kg $CP.kg^{-1}$ gained)	DE ($Mcal.kg^{-1}$ gained)	NE_m ($Mcal.kg^{-1}$ gained)	NE_g ($Mcal.kg^{-1}$ gained)	CF (kg $CF.kg^{-1}$ gained)
Feed 1	0.718	13.105	7.103	4.764	0.211
Feed 2	0.559	10.660	5.789	3.597	0.480
Average feed	1.019	13.847	7.507	5.064	0.238

CP - Crude Protein; DE - Digestible Energy; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth; CF - Crude Fibre.

Figures 74, 75 and 76 – Crude protein, digestible energy and crude fibre of each feed.



From here, it may be concluded that Feed 2 has a lower protein value than Feed 1 and even the average, but higher fibre content.

From the impact (I + T) of each ingredient, the impact of each unit of protein (Figure 77), energy (Figure 78) or fibre (Figure 79) may be determined, and therefore assess which ingredients are better used for each purpose. Corn gluten feed has a small impact in all of them. Other ingredients with low impacts are: soy and sunflower for each unit of crude protein; silage maize, straw and sunflower for fibre; straw for digestible energy.

Figure 77 – Impact of each unit of crude protein in each ingredient analysed (Single score, Ecoindicator 95).

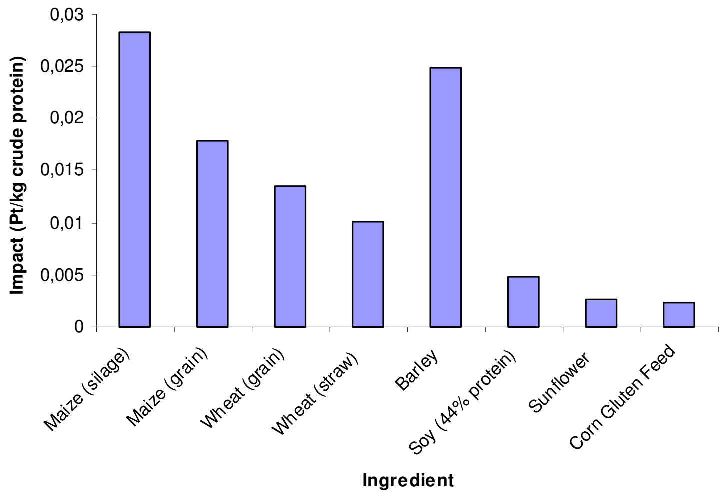


Figure 78 - Impact of each unit of crude fibre in each ingredient analysed (Single score, Ecoindicator 95).

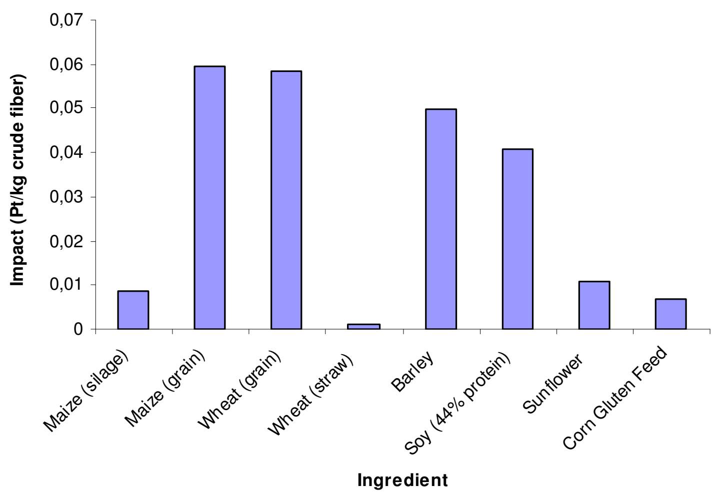
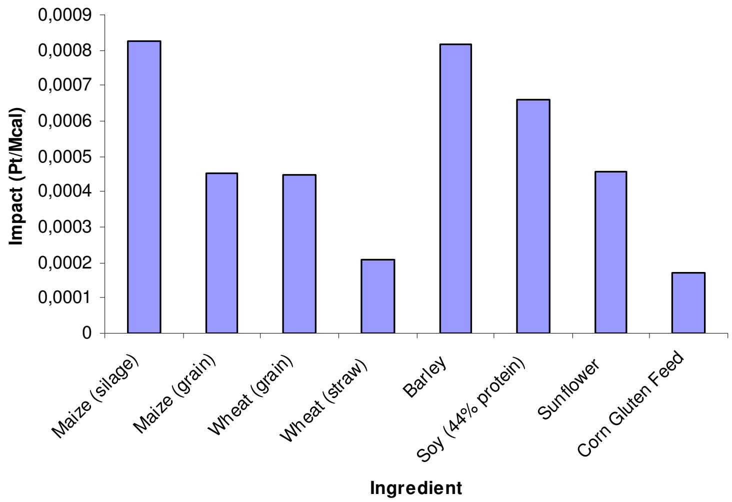


Figure 79 - Impact of each unit of digestible energy in each ingredient analysed (Single score, Ecoindicator 95).



It is also important to determine if the analysed feeds satisfy the animals' needs. In this stage of development, in terms of protein, net energy for grown and maintenance beef calves require the amounts indicated in Table 103 (NRC, 1996). We considered the average weight in the middle of the total period (at 9.6 months), which corresponds to the average quantities in each parameter, as shown in Table 103. Comparing both, Feed 2 has about the necessary quantities of protein and energy, but Feed 1 and the average feed exceed by far that number. For example, the animals use for maintenance 5.55 Mcal.day^{-1}, but both feeds provide about the double.

Table 103 – Feed composition and requirements in protein and energy for maintenance and growth.

Feed	CP (g.day^{-1})	NE$_m$ (Mcal.day^{-1})	NE$_g$ (Mcal.day^{-1})	Average weight (kg)
Feed 1	1076.9	10.66	7.15	298
Feed 2	670.2	6.95	4.32	276.4
Average feed	1529.2	11.26	7.60	298
Requirement for maintenance and 1.2 kg.day^{-1} growth	614	5.2	4.22	275
Requirement for maintenance and 1.5 kg.day^{-1} growth	716	5.55	5.74	300

CP - Crude Protein; NE$_m$ - Net Energy for maintenance; NE$_g$ - Net Energy for growth.

Therefore, in the analyzed categories, by-products seem good options regarding its inclusion in feeds, since the environmental impact attributed to them is smaller, and they still provide the protein, fibre or energy needed for animal growth (as shown by Feed 1 and the average feed). However, they are insufficient in other parameters, as for example in what respects to amino acids (Alfredo Sendim; personal communication). For example, while grain maize has an overall 4.49% content of 12 amino acids (Church and Pond, 1988), corn gluten feed has 0.94% of 4 amino acids (Schroeder, 2004).

Substitution between soy and alfalfa

Since soybeans are one of the ingredients with the highest environmental impact, and furthermore are usually imported from Argentina and often transgenic, their substitution by dehydrated alfalfa was studied. Alfalfa and soybeans have different contents in protein, fibre and energy, but since both play in the feed the role of main protein source, the equivalence was determined regarding that parameter.

Dehydrated alfalfa is obtained from drying of the collected and conditioned fresh alfalfa (*Medicago Sativa* L.), by forced ventilation (Infoagro[79]). The nutritional characteristics of the product remain roughly the same, but the dry matter content is much higher, as shown in Table 104 (Stanton, 2004). In this paper, the impact of fresh alfalfa was determined, and, since at that stage the dry matter content is 26%, it was considered that alfalfa was dried until its dry matter content was 92% (which means that for each ton of dehydrated alfalfa there were 92/26 tons of fresh alfalfa produced). The impact of this final product was used for the comparison.

Table 104 – Nutritional characteristics of alfalfa and soybean meal.

Feedstuff	DM (%)	CP (%)	CF (%)	DE ($Mcal.kg^{-1}$)	NE_m ($Mcal.kg^{-1}$)	NE_g ($Mcal.kg^{-1}$)
Alfalfa dehydrated 17%	92	19	26	0.55	0.28	0.14
Alfalfa fresh	26	19	27	0.54	0.27	0.14
Soybean meal solvent 44% protein	89	50	6	0.76	0.41	0.28

DM – Dry Matter; CP - Crude Protein; CF - Crude Fibre; DE - Digestible Energy; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth.

Table 104 also shows that for the protein content to be about the same in the feed there should be about 2.5 times more alfalfa than there would be soy.

Alfalfa usually is imported from Spain (IACA, 2004), and the region that produces the most quantity is Aragón, from which a transportation distance of 600 km by road was estimated. Therefore, it may be concluded that, using the single score value of Ecoindicator 95 an equivalent amount of each (in protein), alfalfa and soy have about the same impact (both before and after transportation), but in Ecoindicator 99 soybeans have a much larger impact, as shown in Table 105.

Table 105 – Single score (Pt) impact in Ecoindicators 95 and 99 for production and transport of alfalfa and soybeans.

	Dehydrated alfalfa ($Pt.t^{-1}$)	Soybeans ($Pt.t^{-1}$)
Ecoindicator 95	1.7	1.7
Ecoindicator 99	31.3	54.4
Ecoindicator 95 + Transport	2.2	2.2
Ecoindicator 99 + Transport	50.1	72.6

Since alfalfa drying usually involves electricity consumption, which in this case was not considered, its impact is underestimated. Still, these results show that alfalfa is a valid option relatively to soybeans, since its environmental impact is equal or inferior, and it has the additional advantage of being produced in Spain, where correct production techniques and food safety are easier to guarantee. It should also be noted that alfalfa has a much higher crude fibre content than soybeans, and so changing these ingredients has necessarily implications on the remaining content of the feed.

[79] Infoagro, 2005. http://www.infoagro.com.

Another valid option for soy substitution is the use of peas. In this paper such option was not considered, since Portugal mainly imports peas from France and no information could be gathered. Peas capture more nitrogen than soy (requiring lesser fertilization) (MADRP[80]), but pea fields have greater levels of nitrate leaching (van der Werf *et al.,* 2005).

Conclusions

In this paper we analysed different options regarding animal feeds, optimizing the alternatives and verifying whether they would still constitute a nutritionally balanced feed.

From the results obtained it may be concluded that the use of silage maize is an environmental adequate option. Even though feeds consisting of silage maize have to be composed of a high percentage (in mass) of such ingredient, its impact is significantly reduced, especially when compared with the alternative, which is a grain maize-based feed.

The silage-based feed is also environmentally better when compared to the average national feed, and transportation becomes less relevant since silage maize is usually grown nearby or even in the same farm where it is given to animals. In ecological terms, since in this case the unit impact is small, this choice seems plausible. However, even if silage maize is transported, the feed that it constitutes is still better for distances up to 160 to 230 km, depending on the analysis. This means that the use of silage presents large advantages at the farm (less impact in every theme per unit of mass produced), and that clearly overcomes the impact of an eventual transportation.

The most important analysis in which silage maize is not the environmentally better feed is water use. However, to understand if that disadvantage could significantly change the life cycle analysis, we transformed that water in land use. Even though such analysis is only an approximation, it allowed noticing that water use values are important in themselves but, when aggregated, their influence is not enough to change the global impact.

Furthermore, since silage maize is not processed industrially with the remaining feed, its impact is also smaller in this part of the life cycle. However, the assembling of the aggregated feed has a very small overall impact (1%), and production of the ingredients accounts for about 80% of global impact.

But the most important characteristic of silage maize is that the fact that it is produced nearby where it is given to animals, or even in the same farm, makes it easier to control whether its production practices and techniques are those with a least environmental impact. When a farmer buys an average commercial feed, ingredients come from many places and countries (not only these optimised places considered), and therefore the only variable that may be easily controlled through legislation is silage maize.

The fact that the average feed is also a good option is a direct consequence of its large use of by-products. By-products are a valid option for animal feed, since they are usually residues of other industrial processes, and therefore the impact allocation based on

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[80] Ministério da Agricultura, do Desenvolvimento Rural e das Pescas (Portuguese Ministry of Agriculture), http://www.min-agricultura.pt.

market value is always low, even though large quantities of by-product are produced, and therefore a mass allocation would attribute then a larger impact (for example, corn gluten feed - Table 88). However, by-products are deficient in essential amino acids. Furthermore, usually in Portugal by-products are imported, which means that transportation becomes the largest impact in the process, with large contribution to specific themes, like greenhouse of energy use.

Still, the results obtained indicate the strong possibility that the environmentally best feed would be an alternative to all the studied feeds composed by both silage maize and by-products. Such a feed would avoid the direct use of all other crops produced specifically for this end, since those present the highest environmental impact. This option would, however, have to respect all criteria of healthy animal nutrition, and therefore this paper suggests the determination of its exact composition as further work.

Since the goal of this paper is not a single score objective classification of the impact in a given production system or feed, but the comparison between feeds, the uncertainty analysis were very important, since they validated the assumptions and simplifications made. By comparing feeds in similar conditions, the analysis assured that the results obtained always stand.

In what respects to each ingredient, the ones with the highest impact are soybeans and common barley. In the case of soybeans, used as protein sources, a good substitution option must be found. Alfalfa has a slightly higher impact, but is not transgenic, and therefore is a valid ingredient. As for common barley, it may be concluded that producing the cereal for animal feeds has a great impact, but if by-products from other barley products are used the overall impact of the feed may be favoured. Irrigated grain wheat has a low impact, even though in Portugal rainfed cultures are the current trend. Therefore, the trade-off between environmental impacts and water use are of great importance to decide between options. That justifies the method used in this paper for water use internalization on the impact of aggregated feeds. Still, in the case of wheat, we did not consider foreign importation options, because of lack of information. Rainfed production could possibly have a lower impact there, even though transportation would have to be considered. If wheat was imported from France, it would probably be transported by road, and so the global impact could rise considerably.

This paper also did not consider the economic evaluation of the optimization, but such is a very relevant study that may determine which of the trade-offs here suggested pay off. We also did not consider alternative occupations of space, since life cycle assessment methods used in this paper assume that choices do not change the whole agricultural sector enough for such effect to be relevant. LCA results have a very low special and temporal resolution, and do not regard social and economical aspects, as noted by Owens (1997) and Udo de Haes (2004).

Still it is important to notice that because of price regulation, if farmers decide to produce silage maize and use Feed 2, other ingredients this feed uses less or not at all would be produced in smaller amounts. The real environmental impact is dependent on which alternative use of space land formerly used to produce the other cereals and oilseeds used in traditional feeds would have. Furthermore, by-products are often residues from other industries, and if they are not used in feeds they must be given a different end. Real environmental impact comes not only from the direct impact here studied, but also from all other impacts, positive or negative, that choices imply. This means opportunity costs must always be addressed, as noted by Jungk *et al.* (2002) and Manbiot (2004).

Appendix V – Economic balances for steer production in SBPPRL

We show next the calculations for the SBPPRL economic balance[81] in each scenario (with and without support from the PCF, with and without installation support, and for each stocking rate).

Sowing in 2009

Table 106 – Economic balance (values in €.ha^{-1}); steer sold for 250 €, sowing in 2009.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-252	-213	-161	-110	-32	98
	Installation with support	-233	-195	-143	-91	-13	116
With PCF	Installation without support	-235	-196	-144	-92	-14	115
	Installation with support	-216	-177	-125	-73	4	134

Table 107 – Economic balance (values in €.ha^{-1}); steer sold for 375 €, sowing in 2009.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-233	-176	-99	-22	93	285
	Installation with support	-215	-157	-80	-4	112	304
With PCF	Installation without support	-216	-158	-81	-5	111	303
	Installation with support	-197	-139	-63	14	129	321

Table 108 – Economic balance (values in €.ha^{-1}); steer sold for 500 €, sowing in 2009.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-215	-138	-36	65	218	473
	Installation with support	-196	-120	-18	84	237	491
With PCF	Installation without support	-197	-121	-19	83	236	490
	Installation with support	-178	-102	0	102	254	509

Sowing in 2010

Table 109 – Economic balance (values in €.ha^{-1}); steer sold for 250 €, sowing in 2010.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-252	-213	-161	-110	-32	98
	Installation with support	-233	-195	-143	-91	-13	116
With PCF	Installation without support	-240	-201	-149	-98	-20	110
	Installation with support	-221	-183	-131	-79	-1	128

[81] If the value is negative, the costs are higher than the revenue, and if it is possible revenue is higher than costs.

Table 110 – Economic balance (values in €.ha^{-1}); steer sold for 375 €, sowing in 2010.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-233	-176	-99	-22	93	285
	Installation with support	-215	-157	-80	-4	112	304
With PCF	Installation without support	-221	-164	-87	-10	105	297
	Installation with support	-203	-145	-68	8	124	316

Table 111 – Economic balance (values in €.ha^{-1}); steer sold for 500 €, sowing in 2010.

Scenario/Stocking rate (CU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-215	-138	-36	65	218	473
	Installation with support	-196	-120	-18	84	237	491
With PCF	Installation without support	-203	-126	-24	77	230	485
	Installation with support	-184	-108	-6	96	249	503

UNIVERSIDADE TÉCNICA DE LISBOA

INSTITUTO SUPERIOR TÉCNICO

Sustainable Land Uses and Carbon Sequestration: The Case of Sown Biodiverse Permanent Pastures Rich in Legumes

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(Mestre)

Dissertação para obtenção do Grau de Doutor em Engenharia do Ambiente

Orientador: Doutor Tiago Morais Delgado Domingos

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Doutor Carlos Francisco Gonçalves Aguiar

Setembro de 2010

Resumo

A presente dissertação em Engenharia do Ambiente consiste no estudo do sistema de pastagens permanentes semeadas biodiversas ricas em leguminosas (PPSBRL) como opção sustentável para mitigação e adaptação a alterações climáticas. Utilizando várias ferramentas de avaliação de sustentabilidade, determina-se o potencial de sequestro de carbono em PPSBRL e em pastagens naturais, que são a principal alternativa em termos de uso do solo. O aumento do nível de matéria orgânica no solo é a variável chave para o sequestro de carbono e também para os restantes serviços ambientais providenciados por estas pastagens. Por último, apresentam-se as formas de valorizar economicamente os serviços ambientais quantificados anteriormente, nomeadamente através da remuneração do sequestro de carbono pelo Fundo Português de Carbono (FPC), ou da valorização pelos consumidores de carne.

Concluiu-se que as PPSBRL como uso do solo têm vantagens ambientais e económicas generalizadas. Para além de sequestrarem carbono a uma taxa de 5 t $CO_2.ha^{-1}.a^{-1}$, são pelo menos tão positivas para a biodiversidade selvagem como as pastagens naturais, reduzem a erosão do solo e controlam o ciclo da água. Em termos económicos, o apoio do FPC será crucial para aumentar a área deste uso sustentável do solo.

PALAVRAS-CHAVE:

- Avaliação de sustentabilidade;
- Serviços de ecossistema;
- Pastagens;
- Matéria orgânica do solo;
- Carbono;
- Avaliação económica.

Abstract

The present PhD. thesis in Environmental Engineering consisted on the study of the sown biodiverse permanent pastures rich in legumes (SBPPRL) system as a sustainable option for mitigation and adaptation to climate change. Using several sustainability assessment tools, we begin by quantifying carbon sequestration in SBPPRL and the alternative land use of natural grasslands. Increased soil organic matter is the key variable for carbon sequestration and also for other environmental services in these pastures. We end by presenting ways to economically value those environmental services. We namely study the possibility of payments for carbon sequestration by the Portuguese Carbon Fund (PCF), and direct valuation by meat consumers.

We concluded that SBPPRL are a win-win land use, since they have widespread environmental and economic advantages. Besides sequestering atmospheric carbon at an average rate of 5 t $CO_2.ha^{-1}.yr^{-1}$, they are at least as good as natural grasslands for wild biodiversity, reduce soil erosion and control the water cycle. Economically, support from the PCF will be crucial to increase the area of this sustainable land use.

KEYWORDS:

- Sustainability assessment;
- Ecosystem services;
- Pastures;
- Soil organic matter;
- Carbon;
- Economic valuation.

Acknowledgments

The past four years have been an amazing journey. I met so many people that my list of telephone numbers increased exponentially. Since most of these people, directly or indirectly, personal or work wise, helped and influenced me, I believe that it is only fair to refer to them here. So please, bear with the long list that follows for a while.

First and foremost, I would like to thank my advisor, Tiago Domingos. I thank him for complying with the task of directing and correcting my work, patiently and always with a smile. Working with him is a tremendous learning experience. Whenever a difficulty arises, we know we can rely on his judgement – he always finds a way around it. His commitment, optimism and amazing problem-solving abilities are overwhelming. I would have never been able to complete this PhD. thesis under guidance from anybody else.

Sometimes we are lucky to create special bonds with people we spend our whole days with in the workplace. I want to particularly thank, from the people I met at Secção de Ambiente e Energia (SAE, where I have been working for the past four and a half years), my dear friends Ana Simões and Tatiana Valada. During my first years, I learned a lot from working with Ana, who I got used to looking up to. She has the strongest presence I have ever seen in anyone, and she is a reference in all aspects of my daily life. Tatiana became the best friend you can have in the workplace (or anywhere, for that matter). She is so caring, kind and dedicated, always looking to help. Both of them made me enjoy being in IST. They actually made me look forward to Mondays and getting back to work. Their presence lights up the room and makes work fun. In fact, I learned a lot just from who they are. They supported me through the hard times, and laughed with me through the good times. They showed me the best part in being human, and are both lessons in good will for me. I was also fortunate to find friendship and support in Nuno Rodrigues and André Serrenho, two of the greatest and funniest guys I ever knew. I have the luck of working directly with Nuno and his laid-back style has been essential to balance me in my constant stress. And for the existence of this thesis I can thank no one more than Alexandra Marques, who made me overcome my biggest obstacle: starting to write and finding my motivation. Even if we don't work together again soon, I will never forget our winter weekend afternoons working in IST.

I also want to thank several other important people in my life. Clara Fiúza, my best friend for quite some time now, who is always ready to support and guide me when I feel lost, and shares my darkest humor; Lígia Reyes, who has driven me away from work in so many afternoons and nights, who gave new goals and dreams, but always kept me focused on who I am; Maria João Faustino, traveler partner, advisor, counselor and confessor, who, after all the ups and downs that life has put us through, understands me like no one; Paulo Martins, my oldest friend, since we have grown up together and he has been in all the stages and changes that brought me to this day; and finally, my parents, without whose support I would not have even been halfway of where I am today, both academically and as a person. Though I may not always have been the best of friends (or of sons) to you, I deeply thank you. You are all an essential part of me. You provided me with the support and understanding I needed when I was away from work, so that when I was working I could feel balanced and comfortable enough to be productive.

I also wish to thank four special friends I made during my adventure in the Masters course in Economics at Instituto Superior de Economia e Gestão (ISEG). Gui Pedro Mendonça, José Jardim, Nuno Costa and Renata Mesquita helped me to understand Economics. Before our long philosophical and political discussions, Economics was like a foreign language to someone like me, an Environmental Engineer.

I must also name the Masters students in Environmental Engineering of which I was or am the co-advisor: Ana Catarina Henriques, Carlos Lopes, Francisco Amaral, Gonçalo Abrunhosa, Miguel Rodrigues and Natasha Lemos. During the course of their dissertations, I learned probably as much as they did.

I would also like to thank all my colleagues in SAE that I haven't referred before, which was a second home to me in the last years (well, many times it was practically the first): Alexandra Nogal, Ana Gonçalves, Ana Rosa Trancoso, António Lorena, Carlos Teixeira, Cátia Rosas, Cristina Marta-Pedroso, Gonçalo Domingues, Gonçalo Marques, Joana Abreu, João Fernandes, João Rodrigues, Jorge Palma, Josefina Tomé, Marco Reis, Maria Guadalupe Saião, Nuno Sarmento, Oriana Rodrigues, Rui Mota, Sónia Barbosa and Tânia Sousa. I thank them for useful comments and recommendations, but, above all else, I thank them for a fun and healthy working environment.

I would also like to thank the co-authors of the papers resulting from this dissertation, namely João Dias, A.P.S.V. Costa, Ramiro Oliveira, Lídia Farroupas, Fátima Calouro, Ana Barradas, João Paulo Carneiro, Paulo Canaveira, Teresa Avelar, Gottlieb Basch, Carlos Carmona Belo, David Crespo, Vítor Góis Ferreira and Casimiro Martins. I also acknowledge useful comments by Nuno Calado, Pedro Chambel Leitão, Alexandra Lopes and Carlos Aguiar. I particularly thank Helena Martins who revised the final version of the thesis from cover to cover.

I thank most data I used to projects "AGRO 87 – Biodiverse Permanent Pastures Rich in Legumes", developed between 2001 and 2004, and "AGRO 71 – Recovery and Improvement With Pastures of Degraded Soil in Alentejo", from 1997 to 2004.

Part of this work was supported by project Extensity – Environmental and Sustainability Management Systems in Extensive Agriculture, funded by the Life Program of the European Commission (LIFE03 ENV/P/505) and by Project PTDC/AGR – AAM/69637/2006, funded by Fundação para a Ciência e Tecnologia. This work was funded by Fundação para a Ciência e Tecnologia through grant SFRH/BD/25399/2005.

In a final note, I want to acknowledge the groundbreaking role of David Crespo in Portuguese agriculture. His idea inspired this work.

Ricardo Teixeira

August 2009

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Glossary

AFOLU	Agriculture, Forestry and Other Land Uses
ALE	Alentejo
APA	Portuguese Environmental Agency (*Agência Portuguesa do Ambiente*)
BD	Bulk Density
BI	Beira Interior
CDM	Clean Development Mechanism
C	Carbon (atomic)
CO_2	Carbon dioxide (molecule)
CO_2e	Carbon dioxide equivalent (GHG converted to CO_2e using GWP)
DALY	Disability Adjusted Life Years
DDG	Dry distilled grain
DM	Dry Matter
EC	European Commission
EI95	Ecoindicator 95
EI99	Ecoindicator 99
EMAS	Eco-Management and Audit Scheme
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FNG	Fertilized Natural Grasslands
FSC	Forest Stewardship Council
GHG	Greenhouse Gases
GSN	Guaranteed Sustainability Norm
GWP	Global Warming Potential
IACA	Portuguese Association of Producers of Commercial Feeds for Animals (*Associação Portuguesa dos Industriais de Alimentos Compostos Para Animais*)
IC	Impact Category
INE	National Institute of Statistics (*Instituto Nacional de Estatística)*
IOA	Input-Output Analysis
JI	Joint Implementation
KP	Kyoto Protocol
LCA	Life Cycle Assessment
LPN	Liga para a Protecção da Natureza

LU	Livestock Unit
LULUCF	Land Use, Land Use Change and Forestry
MA	Millennium Ecosystem Assessment
NEP	Net Ecosystem Production
MBD	Mineral Bulk Density
NG	Natural Grasslands
NPP	Net Primary Production
NPV	Net Present Value
NV	Normalization Value
ptMA	Portuguese Millennium Ecosystem Assessment
PCF	Portuguese Carbon Fund
PNAC	National Programme for Climate Change (*Programa Nacional para as Alterações Climáticas)*
PPP	Purchasing Power Parity
QF	Quinta da França
RC	Replacement Cost
RO	Ribatejo e Oeste
SBPPRL	Sown Biodiverse Permanent Pastures Rich in Legumes
SIP	Sown Irrigated biodiverse permanent Pastures
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
USLE	Universal Soil Loss Equation
VSL	Value of a Statistical Life
VSLY	Value of a Statistical Life Year
WF	Weighing Factor
WTP	Willingness To Pay
YLD	Years Lived Disabled
YLL	Years of Life Lost

List of units

Symbol	Designation	Definition
Prefixes, suffixes and symbols		
c	*centi*	$= 10^{-2}$
%	*percentage*	$= 10^{-2}$
k	*kilo*	$= 10^{3}$
M	*mega*	$= 10^{6}$
ppm	*parts per million*	$= 10^{-6}$
< x >	*quantity of x per unit of mass*	(used in Chapter 3)
{ x }	*quantity of x per unit of area*	(used in Chapter 3)
Mass units		
g	*gram*	
t	*ton*[1]	1 t = 1 Mg
Time units		
s	*second*	
day	*day*	1 day = 86 400 s
yr	*year*	1 yr = 31 536 ks
Length units		
m	*meter*	
Energy units		
J	*Joule*	
cal	*Calorie*	1 cal = 4.184 J
Other units		
Pt	*Ecoindicator points*	

[1] The unit "ton" can also be written as "tonne". We adopted the first designation in this thesis.

Foreword

In the fall of 2005, I became a member of the Project Extensity team. There was a presentation for the European Commission which took place in Herdade dos Esquerdos, Vaiamonte, where the system of Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL) was developed. During that day, we were guided in the farm by Eng. David Crespo. He was the person responsible for the development of the system.

The most striking fact about this pasture system that I knew nothing about at the time was that the fields were still green, even though it was already Fall. Non-SBPPRL pastures from neighbours were already showing signs of decreased production. How could it be that two fields side by side could exhibit such different endurance?

At one point, Eng. David Crespo stated the following:

“In this land, due to SBPPRL, I have been responsible for the sequestration of enough carbon to compensate my emissions, my children’s emissions, and my grandchildren’s emissions too.”

This thesis is aimed at proving him right or wrong.

1. Introduction

This thesis' main focus is to determine the carbon sequestration potential of one particular pasture system, that of Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL), and its other environmental co-benefits. We study whether SBPPRL are a possible response to the decline in ecosystem services provided by pastures in Portugal.

The present Introductory Chapter is used to define the scope and objective of this thesis. We begin by stating the Portuguese obligation in Kyoto Protocol and by justifying the relevance of studying carbon sequestration in pastures. Since carbon sequestration is an ecosystem service, we state the relation between ecosystem services and agricultural sustainability, in order to show the importance of pastures in the subject. We define the system of SBPPRL in terms of its composition and a priori *effects. Since livestock is one of the key parameters in pasture systems, we then turn to the definition of scenarios for livestock production that we will use later on in this thesis. We end this Chapter by presenting an overlook of the next Chapters of the thesis.*

1.1 The Kyoto Protocol and policy instruments

1.1.1 Setting international standards

The urgent need to respond to the alleged threats of climate change led to the United Nations Framework Convention on Climate Change (UNFCCC, 1998). The framework of this Convention paved the way for the appearance of a mandatory agreement between most countries, stipulating for each a maximum emissions scenario (Harvey, 2004). This agreement was named after the Japanese city in which it was signed, and became known as the Kyoto Protocol (KP). The KP set quantitative targets for each country or assembly of countries, in order to reach a worldwide reduction in net GHG emissions of 5% by 2010, in relation to 1990. The finishing level is obtained by averaging emissions in the years between 2008 and 2012. If any country cannot keep its designated upper bound, there are four mechanisms that it can recur to:

- The carbon-trading scheme, in which over-compliers sell their exceeding credits to under-compliers using a market price;
- Clean Development Mechanisms (CDM), in which developed signatory countries execute projects leading to a GHG emissions reduction in developing countries, and the credits revert to their favor;
- Joint Implementation (JI), in which developed signatory countries execute projects in other signatory countries, with the same effect as CDM;
- Investments in funds managed by independent third parties, or other alternative instruments.

Unlike many important polluters, like most notably the United States of America, Portugal signed the KP. We now turn to the national consequences of the agreement.

1.1.2 The Portuguese target

In the KP, Portugal agreed not to increase its GHG emissions, in relation to 1990, by more than 27%. According to the Portuguese Environmental Agency (APA, 2006a), Portugal is one of the European Union (EU) countries with the lowest *per capita* GHG emissions. However, in the period from 1990 to 2003, its emissions increased 37%, an increase over the KP limit (APA, 2006b). The Portuguese deficit will likely be about 3.73 Mt $CO_2e.yr^{-1}$.[2]

Faced with the risk of under-compliance of the KP, Portugal created an instrument of analysis, which was the National Programme for Climate Change (PNAC, 2006). The function of PNAC is to assess the current situation in terms of the compliance of the KP, build scenarios for future trends, and try to determine additional measures necessary to meet the national goal.

PNAC defined additional measures in almost all economic sectors, and gave birth to an emissions limit for polluting industries within the country (APA, 2006a). Amongst those additional measures, Portugal decided to elect some optional mechanisms, which countries are not required to account for in their inventories. The next section is about such mechanisms.

1.1.3 Optional mechanisms of the Kyoto Protocol

There are strict stipulations in the KP as to how a country's emissions inventory is made, namely regarding what to account. However, there are some items that remain as an option for each signatory country. These options relate to the agro-forestry sector, and are the so-called Land Use, Land Use Change and Forestry (LULUCF) activities, now renamed Agriculture, Forestry and Other Land Uses (AFOLU), under the framework of Article 3.4 of the KP. While most sectors are net polluters, where all that can be done is to minimize CO_2 emissions, and energy biocrops and many sub-products may also be used to produce renewable energy which substitutes fossil fuels, AFOLU acticities are responsible for CO_2 sequestration in soils and living biomass. Therefore, AFOLU activities (IPCC, 2006) do not promote a decrease in emissions, but rather the sequestration of CO_2 (which is not permanent[3]).

Portugal plays a leading role in its account in the KP, since it decided to elect, in the framework of these voluntary AFOLU activities under Article 3.4 of the KP, the activities: "Grassland Management", "Cropland Management" and "Forest Management". The rationale for this choice will be addressed latter.

However, even using such additional measures as these, PNAC (2006) still pointed to an excess in emissions. This requires Portugal to search for new possibilities to compensate the high emissions. To such effect, a Carbon Fund was established by the Portuguese Government in 2006.

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[2] The unit "CO_2e" refers to carbon dioxide equivalent, which is obtained by converting all greenhouse gases into the CO_2 equivalent using their specific global warming potential. This is shown in Section 3.2.

[3] The KP does not regard the issue of permanence, and therefore this mechanism, in the present period, is equivalent to emissions reduction.

1.1.4 The Portuguese Carbon Fund

The Portuguese Carbon Fund (PCF) is an operational instrument which intends to finance several actions with positive returns regarding a decrease in GHG emissions. These actions must be additional to those considered by PNAC, since they mean to fill the current gap of emissions. The fund was started in 2006, with an initial sum of 6 000 000 €.

The fund may be used to acquire credits using one of the four resorts considered in the KP, or, alternatively, it may be used to finance national projects. Even though the political priorities are yet to be defined, it is possible to assume that national projects are the most interesting option. First, the use of any KP scheme would mean that Portugal would be indirectly investing in forestation or energy efficient projects elsewhere in the world. Second, while CDM and JI have many practical implementation difficulties, carbon-trading is subjected to market uncertainties and price fluctuations that make it unreliable as a long-term policy. The emissions trading scheme was, however, designed to guarantee that reductions occur where it is cheaper to generate credits (Wagner and Wegmayr, 2006). The international price thus sets a standard for the price of national projects.

National projects may respect to permanent emissions reduction, or to carbon sequestration. If national projects were used, then the investments would occur in national territory, and since projects may be screened by the PCF itself (instead of indirectly acquired as a cabon credit), there is a possibility of strong complementarities between GHG reduction and other environmental and policy objectives.

1.1.5 Is agro-forestry important in the global carbon balance?

The rise in GHG concentrations in the atmosphere is mostly due to fossil fuel consumption. However, the agricultural and forestry sector has an important role to play in the global carbon cycle. The classical examples are intensive agriculture (due to fertilizer use, for example) and deforestation. But a significant fraction of the responsibility for emissions in this sector is due to livestock and land use changes.

Greenhouse gases and livestock production are deeply related, as shown by the Food and Agriculture Organization of the United Nations (FAO) (Steinfeld *et al.*, 2006). In a recent report, FAO states that the world's livestock sector is responsible for 18% of greenhouse gases emissions (measured in CO_2 equivalent), which is a higher share than transport. Reduction strategies have targeted intensive systems, focusing on nitrogen runoff and emissions related to fuel and fertilizer inputs, whereas efforts to lower methane emissions have focused on the more extensive systems, where lower productivity implies higher methane emissions per unit of product (Subak, 1999). And it is not just livestock that is a problem. The whole chain of meat production has high GHG emissions. Tukker *et al*. (2006) conclude that food and beverages are one of the three components of European Union consumption with the largest environmental impacts, with meat consumption being the most important item within this group. This fact gave rise to a specific study just for the meat sector (Weidema *et al.*, 2008). Emissions from livestock are deeply related to meat consumption (Teixeira and Dias, 2008).

It is also believed that land use and land management practices are responsible for 12 to 42% of total GHG emissions (Watson *et al.*, 2000). It is also estimated that about 80%

of terrestrial carbon pools are stored in soils (Watson *et al.*, 2000). In the European Union (27 countries), total carbon stocks are estimated around 75 billion tons of C[4], 50% of which is located in Sweden, Finland and the United Kingdom due to the large area of peatlands in these countries (Schils *et al.*, 2008).

Changes in carbon stocks occur due to land uses or land use changes. There are steady-state differences between soil carbon levels in different land uses. Soils typically accumulate carbon at a high rate in grasslands and forests, which are net sinks (0-10^{11} $t.yr^{-1}$ C), and to a lesser degree in croplands (1-4 x 10^{10} $t.yr^{-1}$ C). Therefore, carbon losses are expected to occur following conversion to cropland, and increases when croplands are converted to grasslands or forests (Schils *et al.*, 2008). Even though forests are usually considered to accumulate more soil carbon than grasslands, some studies indicate otherwise (Ganuza and Almendros, 2003). Nonetheless, for a while forests store carbon in trees, and not only in soils. If timber is then used in long-lasting goods (e.g. furniture, houses), C may be stored indefinitely. The conversion between land uses itself, when resorting to tillage, implies a spike emission of carbon to the athmosphere, due to increased aeration of deep layers of soil, an corresponding mineralization.

The current balance of the agro-forestry sector in Portugal is negative (Pereira *et al.*, 2009b). According to APA (2006b), the Portuguese agro-forestry sector contributes to 10% of the country's total GHG emissions. This sector has increased its emissions in 7% since 1990, thus contributing to the national deficit. Considering particular GHG, agriculture is responsible for 65% of national N_2O emissions (APA, 2006b), associated with nitrogen fertilizer use and manure management (EEA, 2006). Agriculture is also responsible for 35% of national CH_4 emissions (APA, 2006b), mainly due to animal production (EEA, 2006).

Therefore, grasslands seem to be an important type of land use. They not only store large amount of C, but are also typically grazed by livestock, and therefore are related to two of the key issues respecting to GHG balances. Besides, in Portugal, many grasslands occur in low-density oak forest areas, and therefore they are an important link to sustainable landscape management, with implications far beyond carbon sequestration.

Recognizing this fact, FAO prepared a document (FAO, 2009) based on the results of a workshop held at Rome from 15 to 17 April 2009. The meeting featured several experts and members of the Grassland Carbon Working Group. In the document, FAO advocates the enhancement of carbon sequestration in grasslands as a low cost mitigation option with important environmental and economic co-benefits. The main goal of this document was to spur the discussion on the inclusion of land use and land use changes at COP15 in the Copenhagen summit and afterwards.

The calling made by FAO included a call of attention to the possibility of carbon sequestration in grasslands. The potential has been established for a long time, but questions regarding methodologies used and feasibility potential still remain. According to FAO, original sequestration enhancement possibilities are welcome. In those solutions, the dynamics of CO_2 and soil carbon in the atmosphere-soil-plant system should be connected with beneficial impacts in soil, water and biodiversity. Carbon sequestration via AFOLU activities would be especially interesting if it has other complementarity co-benefits.

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[4] Throughout the present thesis, we use the terms "CO_2", "carbon dioxide" and "carbon" as synonyms. When we are referring to atomic carbon, we use only the letter "C".

Portugal may have one of those original and cost-effective solutions with widespread benefits (Domingos *et al.*, 2009). In order to understand how, we now turn to the history of pastures in Portugal.

1.2 Main factors influencing ecosystem services and impacts in pastures

1.2.1 The history of pastures in Portugal

Since the beginning of the domestication of livestock by humans, fields of herbaceous plants, named in this sense as "pastures" or "grasslands"[5], have been used to feed animals. According to Diamond (1999), domesticated animals must have a flexible diet, consisting mainly of food such as grasses and forages, which are not part of human diet. This makes them less expensive to be kept in captivity than, for instance, carnivores.

Traditional pastures feature only spontaneous species, and were thus grazed by small herds requiring large areas. Figure 1 shows how pastures have become very important in Portuguese agriculture. We can see the decline in agricultural land, part of which was converted to forests and an even larger part that became "abandoned" land. Agricultural abandonment in the last decades in Portugal paved the way for extensive animal husbandry systems to take the place of croplands. When a previous agricultural land is abandoned, a primary mechanism of natural succession takes place. It is thereafter invaded by native herbaceous species first, and then by shrubs. Some of these plant species are palatable but certain types of livestock. As a consequence, many abandoned lands carried on as usable in their most productive seasons of the year as "natural" pastures. Traditional animal production is less labor-intensive and less input-demanding, and so it was one convenient solution to the economic situation in rural areas (Pereira *et al.*, 2009a).

It should be noticed that part of the loss in agricultural area also translates the fact that less area was required for the same level of production, due to the introduction of production factors such as improved seeds, pesticides, mechanical means and chemical fertilizers as substitutes of organic fertilizers. These new technologies led to productivity increases (measured in production per hectare) in areas where the agricultural activity was intensified.

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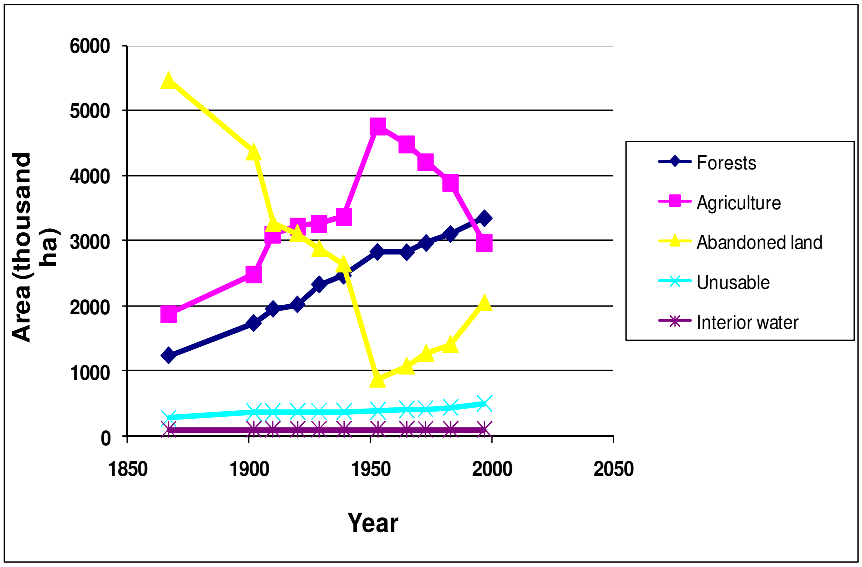
[5] The term "grassland" is usually referred to signify the plant system apart from other components, and pasture to express the whole grazing system (including animals). Since there is no significant gain in meaning from separating the two, and to keep language simple, in our work, we consider the terms "grassland" and "pasture" as synonyms.

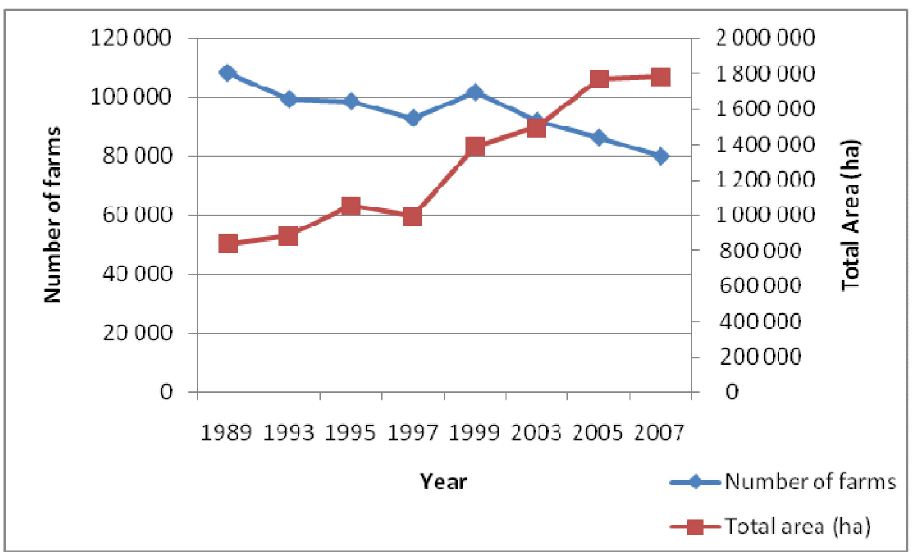
Figure 1 – Relative land use changes* in Portugal from 1850 to 2000.



*** The sum of all uses in each year is equal to the total area of the country. "Unusable area" is composed mainly of urban areas. Source: Pereira *et al.* (2004b).**

But Figure 2 shows that the area of pastures did increase. Even though the trend in the past years has been a decrease in the number of farms with pastures, the total area of pastures has doubled since 1989. Spontaneous herbaceous pastures are, therefore, key to understand the recent changes in Portuguese rural landscapes.

Figure 2 – Number of farms and total area of pastures per year between 1989 and 2007.



Source: INE (http://www.ine.pt, 2009).

To understand the whole chain of effects caused by the land use changes in Portugal, and also possible responses, there is one framework that can be used, and which we present next.

1.2.2 Main ecosystem services and impacts from pastures in Portugal

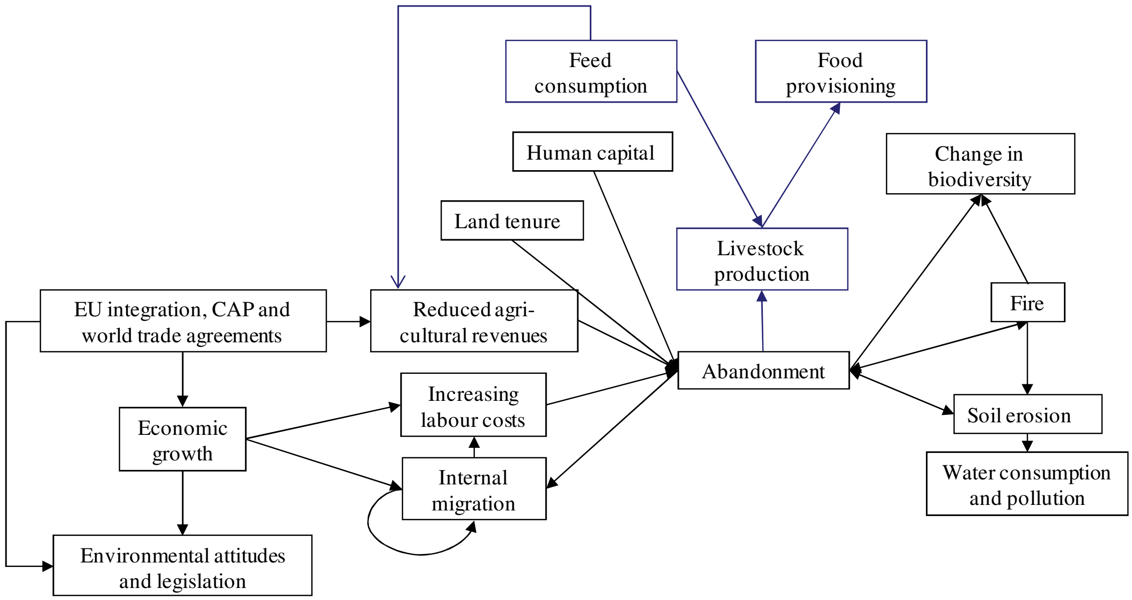
The Millennium ecosystem Assessment (MA) was launched by former United Nations (UN) Secretary-General Kofi Annan (MA, 2005). It aimed at a general worldwide assessment of the state of ecosystem services. Focusing on the consequences of ecosystem change for human well-being, the MA set the scientific basis for conservation, protection and valuation actions to be taken. Under the MA famework, several national, regional and local assessments were authorized. Portugal was one of the sub-global assessments. The Portuguese Millennium ecosystem Assessment (ptMA) had several case studies at three scales: (1) the national scale(whole country), (2) the basin scale (the Mondego and the Mira basins), and the local scale (Sistelo, Quinta da França, Ribeira Abaixo and Castro Verde).

At the national scale, the ptMA team chose the following services to be assessed (Pereira *et al.*, 2004b):

- Biodiversity;
- Provisioning services: water, food, and fiber;
- Regulating services: climate regulation, soil protection and runoff regulation;
- Cultural services: recreation.

The first step of the assessment was the determination of the main drivers of change in ecosystem services. The MA conceptual framework defines driver as any natural or human-induced factor that directly or indirectly causes a change in an ecosystem (MA, 2005). The ptMA determined the main drivers of change for Portugal through expert judgment, extensive literature review and workshops with the research team (Pereira *et al.*, 2004b). Since the increase in pastures has been a key change in land use in Portugal, we can adapt the ptMA findings to pastures, and include other specific environmental impacts, thus obtaining Figure 3.

Figure 3 – Main drivers of change in Portuguese Ecosystems regarding abandonment, according to the ptMA.



Adapted from: Pereira *et al.* (2004b). Original ptMA boxes are shown in black, and the boxes included by the present work are shown in blue.

A brief explanation of Figure 3 may be adapted from Pereira *et al.* (2004b)[6] as follows. The rise in the attractiveness of the industrial and service sectors in Portugal led to higher labor costs in agriculture (both in hired labor or opportunity costs for farmers). But the entry into the European Common Market and global trade agreements dropped agricultural prices, prices of which subsidies were hardly a sheer compensation. Agricultural firms then had to adapt. Some decided to abandon the agricultural activity, leaving fields unmanaged, or using them for extensive livestock production[7].

Many of these abandoned lands were used for extensive animal husbandry. But these "natural pastures" have significant problems. They can feed a small number of animals, which do not graze the totality of plants. Having resulted mainly from abandoned land, natural pastures are easily invaded by shrubs, which are primary stages of ecological succession. This process of succession may stop at very uninteresting degraded stages, and only active mangement can make them evolve further to climax. These shrubs have always had a tendency to appear, but were traditionally controlled by selective harvesting and use for animal bedding. Rising labor costs made this procedures, done by human hand, costly. Besides, less people managing such traditionally humanized landscapes as those in Portugal decreases discontinuities from compartmentalization, creating continuous areas of woody species left untouched and free to invade the pasture. This highly increases fire risk and severty (ISA, 2005).

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[6] For a full explanation, se Pereira *et al.* (2004b), page 22 and next.

[7] Options for farmers were the intensification of their land use, trading labor for mechanization and production factors, or to convert their production to more extensive (in the sense of less labor-demanding) land uses (such as afforestation, pastures or simple abandonment instead of crops). Intensification was chosen if land owners had enough financial resources, soil quality was high and water was available. Most farmers simply switched land use or purely abandoned the activity, as shown before in Figure 1.

For that reason, the necessity to control shrubs quickly became obvious. Even though animals control shrubs by stomping, degraded spontaneous pastures could only sustain low stocking rates, and therefore they are inefficient as a control method. Since it is too dangerous to use fire as a shrub control mechanism, mechanical action must be taken to shred the shrubs, decreasing fire risk and providing renewed conditions for nutritious plants to grow. But mechanical actions also have their flipside. They are usually intrusive and destructive for soil structure, leading to erosion phenomena and also carbon loss[8]. So, in the last 50 years, in an increasing fraction of Portuguese agricultural area, we had either high forest fire risk, which takes place without shrub control, or high soil erosion and carbon loss, which has been a natural consequence of damaging mechanical means to control shrubs. Fire and soil loss have dramatic consequences in terms of soil water holding capability and biodiversity promotion.

These pastures, however, provide the important provisioning service of food production (namely meat and meat products). But, even if livestock production is less labour-intensive and capital-intensive than alternative activities, it is also true that the degraded conditions of natural pastures do only support feeble stocking rates, with high inter and intra-annual variability in provision. Therefore, livestock will always require supplementation with forages or commercial feeds, which are expensive and further contribute to the decrease in agricultural revenue.

It is, then, clear that the use of abandoned land as pastures plays an important (negative) role in the (in)sustainability of the identified trend for land use changes by the ptMA. The previous framework clearly links environmental impacts with economic liabilities. It could be so that a solution for the promotion of ecosystem services also improves the economic viability of livestock production. In the past couple of decades, several strategies were devised and adopted to improve pasture performance. We now turn to those improvement strategies.

1.2.3 A response for the improvement of ecosystem services and impacts in Portuguese pastures

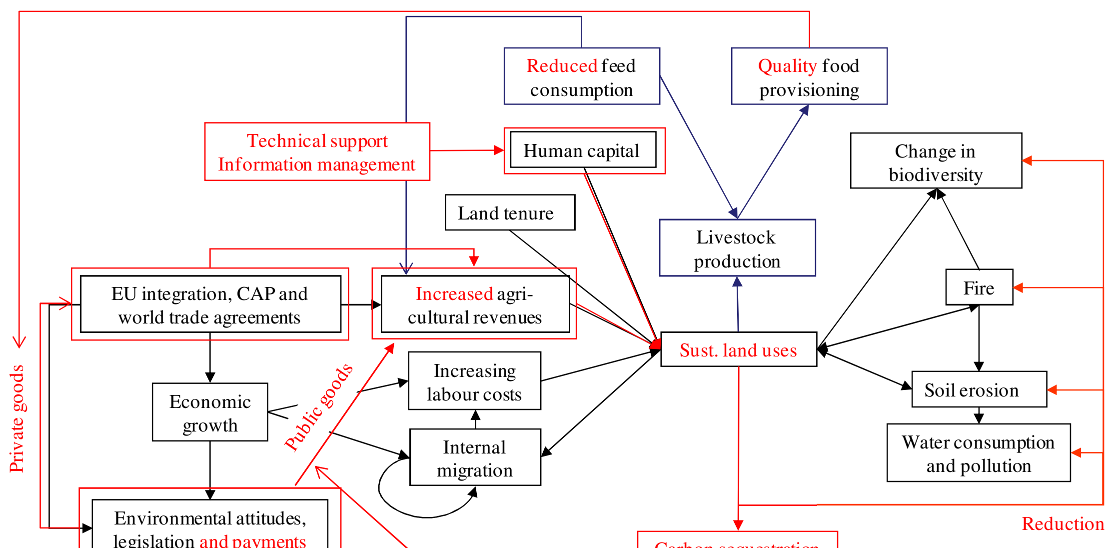
In the ptMA, some proposals for responses were formulated. Project Extensity was referred in the ptMA as one example of a coherent set of responses (Pereira *et al.*, 2004b). Project Extensity – Environmental and Sustainability Management Systems in Extensive Agriculture, was funded by the Life Program of the European Commission (LIFE03 ENV/P/505). The project took place between November 2003 and February 2008, reaching 86 farmers and over 70 000 ha spread throughout the country. In this farms, a sustainability management system was implemented. The system comprehended several levels, which went from the production of simplified Sustainability Reports to the implementation of the Eco-Management and Audit Scheme (EMAS) of the European Union. To obtain data, an LCA approach was privileged, and an information system was developed to collect it more efficiently. Once the system was implemented, various monitoring and analysis work projects were carried out, namely monitoring of biodiversity and the food chain. At the same time, consumers were scouted for their preferences through surveys. The reason why Extensity's responses are

[8] If the type and wight of the machinery is selected according to slope and soil type and condition, and if biomass is left and integrated into the soil, there are no evidences of such erosion. However, such is not traditionally the case in Portuguese agriculture, where tillage is still the most commonly used method of shrub control (Nuno Calado, personal communication).

perfectly adapted to pasture improvements is that the target farmers of the Project were those practicing extensive agriculture.

Responses from Project Extensity are shown in Figure 4, which may be read as follows. Technical support and knowledge for farmers increases human capital. This capital may lead them to search for sustainable land uses. Sustainable pastures must be more productive, in order to reduce feed consumption, and provide quality food (which is then also sold at a higher price). They must also reduce the environmental impacts of natural grasslands and, if possible, provide environmental services. One of those services should be carbon sequestration. Those land uses will only be sought if they increase farmers' revenue. This increase in revenue may be triggered by new opportunities from environmental regulations and payments for private goods and public goods. If carbon sequestration is one of those public goods, more chances of revenue exist, since carbon is the only environmental commodity for which there is a market currently established.

Figure 4 – Possible responses to improve ecosystem services in Portuguese Ecosystems regarding abandonment, according to the ptMA.



Adapted from: Pereira *et al.* (2004b). Original ptMA boxes are shown in black, and the boxes included by the present work are shown in blue. Responses are shown in red.

Figure 4 confirms that changing structural conditions, namely regarding the economic (increased revenue) and social (information and knowledge) aspects of rural zones, will positively influence the adoption of sustainable land uses. One important conclusion to be drawn from this analysis is that socio-economic and environmental sustainability go hand in hand in Portuguese agriculture. Rather than taking the economic performance and the environmental performance as substitutes, it is possible to design policies which are complementary in their goals. The rational use of public policies and private entrepreneurship may lead to practices which are environmentally positive.

This response, set by Project Extensity, is, then, an important course of action. But it remains to be determined whether there is a particular type of pasture that fits to this response, or weather it is only a theoretical possibility. The aim of this work will be to enquire if there is at least one type of pasture that provides the service of carbon sequestration will all other co-benefits refered here. In order to do so, we need to know first what kinds of pastures exist in Portugal today.

1.3 Which types of pasture are there in Portugal today?

1.3.1 Data sources

The main source of information for the characterization of Portuguese agriculture is the General Agricultural Survey done by the National Institute of Statistics (INE, the Portuguese acronym for *Instituto Nacional de Estatística*) every ten years. Since the last such Survey was done in 1999 (and was repeated in 2009, but results are yet unpublished), it is difficult to extrapolate the present situation of Portuguese agriculture from its findings.

However, a second important source of information is Project Extensity, which was directly aimed at farms with mostly extensive production systems, and pastures are, almost by definition, extensive or semi-extensive systems (as we mentioned in section 1.2.3). Therefore, the database collected from the Extensity farmers is both an update and a fine tune of the Survey done by INE.

1.3.2 Types of pasture

After inspecting data from both sources, it is possible to notice that pasture type varies with the following characteristics[9]:

- If the species are spontaneous or sown;
- If they are biodiverse[10] or not;
- If there is fertilization or not;
- If they are rainfed or irrigated;
- If they are under trees or in open areas.

However, some combinations of these options are not commonly found. To look at the representability of each type, the Extensity farmers were inquired as to what kind of pasture they have. Results are in Domingos *et al.* (2008), and can be summed up as shown in Table 1[11]. The same results may also be seen graphically in Figure 5. Table 1 shows that rainfed pastures clearly dominate. There are three significant groups of rainfed pastures[12]:

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[9] Some comments must also be done about the history of the field. However, those are now ignored for simplicity.

[10] The criterion for biodiversity is highly debatable. For rainfed pastures, 6 different species and varieties are considered to be the cut-off point for official uses (Rurual Development Programme, National Inventory Report). Still, some natural pastures comply with this requirement. The main originality of the SBPPRL system studied in this thesis is the fact that plant biodiversity is achieved by sowing.

[11] Note that in our analysis we only referred to permanent pastures. Temporary pastures are functionally similar to crops, since they require yearly sowing.

[12] In Project Extensity, two very large agricultural firms biased results significantly – Companhia das Lezírias and Grupo Sousa Cunhal. Together, they have more than half of the total area of pastures in Extensity. If we do not consider them, then sown biodiverse pastures are the majority.

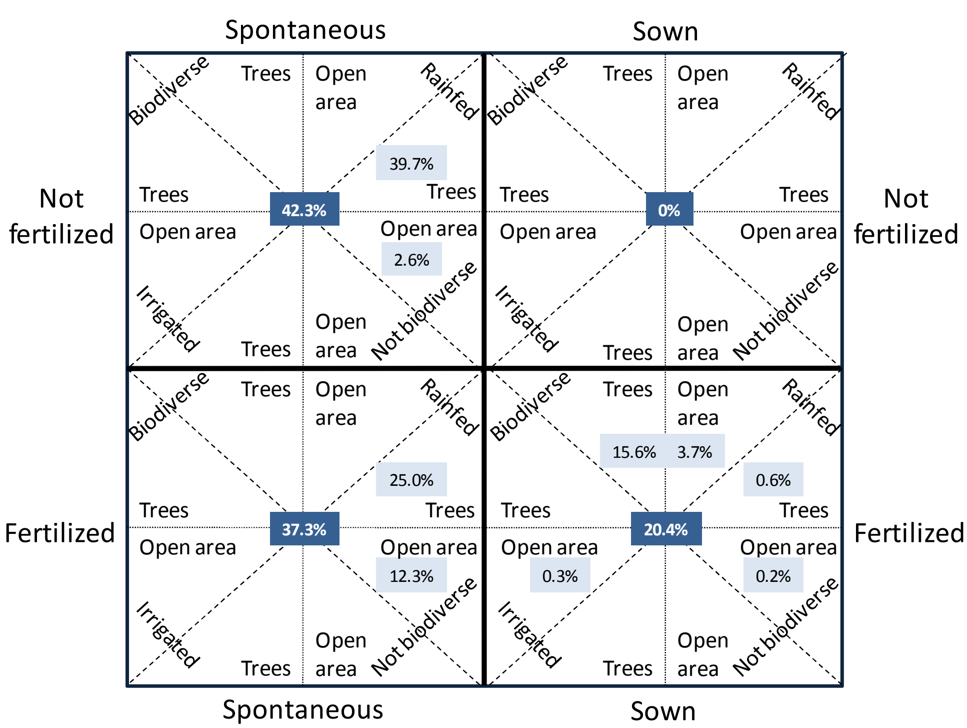
- Spontaneous unfertilized pastures;
- Spontaneous fertilized pastures;
- Sown biodiverse pastures.

Table 1 – Percentage of area for each type of pasture in the universe of farmers in Project Extensity.

Spontaneous or sown?	Fertilized?	Biodiverse[13]?	Rainfed or irrigated?	Under trees?	Area (%)
Spontaneous	Yes	No	Rainfed	Yes	25.0%
Spontaneous	Yes	No	Rainfed	No	12.3%
Spontaneous	No	No	Rainfed	Yes	39.7%
Spontaneous	No	No	Rainfed	No	2.6%
Sown	Yes	Yes	Rainfed	Yes	15.6%
Sown	Yes	Yes	Rainfed	No	3.7%
Sown	Yes	Yes	Irrigated	No	0.3%
Sown	Yes	No	Rainfed	Yes	0.6%
Sown	Yes	No	Rainfed	No	0.2%
All other options					0.0%
Total					*** 100.0%**

*** Total area of pastures was 17.950 ha.**

Figure 5 – Schematic representation of the different options for pasture types, and respective percentage of area of farmers in Project Extensity.



[13] In the case of Project Extensity, the arbitrary cutoff point for biodiversity was if more or less than 6 different species were sown (if it is a sown pasture) and are present today (for any type).

If Project Extensity has a representative sample of the fraction of Portuguese farmers with pastures, then we could assume that these three systems are also the most important in the rest of the country. But is this a fair assumption?

Considering results from the INE's Survey, there are two major differences. The first difference is that in 1999 only 55% of pastures in the country were under tree cover. For Project Extensity's areas, the figure rises to about 80%. This is a consequence of the fact that the cork and holm *Montado* zones are over represented in Extensity. *Montado* is an agro-forest system where usually pastures and oak trees coexist[14].

The second difference is that about 20% of the Extensity areas are sown biodiverse pastures. Moreover, they correspond to one specific system, the sown biodiverse permanent pastures rich in legumes (SBPPRL), which is the subject of this work and will be thoroughly analysed in chapters to come. However, this system is not present in 20% of the country's pasture land. Data from INE shows that there are about 1.8 million hectares of pastures in Portugal (Figure 2 above). But if we look at the number of hectares of these pastures which were purchased to Fertiprado (shown in Section 1.4.4), the main firm in the pasture seed business in Portugal, we see that only about 90 000 ha were sown, or about 7%. Therefore, the fraction of the area of sown biodiverse pastures is overestimated in the Extensity sample[15]. This is not a disadvantage, however, since, as will become evident in the next section, we will particularly focus on this type of grassland system from this Chapter on.

Regarding other systems, naturally (not sown) biodiverse pastures are underestimated among Extensity's farmers. This is because those pastures are mainly present in the Northern regions of Portugal. Those areas are also outside of the geographic scope of the present thesis, which is Central and South Portugal (since it is the region where SBPPRL are mostly present, and also the regions for which we have data). Sown pastures which are not biodiverse are not significant in area, since they correspond to failed installations of sown biodiverse pastures or to common mixes of one grass and one legume.

Therefore, we conclude from this analysis that it is possible to define three types of pastures which are the most representative of Portuguese pasture areas, all of which are rainfed. For purposes of linguistic and conceptual simplicity, we now define each of them as:

- Natural pasture (or grassland[16]) – a natural pasture is a spontaneous unfertilized rainfed pasture, which may or may not be under tree cover;
- Fertilized natural pasture (or grassland) – a fertilized natural pasture is a spontaneous fertilized rainfed pasture, which may or may not be under tree cover;

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[14] According to Pereira *et al.* (2004), "the *Montado* system is an evergreen oak woodland, where the predominant tree species are the cork-oak (*Quercus suber* L.) and holm-oak (*Quercus rotundifolia* Lam.). Montado is an agroforestry system, where the main activities are cork, livestock and cereal crop production. Montado is an economically and ecologically important system and is characteristic of Portugal and Spain (where it is called *dehesa*)".

[15] Due to the previous footnote, if we do not consider the bias from the two largest firms in Extensity, this difference would be even larger.

[16] Recall that we are using the terms "grassland" and "pasture" as synonyms.

- SBPPRL – a SBPPRL is a specific case of a sown fertilized rainfed pasture, consisting on a high number of plant species, amongst which there are many legumes.

We must notice that, when comparing these three systems, we are comparing three degrees of intensification. Natural pastures do not require any particular input or maintenance operation, apart from periodic shrub control operations. Their fertilization is an intermediate grade, and SBPPRL are the most "intensive" system, since they require seeds and other inputs, as we will discuss later on[17].

Keep in mind that we stated before that pastures are important, both as a driver (in the case of natural pastures, which are the reference situation in the largest fraction of the area) and as a response (in the case of optimization options such as fertilization or sowing), to the conditions and trends affecting ecosystem services. While natural pastures are straightforwardly defined, since they consist on any area with spontaneous herbaceous plants which is used for grazing, SBPPRL are an innovation which requires a clear definition.

1.4 Sown biodiverse pastures (SBPPRL)

1.4.1 Why biodiversity?

The generalized positive effects of plant biodiversity are now well established in the literature. Biodiversity in grassland composition increases plant productivity and the stability of nutrient retention in soils (Schläpfer and Schmid, 1999), by promoting the storage of both carbon and nitrogen in soils (de Deyn *et al.*, 2009).

Some authors, such as White *et al.* (2004), have argued that high species diversity in all plots of a given managed area is a positive factor in grassland areas. However, other authors, such as Duru *et al.* (2005) have argued that, even though the role of local biodiversity is unquestionable, it is also possible to maintain in the same managed area a mosaic of parcels, each one with low functional diversity, but highly diverse as a whole. Grazing intensities should also vary throughout the managed area, in order to allow different types of herbaceous plants to grow (McIntyre *et al.*, 2003).

One thing is for sure: biodiversity, either local or across a given range, is positive both to plant productivity and to ecosystem functioning (Schläpfer and Schmid, 1999; Spehn *et al.*, 2005). Sowing many species is important in the Portuguese heterogeneous soils. But it is functional diversity, more than number of species, that has the most effect in ecosystem processes (Díaz and Cabido, 2001; Wardle *et al.*, 2000). It is because biodiverse pastures have a high percentage of nitrogen-fixing legumes that they work like a well-oiled machine – legumes capture nitrogen, grasses use it to grow, sequestering carbon from the atmosphere.

1.4.2 Development of the SBPPRL system

As we've mentioned before, there are several types of pasture. Today, the sum of improved and spontaneous pastures makes up for about 70% of the world agricultural area (Suttie *et al.*, 2005). In Portugal, there are now around 1.8 million hectares of

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[17] Since SBPPRL are more "intensive" and also more productive, there is some area which is vacated for the same national animal stocking rate.

grasslands (INE, http://www.ine.pt, 2010), divided mainly between the three above mentioned grassland systems: Natural Grasslands (NG), Fertilized Natural Grasslands (FNG), and Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL). These three types of pasture correspond to three different degrees of intensification.

Data shown before supports the assessment that NG is by far the most used grassland system in Portugal. These pastures are relatively poor in terms of feedstock and have several associated environmental impacts. They consist of either fallow stages from long cereal rotations, or spontaneous vegetation in previous croplands which have since been converted to areas for livestock feed. NG typically have no specific management, except for occasional operations to control shrub growth. The most widely used operation is tillage.

The only difference between NG and FNG is that the latter are fertilized. The species and varieties of spontaneous grasses and legumes are the same, but fertilization increases productivity. Therefore, advocates of FNG claim that fertilization is a compromise between productivity and natural values. Therefore, in FNG, frequent shrub control methods are also required. Since the most significant area of pastures in Portugal is located in cork and holm oak agro-forestry systems (*Montado*), all types of natural pasture give room for some natural regeneration of trees. Natural regeneration is the result of inefficient control of shrubs, which are primary stages of the succession mechanism of *montado*. Furthermore, advocates of FNG claim that methods of shrub control other than tillage will benefit the soil nutrient recycling system. Shrubs have deeper roots than grasses and legumes, and therefore access nutrients in deeper layers of soil to grow. Control operations will then shred their aboveground biomass. This biomass remains on the ground and is incorporated in the first layer of soil, which is then used by pasture plants to grow.

Starting from the reference situation of natural (degraded) pastures, some other farmers and agricultural scientists believe that fertilization alone does not provide the best results in terms of plant productivity and animal feed quality. Advocates of sown pastures believe that the introduction of specific species or varieties, either absent or in lesser percentage in spontaneous grasslands (as, for example, some varieties of legumes) will establish a functioning ecosystem with complementary ecological niches and improve production.

This line of thought led to the development in Portugal in the 70s of an alternative, using the concept of “biodiversity engineering” as a defence against the very diverse Mediterranean natural conditions, which are SBPPRL. They have been installed throughout some regions of the country in the past decade, as well as in some minor areas in Spain and Italy.

The SBPPRL system consist of diverse mixes of up to twenty different species or varieties of seeds, and are rich in legumes. Commonly SBPPRL are more productive than natural grasslands, and are also richer in number of species (Carneiro *et al.*, 2005). There are less gaps in plant cover throughout the plots, since species variability ensures that the species most suited for each spatial conditions will thrive. Even though there is a well documented experience with the use of sown pastures (FAO/CIHEAM, 2008), this specific system only exists in Portugal and, to a lesser degree, Spain and Italy. There are many studies on the role of biodiversity in productivity, but SBPPRL remain the only widespread large-scale application of what may be called “biodiversity engineering”.

Since SBPPRL are a relatively new system which, to our knowledge, has not yet been fully studied, we decided to dedicate this chapter to its description. The next sections will be used to explain their environmental and economic effects.

1.4.3 Qualitative description of SBPPRL

SBPPRL have higher productivity than natural grasslands, and are also richer in number of species. Their high productivity is due to the fact that biodiversity allows the most adapted plants to prosper in each zone. The seed mix is designed specifically for each location after soil sampling and analysis. Species in the mix are adapted to soil physical and chemical characteristics, as well as to local climate conditions, and therefore there is no single representative mix. However, some very common sown species in SBPPRL mixes are *Trifolium subterraneum, Trifolium michelianum, Ornithopus spp., Biserrula pelecinus,* annual *Medicago spp.,* and grass species of the genera *Lolium, Dactylis and Phalaris*. The mixes of sown species are often enriched with seeds from spontaneous plants such as *Plantago spp*., *Vulpia spp.* and *Bromus spp* (Carneiro *et al.*, 2005; David Crespo, personal communication). Legumes are inoculated with bacteria of the genus *Rhizobium*. These bacteria induce nitrogen-fixing nodules in the roots of legumes. The fixated atmospheric nitrogen is then used by grasses. Therefore, the overall system is self-sufficient in terms of nitrogen. Legumes cover more than 50% of first-year SBPPRL (Carneiro *et al.*, 2005). As pasture settlement progresses, legumes increase and eventually dominate. Percentage of legumes in the plant cover of a mature SBPPRL (more than 5 years) is around 25 to 30%.

Increased productivity in SBPPRL allows a sustainable increase in animal carrying capacity. Animals graze the plants, which have an annual life cycle. High plant productivity implies increased atmospheric carbon capture through photosynthesis. Part of the biomass produced is stored in soils due to the high density of yearly-renewed roots. Storage is in the form of non-labile Soil Organic Carbon (SOC), which is part of the Soil Organic Matter (SOM) pools. SOM pools are also increased by leaves' senescence and decomposition, and by animals returning undigested fibre to the soil.

Increasing SOM improves soil nutrient availability and water holding capacity, thus increasing plant productivity[18] and reducing surface runoff of water, which in turn decreases sediment loss and soil erosion (EEA, 2004). Decreasing water runoff and soil erosion have positive effects even outside the plot. Sediments, nutrients, organic matter and pesticides carried in water contribute to silting, eutrophication and contamination of surface waters. These effects are known, but their true costs are still hard to estimate[19]. Nitrogen fixation by legumes eliminates the need for nitrogen fertilizers, whose production is highly energy demanding, and therefore responsible for high greenhouse gas emissions. Finally, both increased stocking rate and reduced fertilizer use increase the economic viability of the farms. This is particularly important because socio-political and economic conditions are barriers to the successful implementation and

[18] Water holding capacity allows the establishment of a soil water reserve which is then used throughout the months with higher water deficit. This effect attenuates inbalances in productivity during the year..

[19] Note that at the regions where SBPPRL are mostly sown (South and Central Portugal), surface runnof only takes place at extreme rainfall events and in high slopes. Therefore, the positive environmental effects of decreasing runnof seldom take place strictly due to SOM. It must also be noticed, however, that the effect of increasing water holding capacity always takes place in soils with high SOM. This effect is agronomically very important, as it increases production.

management of pasture systems which provide environmental services (Neely *et al.*, 2009).

It should be noticed that grasslands and agricultural soils do not store carbon indefinitely. As SOM content increases, so does the organic matter mineralization rate, while the soil's storage rate decreases. As storage and mineralization rate become equal, eventually a steady-state is reached. In that steady-state, there is no carbon sequestration, but the system maintains all the other advantages. Therefore, SBPPRL could be relevant for carbon sequestration in the short term, but it's other environmental and economic benefits that can prove their value in the long term.

Increasing SOM, nutrient availability and water in soils provides both mitigation and adaptation to climate change. SOM accumulation through an increase of SOC is the mechanism through which carbon is sequestered in grassland soils. This is particularly important for Portugal, since, as we refered before, it was one of the few countries to elect the "Grassland Management" voluntary activity, in the framework of the Land Use, Land Use Change and Forestry (LULUCF) activities, now named Agriculture, Forestry and Other Land Uses (AFOLU), under Article 3.4 of the Kyoto Protocol. This choice was made mainly because of the implementation of the SBPPRL system in Portugal. However, there is currently no study published on the potential of the SBPPRL system to increase SOM. According to IPCC guidelines, Tier 1 approaches imply that one sequestration factor is attributed, regardless of related field data. The present work should be a first step: we study how much, on average, SBPPRL increase SOM, in relation to the baseline, which are natural grasslands.

1.4.4 Implementation of SBPPRL in Portugal

Fertiprado started selling those mixes of SBPPRL seeds in 1996, and have since then become the most representative seller in Portugal. Therefore, the data series for Fertiprado sales provides an approximate depiction of the evolution of the installation of SBPPRL in Portugal. Therefore, they can also be used to extrapolate future installations of pastures.

In order to do so, we observed that sales have followed an approximately logistic increase. After an approximately constant increase during the first years, there was an accelerated increase in sales after the year 2000. That increase has been lowering from 2005 on. To model this logistic curve, we chose a Verhulst-Pearl function for population growth. This is a non-linear function using two parameters, *r* and *K*, the first one being the growth rate in the initial stage (yearly installation) and the last one the carrying capacity, corresponding to the asymptotic maximum (maximum area of pastures installed). In the case, population growth is used as an analogy for area growth due to spreading of the word between farmers, in the absence of any extra support, as for example the Terraprima-Portuguese Carbon Fund (PCF) Project.

The logictic equation we used is:

$$\frac{dP}{dt} = rP\left(1 - \frac{P}{K}\right), \tag{1.1}$$

where P is the area of SBPPRL installed in the year t. This differential equation has an exact solution, which is:

$$P(t) = \frac{K \cdot P_0}{\left(K - P_0\right)e^{-rt} + P_0}, \tag{1.2}$$

where P_0 is the SBPPRL area installed in the first year[20] (1996), and *t* is the number of years. This function is not linear in *t*, and therefore must be calibrated using non-linear estimation. We used actual Fertiprado sales for the calibration, and applied the numerical algorithm in software SPSS 16 to perform successive iterations on the parameters. The final values established for the parameters are those that minimize the mean quadratic error in the estimation of the input data (Fertiprado sales). Calculations were made for the whole country, and also for each of the three most representative regions of SBPPRL installation: Beira Interior, Ribatejo e Oeste, and Alentejo.

Parameters obtained, as shown in Table 2, indicate that the maximum area obtainable (parameter *K*) in the sum of the three regions would be 109 535 ha. Around 80% of this area is in the Alentejo region.

Table 2 – Estimated parameters for the logistic model of SBPPRL area installed in Portugal.

Estimated parameters	Alentejo	Ribatejo e Oeste	Beira Interior	Total
r (yr^{-1})	0.447	0.354	0.372	0.429
K (ha)	86 739	11 091	12 415	109 535
P_0 (ha)	1,783	359	325	2,443

r, K, P_0 – parameters in the logistic function.

Table 3 shows observerd areas and areas calculated using the calibrated logistic model for the total of the three regions.

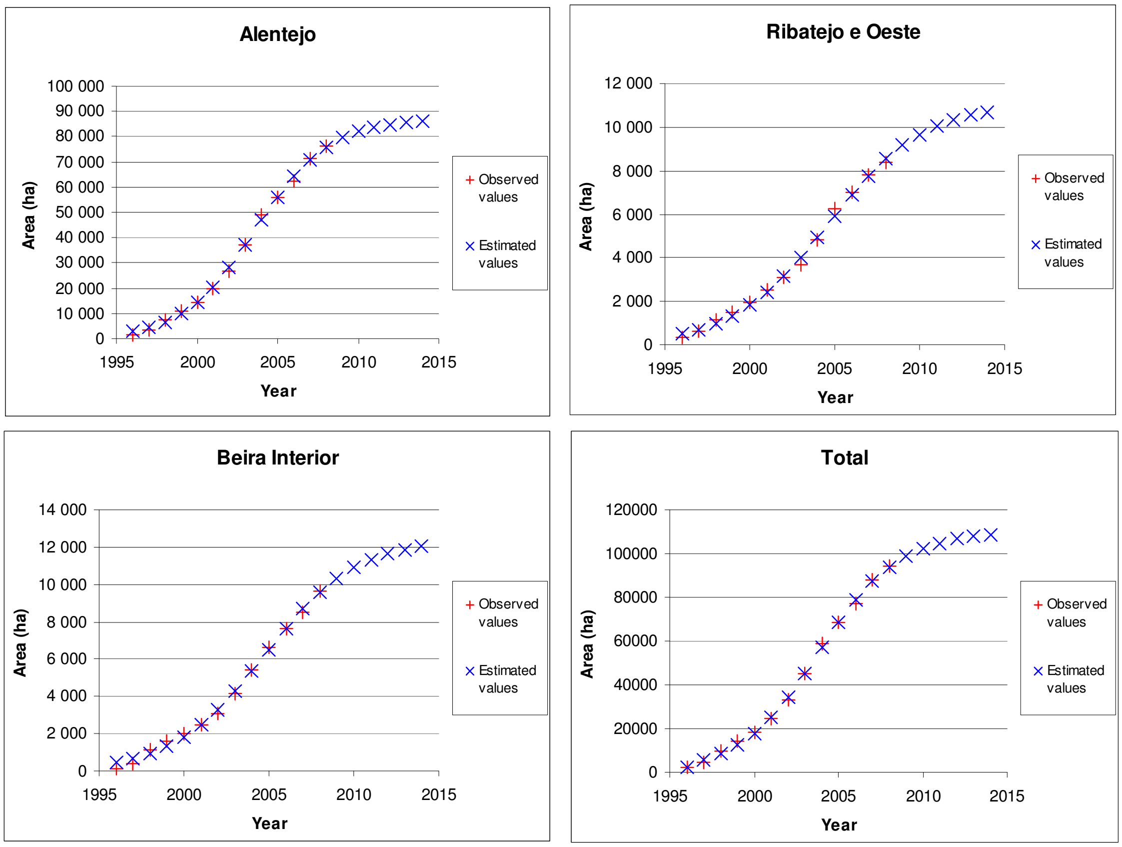
Table 3 – Cumulative area of SBPPRL (observed and calculated using the logistic model), according to sales from Fertiprado.

Year	Observed area (ha)	Calculated area (ha)
1996	2 071	2 443
1997	4 459	5 593
1998	9 439	8 361
1999	14 042	12 336
2000	18 505	17 866
2001	24 564	25 233
2002	32 907	34 494
2003	45 081	45 326
2004	59 120	56 977
2005	68 786	68 431
2006	77 045	78 738
2007	87 751	87 301
2008	*94 260	93 954
2009	-	98 859
2010	-	102 339
2011	-	104 739

[20] Note that, for estimation purposes, P_0 is a parameter and not an input, in order to avoid anchoring the model to the initial point.

Figure 6 represents values in Table 3, and also for each of the regions. Visual inspection clearly shows that the logistic pattern is a good fit, particularly for the sum of the three areas. It is important to stress that observed values can only be considered until 2008, since 2009 marked the beginning of the Terraprima-PCF Project (we will explain what we mean by this Project in due time, namely in Chapter 4).

Figure 6 – Observed and estimated area using a logistic model, including forescast.



Upper right corner: Alentejo region; lower right corner: Beira Interior region; upper left corner: Ribatejo e Oeste region; lower left corner: total.

1.4.5 Interpretation of the reasons for the pace of SBPPRL implementation

If SBPPRL are indeed a good option for farmers, with many economic and environmental advantages, then it is somewhat puzzling at first sight that the rate of their implementation has been decreasing.

There are many possible explanations for the fact that some farmers stop installing pastures and other farmers never install them at all. Many of them have to do with social reasons, namely cultural barriers. There seems to be an intuitive idea amongst farmers that annual pastures are more productive than permanent pastures, even though annual pastures are more expensive and do not show better results. This idea is persuasive because farmers are economic agents sensitive to social representations of their activity, and believe that showing soil management implies actively performing actions such as sowing every year (João Paulo Crespo, personal communication). One dramatic example of this are most parts of the Trás-os-Montes region, where farmers are socially well-regarded as good soil managers if they keep grasslands completely safe from infestants all year round. They achieve this by recurring to very frequent tillage, sometimes as often as two or three times a year (António Martelo, personal communication). Due to the slope of most plots in the region, this accelerates soil loss and, in the medium run, greatly decreases productivity. Keeping a permanent pasture also implies good management, but in a less visible way, and thus the social perception of the activity of the farmer is harder.

We must also note that SBPPRL is a relatively new and innovative system, which implies some knowledge transfer not only to break social barriers, but also to guarantee that farmers manage the pastures correctly. Financially, the agricultural sector faces risks that the farmer does not control (price fluctuation of inputs and of products sold, climate, among others), even for activities that the farmer dominates. Uncertainty is even higher for innovations, and so farmers are more risk adverse.

For example, the year 2008 was exceptional, in that the price of almost all production factors (fuels and fertilizers, mainly) raised significantly (Pedro Silveira, personal communication), while the sale price for livestock, meat and meat products seeply decreased. Even though it could be argued that SBPPRL can be a defense against such uncertainty (since they can be more productive, and better soils are more reliable against uncertain climate conditions, they are always better than the alternative of natural pastures), there is an important block to their installation. SBPPRL require a significant initial investment for the installation, which must be made in the first year, and 25% of which can be supported by public funds at most. The first year is a particular year in management as well, since grazing must be limited in order to allow plants to complete their cycle undisturbed, and establish a good seed bank. This seed bank ensures the renewal of the pasture each year, and therefore is crucial for its maintenance. Some farmers cannot support this first-year loss.

Besides, one must consider that there is a whole range of farm sizes and natural conditions. For some farmers, the expected revenue is higher (farms with naturally good initial soil and climate conditions), and the costs are lower (for example, for very large areas some fixed costs are diluted). These farmers, who had the largest benefit from sowing SBPPRL, were the first ones to install them, and they installed them where it was less expensive to do so. We may depict the most likely sequence of adoption of the system as follows:

- First, only well informed farmers with large areas and capital to invest installed SBPPRL. First-movers installed them in places where the costs with machinery are lower, namely in valleys and other clear areas.
- Then, after all clear areas were filled, farmers persuaded by the results obtained with this system started sowing pastures in agro-forestry areas within their farms. It is unknown in what fraction of the farm farmers stop sowing pastures, but since they sow them essentially for livestock feed, it is plausible to assume that they sow as many pastures as needed to end concentrated feeds consumption. We will explore this assumption in further detain in Chapter 3.
- Other farmers became aware of the existence of the system and of its results, and the area expanded at a high rate afterwards.
- As all farmers who expect to receive a positive income from sowing SBPPRL install pastures, the rate of installation starts decreasing, up until the point when

almost all other farmers with grasslands can expect a negative revenue from SBPPRL.

If this sequence of events is correct, than the existence of a specific support such as, for example, payments from the PCF would make the system worthwile for farmers who, otherwise, would not have a positive income from SBPPRL. Those would be the farmers that would then be interested to install pastures, and they would be the implementation scenario for such a project. But to know if the PCF could support such a system, we must first establish why SBPPRL promote cabon sequestration.

1.5 Comparison of natural pastures and SBPPRL

In this Section we will qualitatively compare natural pastures and SBPPRL in terms of the three sustainability sub-systems we defined. We will not mention fertilized natural pastures, since they are an intermediate situation between those two.

The basic scheme of socio-economic and environmental effects in livestock production in natural pastures is shown in Figure 7. Figure 7 should be read as follows.

1. Natural pastures are less productive[21], both above and belowground. Low belowground productivity is translated by low SOM.
2. Low SOM and low soil cover by pasture plants imply a bad soil structure, leading to more erosion, less water retention and consequent decreased flood regulation. More superficial runoff leads to increase erosion as well.
3. Since natural pastures are less productive, they feed fewer animals, and so are typically exploited with low sustainable stocking rates. Low stocking rates mean that animals do not control shrubs, which increase in the field. Besides, low productivity means that shrubs increase, since they find small competition from herbaceous plants (grasses and legumes).
4. Livestock in pastures emits methane, which is a GHG.
5. Since pastures are not productive enough, the only way to balance the animals' diet is to recur to concentrated feeds. These feeds are composed mainly of cereals and oilseeds, which required fertilizers to be produced.
6. Lower SOM has a negative effect on soil fauna, but low stocking rates and more woody vegetation may have a positive effect on biodiversity.
7. Constantly low SOM means that no carbon sequestration occurs.

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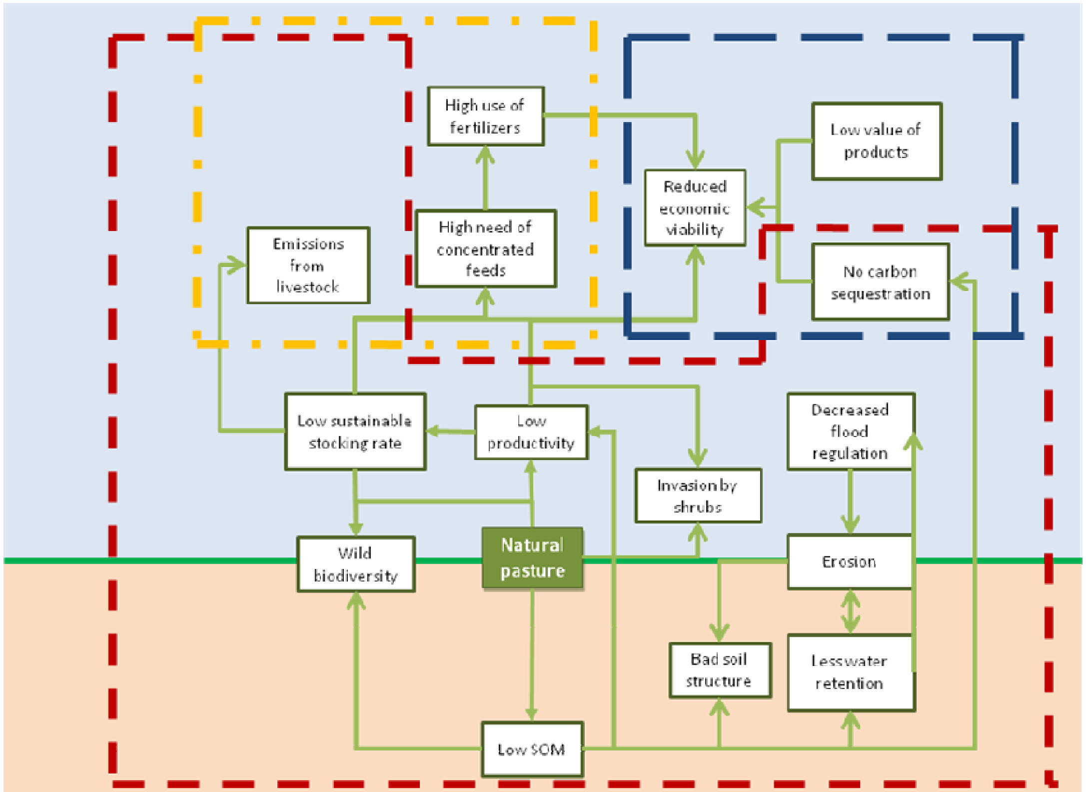
[21] Note that all qualifiers (e.g. low, high, more, less) are attributed to each pasture type (natural, SBPPRL) in relation to the other, except when explicitly referred.

Figure 7 – Causal scheme of effects of livestock production in natural pastures.



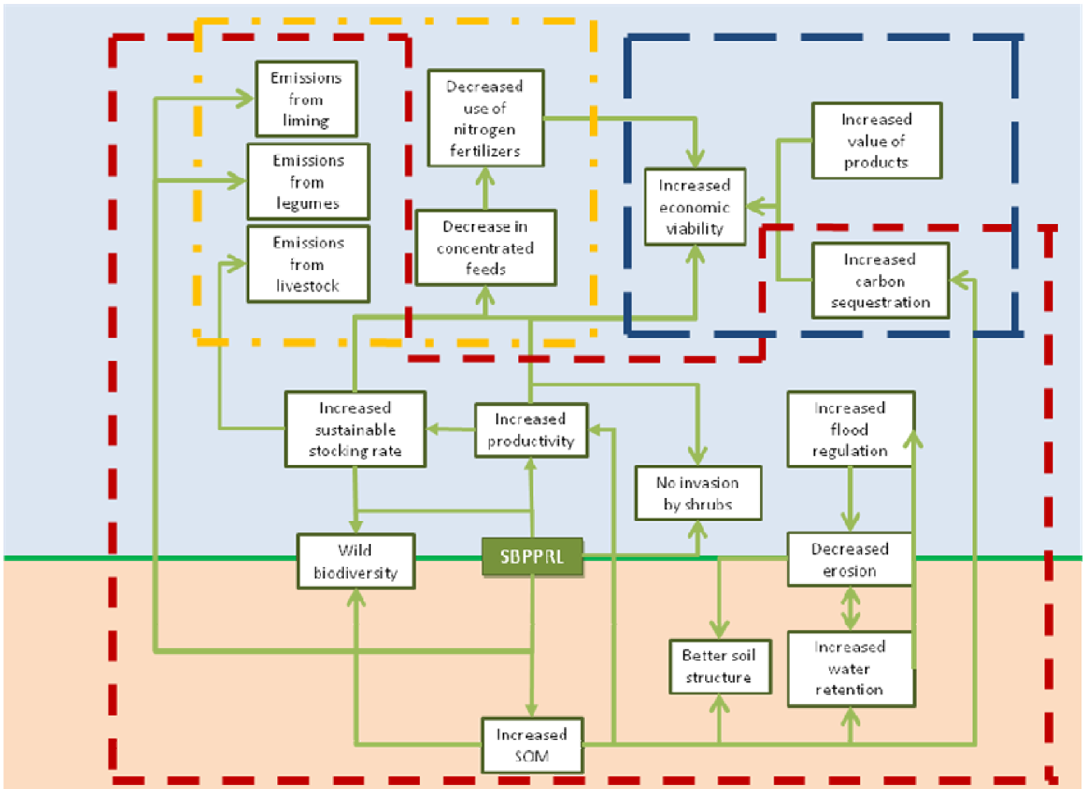
Red dashed line – Direct environmental effects. Yellow dashed line – Indirect environmental effects. Blue dashed line – Socio-economic effects.

Regarding SBPPRL, the scheme in Figure 7 becomes similar to the one in Figure 8. Again, we can make nine statements similar, one by one, to ones made for natural pastures. The analysis of each link at a time is as follows.

1. SBPPRL are more productive, both above and belowground. High belowground productivity is translated by high SOM.
2. High SOM and increased soil cover by pasture plants imply an improved soil structure, leading to less erosion, more water retention and consequent increased flood regulation. Less superficial runoff leads to a decrease in erosion as well.
3. Since SBPPRL are more productive, they feed more animals, and so are typically exploited with high sustainable stocking rates. High stocking rates mean that animals control shrubs either by stomping or by using them in their feed, since shrubs are rich in fiber, to complement for excess protein from the consumption of legumes. Besides, higher productivity of grasses and legumes leaves fewer resources available for shrubs.
4. Increased stocking rates mean that livestock in pastures emits more methane, which is a GHG. There are also emissions due to biological fixation of nitrogen by legumes, and emissions from liming, a process required to increase the pH of soils (if they are too acid).
5. Since SBPPRL are more productive enough, there is less need to recur to concentrated feeds. These feeds are composed mainly of cereals and oilseeds, which required fertilizers to be produced, and which are then avoided.

6. Higher SOM has a positive effect on soil fauna, but higher stocking rates and less woody vegetation have a negative effect on biodiversity.
7. High SOM increases mean that carbon sequestration occurs.

Figure 8 – Causal scheme of effects of livestock production in SBPPRL.



SBPPRL – Sown Biodiverse Permament Pastures Rich in Legumes. Red dashed line – Direct environmental effects. Yellow dashed line – Indirect environmental effects. Blue dashed line – Socio-economic effects.

Differences between the two methods are summarized in Table 4.

Table 4 - Differences between baseline and proposed scenarios.

Baseline Scenario	Proposed Scenario
Degraded natural grasslands / former cropland areas:	**Sown biodiverse permanent pastures rich in legumes:**
• Net carbon emissions (from animals)	• Carbon sequestration by agricultural soils and improved soil fertility
• Low stocking rate	• Increased stocking rate
• Shrub invasion and fire	• Shrub control and reduced fire risk
• Low inputs and machinery	• Increase in production factors' consumption
• Synthetic nitrogen fertilization	• Nitrogen fixation by legumes
• Erosion and low water cycle regulation	• Benefits in soil and water cycle regulation

This *a priori* descriptive comparison constitutes the basic hypothesis we tested in this thesis. In the next Chapters, we will justify each of the seven statements made before.

When speaking of pastures, we must always keep in mind their final goal, which is to feed livestock, and, as evidenced in Figure 7 and Figure 8, animals are a crucial part of both the system's environmental and economic effects. So, the final part of the present Introduction regards the definition of scenarios for livestock production (type of animal and life span in pastures).

1.6 Pastures and meat production

Since the beginning of the domestication of livestock by humans, fields of herbaceous plants have been used to feed animals. According to Diamond (1999), one of the important reasons for any animal to be domesticated is to have a flexible diet, particularly if it consists on food such as grasses and forages, which is not consumed by humans. This makes them less expensive to be kept in captivity than, for instance, the domestication of carnivores, which would mean that other animals would have to be bred and fed to them.

Therefore, herbivores have been the natural choice for animals domesticated for food. They are, together with fish, the main source of animal protein in human diet. Grasses and most legumes are traditionally the choice to feed livestock, since they are not used in human diets.

Traditionally, these animals were, then, fed in pastures and from excess crops. Pastures could not support large stocking rates, and meat production was, a consequence, low. In many countries, including Portugal in the beginning of the 20th Century, meat was a luxury in many households.

With the growth of the concentrated feed industry, animals begun being placed in small areas called stables where they are fed only feeds (and, to a smaller degree, straw or forages. This new option for animal production is considered intensive production.

1.6.1 Intensive vs. extensive production

1.6.1.1. <u>An issue of intensity</u>

The intensity of animal or agricultural production is an issue connected with the relative *intensity* of production at a given place (tons per hectare for plant production, and heads per hectare for animal production). Thinking in terms of animal production only, when comparing both systems we are comparing how much area we need to produce each animal.

This concept of area, however, has both direct and indirect consequences. Direct consequences deal with the fact that in extensive production a relatively low number of animals graze relatively extensive areas. Meanwhile, in intensive production systems, a relatively high number of animals are stabled indoors. Therefore, switching from extensive to intensive production, there are areas that become vacant and may be occupied by other land uses. Whether the environmental balance of this substitution is positive or negative depends on the specific alternative land use.

However, significant indirect effects also arise, which deal with the use of commercial feeds. The relevance of commercial feeds derives from the fact that in both intensive (always) and extensive (during the least productive seasons) production systems of meat it is necessary to provide feeding to the animals elsewhere than the pasture.

Since the 1950's, consumption of meat products has increased steadily. It is considered that 1 kg of beef requires 7 kg of high-protein feedstuffs (Brown *et al.,* 1999), 1 kg of pork requires 4 kg of grain (CIWF, 1999) and 1kg of poultry requires 2 kg of feed (CIWF, 1999). Therefore, a higher meat production corresponds to a higher ingredient demand. Today, 95% of the world's soybean production and a third of commercial fish catches are used for animal rather than human feed (Millstone and Lang, 2003). The area needed to produce feeds for the increasing number of animals is, then, high; 75% of all agricultural land in the United States is used to produce ingredients in livestock feed (Millstone and Lang, 2003). This contributes significantly to the impacts of crop production, aggravated by the impact of the animals themselves, namely regarding soil loss and desertification – 85% of all topsoil loss in the United States is attributable to livestock ranching (Millstone and Lang, 2003). Therefore, an increasing number of studies have tried to assess sustainability in agriculture (Lewandowski *et al.*, 1999), particularly researching the environmental impact of feeds (Cederberg and Mattsson, 2000; van der Werf *et al.,* 2005).They invariably conclude that concentrated feed production and transportation is the main souce of environmental impact in the meat production chain.

In Portugal, the intensification trend was also followed. The animal feed industry is the third most important in the agricultural sector, representing 10.5% of total business volume in 2002 (IACA, 2004). The total production of feeds in the same year has been estimated in almost 3.5 million tons (IACA, 2004).

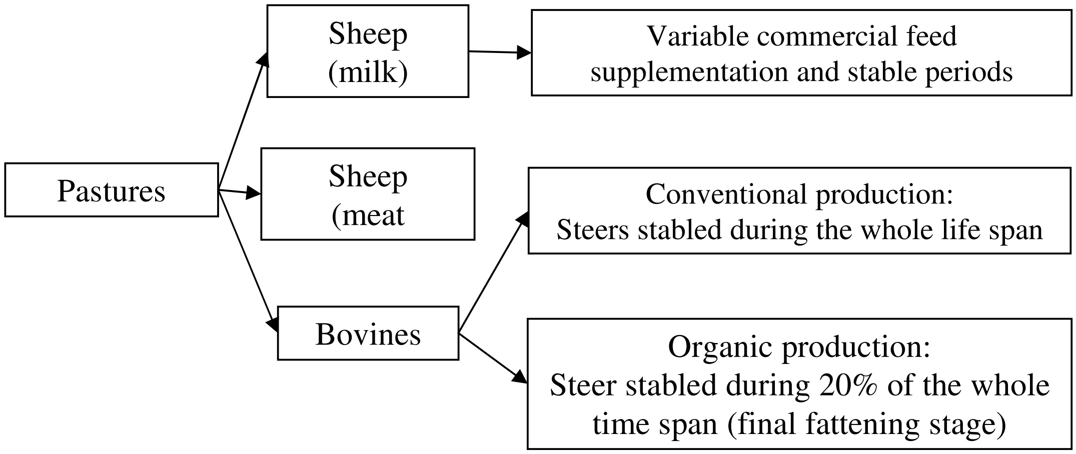
However, there is not always a competition between intensive and extensive production. Given the current structure of Portuguese livestock production, in some cases the comparison does not make sense.

1.6.1.2. Which animals go were?

To understand in which cases the intensive and extensive approaches are rivals, again we turn to results from the survey to Project Extensity's farmers in Domingos *et al.* (2008). There are three types of animals which are produced resorting to pastures (Figure 9). Sheep fed for meat production are always kept in pastures, whereas sheep for milk production are subjected to a mixed system consisting of variable time in pastures and stables, depending on the stage of lactation. Bovines, usually used for meat productions (amongst Extensity farmers), are either stabled during their whole time span[22] (purely intensive production), or only during the final period of fattening (extensive production). Note that, while stabled, animals are fed with concentrated feeds and straw or forages. We will come back to this issue when we analyse animal feeds.

[22] Time span, in this context, is defined as the time between weaning and slaying.

Figure 9 – Different types of animals and production methods, according to the survey to Extensity farmers.



Livestock production gives origin to different products (e.g. meat, milk, leather). Since each one has a particular economic value, we tried to restrict our analysis to one, which was meat. This choice is linked to another one, which was the type of animal studied. Each animal gives origin to several meat types, all of which are important in a balanced diet. However, it would be impossible to make a thorough study of all of them, given the specific economic data involved in each one, and also the characterization effort required to fully depict the whole system (e.g. feeds used). Therefore, in our work, we had to keep to just one type of animal. We chose bovines, namely those involved in the meat production system – breeding cows and steers. But even making this choice, specific scenarios for fattening and weaning must be defined. This is the subject of the next section.

1.7 Definition of scenarios

So from everything we stated before, we can now define the scenarios we chose to study. We will study three types of grassland, from the less extensive one (natural pastures) to the most intensive one (SBPPRL). We also determined that the product we will focus on will be bovine meat.

We justify our choice of bovine meat as the main indicator because it is currently the main source of income to most animal producers in the country, at least between those with SBPPRL (Pedro Silveira, personal communication). This will be clearer when we characterize the economic costs of pastures and describe the revenue and public supports available. Furthermore, bovine meat is still one of the most consumed meat types by Portuguese households. And, last but not least, bovine meat is the product for which more information is available.

Since we chose bovine cattle, we must decide on which scenarios to use for animal feed and weaning. Those scenarios were devised from the experience of project Extensity.

In extensive production, cows typically spend their entire life cycle grazing in pastures, and may or may not be supplemented with concentrated feeds or forages. Each cow has one steer per year, which may be weaned immediately after birth or around 6 months after. When weaned at birth, steers follow a completely intensive production life cycle, since they are stabled and fed entirely on concentrated feeds and forage. When weaned at six months of age, they may be kept in pastures or they may be partially or totally stabled, and in each of the cases they may or may not be fed with concentrated feeds. Steers are then slayed at 12 to 24 months of age. Which percentage follows each path is unknown. However, most steers are transferred to intensive production after weaning. This is because steers that stay in pastures for more than 6 months give origin to meat which is darker in color, and therefore is disregarded by consumers. We will return to this is issue in the last Chapter.

In this thesis, we consistently used a scenario consisting on the following characteristics:

- Functional unit: one average area unit per time unit (hectare per year) of a pasture plot.
- Field data were obtained at the plot level (individual SBPPRL and NG, regardless of the rest of the farm). We restrain our analysis to the conditions in which data was obtained, which were that only cows grazed the plots, eating the same amount of concentrated feed in both SBPPRL and NG plots. As we will show, SBPPRL have higher stocking rates than NG, but this is due to more grazing days.
- Cows have one steer per year. Steers are removed from the pasture after weaning at six months of age.
- For a whole-farm assessment, we consider that farmers sow SBPPRL in their farms in order to eliminate the need for concentrated feeds as much as possible.

Despite the use of this scenario, we refer as often as possible alternative scenarios and their consequences. For example, for the carbon balance in Chapter 3, we also assumed changes in stocking rate due to transfers of steers from intensive to extensive production, as well as feed intake from cows, and respective decrease as a way to make use of the higher sustainable stocking rate of SBPPRL.

1.8 Is our study on the right side of the fence?

The system boundary for our study is the animal farm gate. Therefore, we do not study the whole life cycle of meat. This may seem strange at first, since meat is one of the products with higher environmental impacts. Project EIPRO – Environmental Impact of PROducts (Tukker *et al.*, 2006) studied which were the groups of products with the greatest impacts. They conclude that those groups are food and drink, housing and private transport.

For Portuguese meat production, a specific study was done by Simões *et al.* (2005). All steps in the life cycle of meat were considered, namely the production of animal feed (concentrated feeds and pastures), slaughtering, transportation, storage, private transport to shops and cooking. It was determined that about 90% of the total impact in two categories (namely greenhouse gas emissions and energy resource consumption) was due to the first part of this cycle, which occurs until the animal leaves the farm heading for the slaughterhouse.

Our choice of system boundary is, then, justified by this fact. We only go as far the animal farm gates, because up until that point is where most of the significant impact is. Besides, we are comparing meat produced by several different pasture systems. The systems are only different up until the animal leaves farm. Afterwards, we consider that they follow the same path to the plate[23].

1.9 Objectives and overview of the thesis

In this first Chapter, we defined the systems studied and the scenarios we use next. Considering the framework set by this analysis, our main objectives are the following:

- Determine the potential for SBPPRL to sequester carbon, showing that they are a carbon sink as a bovine production system;
- Determine whether SBPPRL provide other environmental services besides carbon sequestration;
- Connect research results with the PCF Project.

Our study is limited to one product, bovine meat, produced in three different systems of rainfed pastures (considering only marginal substitutions between them, so that no global large scale effects take place). Even though we are including indirect (life cycle) effects, the border of our analysis of the production chain is the selling of the animal to the slaughterhouse. From then on, all three options are similar. Throughout this work, and unless specifically noted, the functional unit of the analysis is 1 hectare of pasture. Throughout our study, we discuss as much as possible the effect of changes in assumptions in final results.

The remainder of the thesis is structured as follows.

We noted, while describing the system effects of SBPPRL in a causal relation scheme, that SOM is the key variable to understand their environmental and agronomic effects. It is, for example, by accumulating SOM that pastures retain carbon in the soil. Therefore, in the second Chapter we determine a model for SOM increases in SBPPRL and alternative natural grasslands, using field data. Because SOM is so important in the rest of this work, we focus in depth on the analysis and discussion on results in Chapter 2.

In the third Chapter, we begin by quantifying carbon sequestration in pastures. We compare that value with other results for other agricultural land uses. Then, we determine the carbon balance, including emission sources such as livestock, grasses and limestone, to study if the system as a whole is indeed a sink. Then, we turn to other environmental services, such as soil protection and biodiversity. We then broaden the scope of our analysis by calculating the life cycle impacts of the system. We end with a brief reference to irrigated SBPPRL, and the comparison with the possibility of biofuel production.

Then, having studied the environmental aspects of the system, in the fourth Chapter we turn to economic considerations. We study consumer valuation of quality products, and then focus on the possibility for payments for carbon sequestration in SBPPRL, which makes the connection from applied research to policy advisory. We study whether the PCF could have an advantage from paying carbon sequestered in pastures, and then proceed to describe the project which was actually submitted to and approved by the

[23] Note that this may not be completely accurate. Meat quality may be different depending on the way the animal was produced. And the quality of meat determines the stores where meat is sold, the way it is cooked and the quantity that is consumed. However, since we have no data on any of these factors, we had to neglect them.

PCF. To wrap up our work, we will propose a research plan to provide answers to open questions.

2. Soil organic matter dynamics in Portuguese pastures

In Chapter 2, the SBPPRL system is described and compared with natural pastures. The hypothesis for the present thesis is presented by formulating nine statements concerning the qualitative differences between the two options. We build and calibrate a model to study the short-term dynamics of soil organic matter (SOM), which is the key parameter to characterize the environmental and agronomic effects of pastures. SOM results obtained from the models show which grassland system provides higher SOM increases. We also study several methodological issues on the models used, like the sampling depth and assumptions regarding which factors influence the mineralization rate. Finally, we conclude on the average potential of SBPPRL to increase SOM concentration.

2.1 A primer on soil science

2.1.1 Soil structure and type

Soils are composed of a matrix, which is the solid component, consisting of organic and inorganic elements which form a porous system. The pores of this matrix are filled with water and gaseous substances, such as oxygen. Soils are characterized according to the characteristic of both the inorganic and the organic parts, as well as the percentage of each.

Regarding the inorganic part, soils are also characterized according to the rock that gave them origin, and also according to their texture. Texture is a measure of the size distribution of the solid particles. Some particles are relatively thick in diameter. But the smallest particles are those responsible for most chemical properties of soils (de Varennes, 2003). These small components are three:

- Sand – Particles with diameters between 2 and 0.02 mm;
- Lime – Particles with diameters between 0.02 and 0.002 mm;
- Clay – Particles with diameters smaller than 0.002 mm.

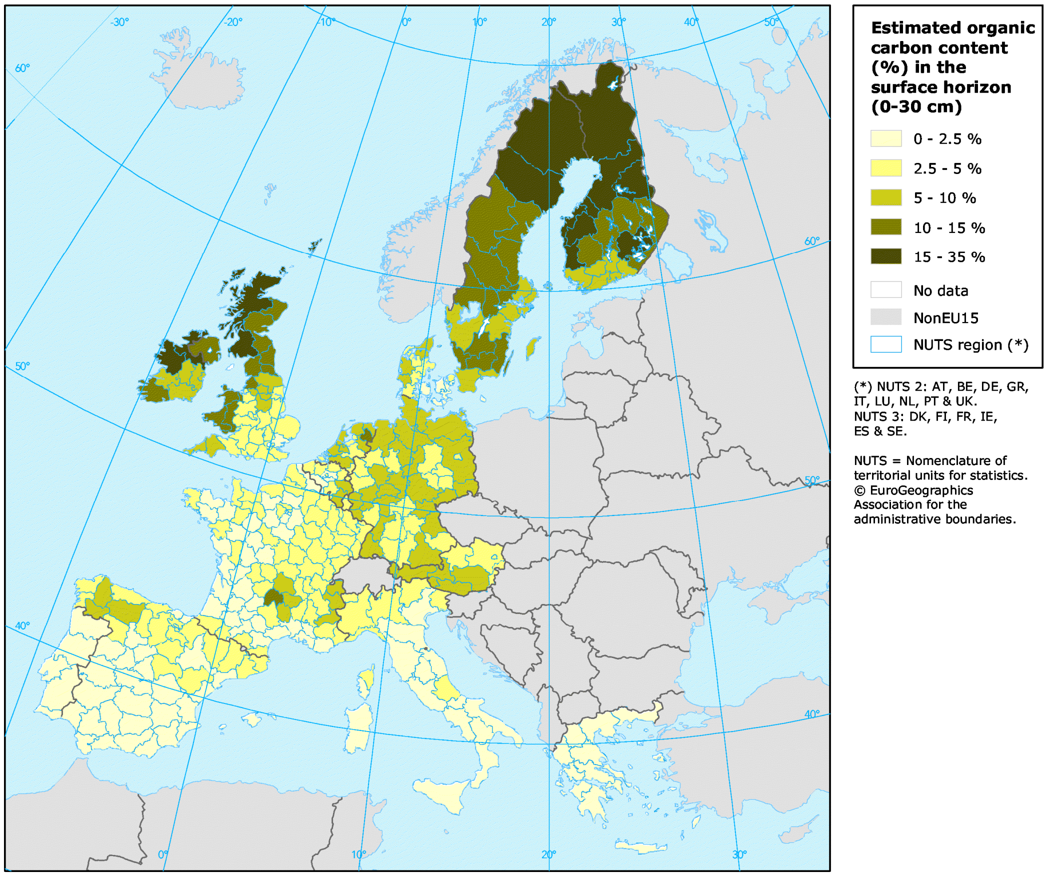
This is a particularly important classification for soils, since it translates how soils are aggregated, and hence also hydraulic characteristics such as field capacity, which is higher in clay soils (de Varennes, 2003).

Regarding the organic component of soils, note that this soil organic matter (henceforward called only SOM) is composed, as any type of organic matter, by elements such as nitrogen, carbon and hydrogen. Soils with a natural high SOM level are those where there is a slow decomposition of these organic elements in soils. This process, known as soil respiration or SOM mineralization, is executed by bacteria in aerobic conditions. In most conditions, mineralization is influenced by soil oxygen, soil moisture, pH, temperature, C:N ratio, type and abundance of clay minerals (Bot and Benites, 2005).

Therefore, low SOM mineralization rates occur in zones with less oxygen diffusion, such as damped areas, or cold regions (de Varennes, 2003). According to EEA (2004), 57.1% of Portugal's soils have low or very low SOM (between 0.5 and 2.0%), as shown in Figure 10. SOM in Europe is a characteristic of latitude, since it affects all

Mediterranean countries: Portugal, Spain, Italy and Greece, and even a very significant part of the French territory.

Figure 10 – European soil organic matter content (%) in the surface horizon (0-30cm).



Source: http://local.pt.eea.eu.int/.

The richness of the soil in terms of SOM is one of the most important parameters to determine the soil's quality. It is also the key to understand the ecosystem services provided by soils.

2.1.2 The key role of SOM

SOM is composed of living organisms (bacteria, fungi, plant roots and animals), dead animal and plant tissues in several stages of decomposition but still recognizable, and a complex mixture of decomposed, modified or reprocessed material called humus (which is usually 60 to 80% of all SOM) (Bot and Benites, 2005; de Varennes, 2003).

SOM quantity influences the quality[24] and resilience[25] of soils, since it determines the soil's physical and chemical proprieties. Humus is responsible in the most part for the aggregation of clay, and therefore is essential to keep soils stable. High SOM

[24] Quality of soils is defined, according to de Varennes (2003), as the capacity of soils to accept, retain and recycle water, nutrients and energy.

[25] Resilience of soils is defined, according to de Varennes (2003), as the capacity of soils to (almost) revert to the initial state after it has been disturbed (e.g. SOM returning to initial values after tillage).

concentration implies a high capacity for water and nutrient retention, and is therefore crucial in terms of plant productivity (Trumbore and Czimczik, 2008).. As more organic matter decomposes and enters the structure of the soil, more nutrients are being recycled and made available to plants[26] (de Varennes, 2003). We can now understand the environmental outcomes of these processes.

2.1.3 Environmental services provided by soils

The importance of soils in frequently underestimated. Soils are a resource which, despite all tries, is still hardly substitutable. Soils with high quality and resilience have mainly four functions, which represent environmental gains (de Varennes, 2003):

- To support plant growth, both mechanically and in terms of water and nutrient feed. Soils with high SOM levels are better for plants, which become more productive;
- To provide conditions for the recycling of both animal and plant residues and dead tissues, making their components available again in the ecosystem;
- To establish ecological niches for a fauna that goes from bacteria and fungi to small mammals, thus promoting biodiversity;
- To regulate the water cycle, both in terms of water flow quantity and quality, making water available for plants and at the same time regulating the outflow and thus preventing overflows in superficial waters due to high precipitation events.

Plant vegetation cycles are dependent of soils as a means to develop fixed roots with which they exchange heat and mass. Soils are relatively stable means, thus protecting the roots and seeds of high temperature variability. They also hold incoming water from precipitation and irrigation. This water is later provided to plants as they need it for use and transpiration. Soils also filter and withhold many substances which otherwise could be toxic to plants and downstream water masses (de Varennes, 2003).

In fact, diminishing water surface runoff and soil erosion have positive effects beyond the borders of the farm and soil type. Sediments, nutrients, organic matter and pesticides carried in water contribute to silting, eutrophication and contamination of superficial waters (EEA, 2004). These effects are known, but its true costs are still hard to pinpoint. Figure 11 shows the annual soil risk erosion by water in Europe, which is clearly higher in Mediterranean countries (including Portugal).

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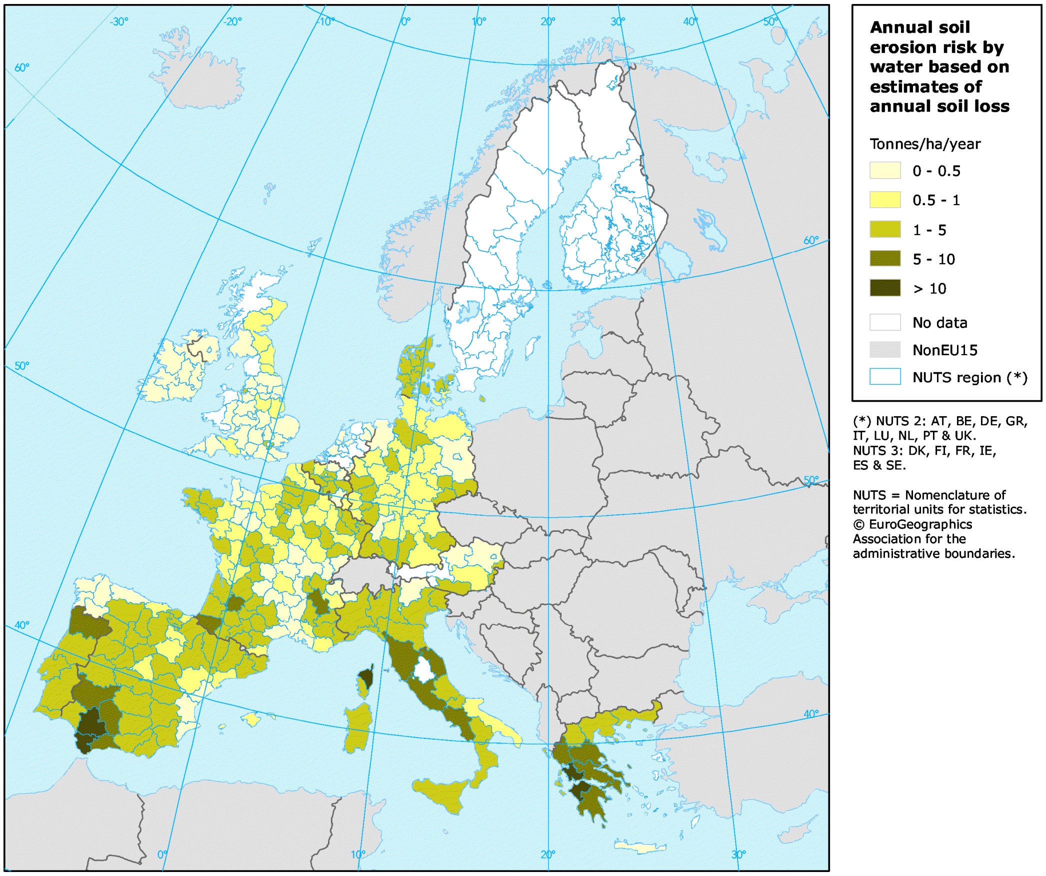
[26] Nitrogen, phosphorus and potassium are usually the major nutrients made available by soils with high SOM concentration, but there are other important ones, such as sulphur and carbon compounds.

Figure 11 – Annual soil erosion risk by water, based on estimates of annual soil loss.



Source: http://local.pt.eea.eu.int/.

But there is one effect which has not been mentioned yet. SOM is composed by the stoichiometric percentage of 58% of carbon. Since soil organic carbon (henceforth referred to as SOC) is mostly formed after decomposition of plant and animal residues, it is the result of atmospheric carbon (CO_2) sequestered by plants during photosynthesis[27]. At the same time, soil respiration, the process through which organic matter is decomposed, produces CO_2 (Trumbore and Czimczik, 2008).

2.2 Effects of pastures in the soil

As we have shown, SOM quantity influences the quality and resilience of soils, since humus is responsible in the most part for its aggregation (Bot and Benites, 2005). High SOM concentration contributes to many environmentally positive effects, as shown in the previous Chapter.

Pastures are one specific land use with widespread benefits to organic matter pools in soils. Pastures provide particularly high and stable SOC pools. Guo and Gifford (2002) indicate that SOC stocks decline after land use changes from pasture to cropland by 59%, and increase after land use changes from crop to pasture by 19%. Martens *et al.* (2004) show that for systems undisturbed for 130 years, pastures' soils had 25% more

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[27] Even in the case of the decomposition of animal material, somewhere in the trophic chain of the animal there was the primary consumption of plants, and therefore photosynthetic carbon.

carbon than cropped soils. Improvements such as fertilization and sowing in pastures increase SOM accumulation by increasing pasture productivity.

Therefore, pastures provide particularly high and stable carbon pools. But it is still necessary to determine exactly how SOM accumulates in pastures. Several authors have also tried to model SOM or soil organic carbon (SOC) in time. Six *et al.* (2002) found that an asymptotic curve depicts SOC variation with carbon input. Six *et al.* (2004) distinguish several SOM pools, based on their stability, and determine a saturation upper bound for all of them. Other, like West and Six (2007), have looked into soil carbon saturation. They argue that it is possible to increase SOC level by land management (e.g. no-tillage) and increased inputs (e.g. fertilizers), but only up to a given point. In time, SOC reaches an upper bound. This upper bound is that for which organic input into the soil and organic matter mineralization (soil respiration) are equal.

There is a significant quantity of studies that does point to temperature as the main parameter that controls soil respiration. Cao *et al.* (2004), for example, argue that grazing intensity also affects mineralization rate – low grazing intensity plots have higher soil respiration than high intensity ones. They claim that grazing increases the dependence of the mineralization rate on temperature. These results are supported by Raich and Tufekcioglu (2000), who have studied soil respiration rates for several land uses across several biomes. Their results show no significant differences between crop soil and either naked soils or grassland soils. Forest soils are the only ones with a slightly lower mineralization rate. What does seem to matter in terms of soil respiration is aboveground net primary productivity in grasslands, and meteorological factors such as temperature and precipitation. However, other studies such as Fitter *et al.* (1998) showed that soil respiration in grasslands depends on radiation fluxes, but is unrelated to temperature. They argue that temperature usually shows up as a significant parameter in short-term studies only.

What all studies seem to confirm is that pastures have a high potential for the accumulation of stable SOM. This accumulation is obviously limited by an upper bound. In pastures, if there are no land use conversions or other management activities, discounting for climate effects, SOC reaches a long term steady state equilibrium.

For Portugal, some early results hinted that in 10 years soils with Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL) increase SOM from 1 to 3% (Crespo, 2004). This value is confirmed by some empirical data. At Herdade dos Esquerdos, in Vaiamonte (Portalegre, Portugal), following a programme of SBPPRL installation, SOM concentration across the farm increased from between 0.7% and 1.2% in 1979 to between 1.45% and 4.40% in 2003 (Crespo *et al.*, 2004; Crespo, 2006a, 2006b). This SOM increase is predictably higher than that of any natural grassland under any form of management. However, we lack a systematic study that compares SOM increases for various types of pasture.

2.3 Determining SOM dynamics in pastures

For the rest of this section, we develop a model to determine the average trend of SOM concentration in NG, FNG and SBPPRL[28]. Our main objective is to determine a the average SOM accumulation potential in each grassland system. We hypothesise that the

[28] This Chapter is based on Teixeira *et al.* (2010c).

variation in SOM over time is the balance between SOM input and output in a plot. The implication of this model is that SOM asymptotically reaches a long-term equilibrium.

The dynamic parameters in this model are the SOM input and the mineralization rate. In order to estimate the values of these parameters, we calibrate the model statistically using field data. Data was collected from 2001 to 2005 in several locations in Portugal, during two demonstration projects. Project AGRO 87, "Sown biodiverse permanent pastures rich in legumes – a sustainable option for degraded land use" (Carneiro *et al.*, 2005) collected samples in six farms. At the same time, Project PAMAF 4073, which was continued as Project AGRO 71, "Recovery and improvement of Alentejo's degraded soils using grasslands" collected samples from two additional farms . We filled-in some missing data, since some samples were not collected.

We use two statistical methods for calibration: one where all parameters are specific for each grassland system, and one where there is only a specific SOM input. We then compare the dynamics of the three systems in 10 years. Finally, we validate the results obtained and draw some conclusions. In order to do so, we compare our results with other data and studies.

2.3.1 Characterization of the plots

Data was obtained from rainfed pastures in eight farms in Portugal from 2001 to 2005 (Table 5, location in Figure 12). Plot areas ranged from 5 to 15 ha. Each plot's soil and landscape type was approximately homogeneous, in terms of soil and previous use. These pastures were not isolated test sites. They were located in private land currently used by farmers for animal production. Prior to the beginning of the projects, plots were used in a system of long cereal/fallow rotations – one year of crop production for each five to seven years of fallow (which was used as a "natural pasture" featuring spontaneous herbaceous plants). In Farm #1 the NG plot was fertilized in 2002, and so the NG system was lost. In Farms #7 and #8 (Project Agro 71), FNG were not studied. Almost no samples were collected in 2002.

Table 5 – Soil and site characterization in the sites of Projects Agro 87 (farms 1 to 6) and Agro 71 (farms 7 and 8).

Farm No.	Farm	Location	Soil original material	Texture*
1	H. Cabeça Gorda	Vaiamonte	Gneiss	Loam
2	H. Mestre	São Vicente	Limestone	Loamy clay
3	H. Claros Montes	Pavia	Granite	Loamy sand
4	H. Refroias	Cercal	Schist	Loamy sand
5	H. Cinzeiro e Torre	Coruche	Sandstone	Sand
6	Quinta da França	Covilhã	Granite	Loamy sand
7	H. Monte da Achada	Castro Verde	Schist and Greywacke	Sandy loam
8	H. Corte Carrilho	Mértola	Schist	Loamy sand

*** The "feel" method was used. The textural class is ascertained by rubbing a sample of the soil in a moist to wet condition, between the thumb and fingers.**

Figure 12– Map of Portugal, with the indication of the sampling sites of Projects Agro 87 (farms 1 to 6) and Agro 71 (farms 7 and 8).

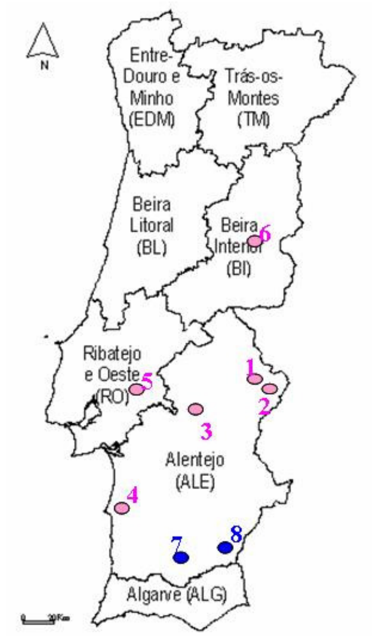

Table 6 shows the main meteorological characteristics of all test sites. Most sites are similar in their characteristics, except for farm #6, which is the only one in Central Portugal.

Table 6 – Main meteorological and texture characteristics of the sites of Projects Agro 87 (farms 1 to 6) and Agro 71 (farms 7 and 8).

Farm No.	Average daily temperature (ºC) [2]	Precipitation (yearly Oct-Sept, mm) [1]				
		2001-02	2002-03	2003-04	2004-05	2005-06
1	16.8	381	548	494	257	549
2	16.3	433	-	462	-	-
3	15.5	476	544	-	190	444
4	16.3	610	750	597	267	-
5	15.5	504	468	-	272	564
6	11.3	-	-	1195	624	-
7	15.5	475	458	478	207	546
8	16.8	451	477	480	-	-

Sources: [1] Actual data for the closest meteorological station in the SNIRH database (http://snirh.pt/). Empty cells are for years when a valid data series was not collected. [2] Portuguese Environmental Agency's Atlas of the Environment (APA, 2009); values are approximat mean figures for the period of 1930 to 1970.

Table 7 shows fertilization applied in SBPPRL and FNG in all sampling sites. Fertilization needs were determined according to the initial soil analysis of all fields. Both grassland systems were subjected to the same fertilization rates during all years of the project. The difference during the installation of SBPPRL is that, previous to sowing, plots are tilled in the upper layer of soil, and a phosphate and potassium fertilizer (superphosphate or 0:21:21) is used. Limestone is added in SBPPRL if soil pH is lower than 5.3 to lower acidity to optimum levels for legumes. Other micronutrients are added, like zinc sulphate copper sulphate or borax. Molybdate (a salt of molybdic acid) is added together with the seeds. SBPPRL were installed with 30 $kg.ha^{-1}$ of seeds.

Table 7 – Fertilization applied in SBPPRL and FNG in the sites of Projects Agro 87 (farms 1 to 6) and Agro 71 (farms 7 and 8).

Farm No.	Quantity applied ($kg.ha^{-1}$)				
	Limestone *	Superphosphate 18%		0:21:21	
		Sowing *	Maintenance	Sowing *	Maintenance
1	0	350	200	0	0
2	0	350	0	200	0
3	2000	0	350	0	200
4	2000	0	250	0	200
5	1000	0	0	450	200
6	2000	200	200	0	0
7	1000	0	100	300	200
8	0	1000	0	0	200

*** For SBPPRL only. SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.**

Table 8 shows the average yearly stocking rate in each grassland system (considering all farms). Farmers registered the number of animals put on each plot each day. That information was then averaged in a year (considering also the days when there were no animals in the pasture) (Carneiro *et al.*, 2005). Note also that the two types of natural pasture (NG and FNG) were obtained from the division in half of the same plot (part fertilized and part not) and were thus under the same livestock management. In 2004-05 figures are much lower because of a severe drought in Portugal. Nevertheless, Table 8 shows that SBPPRL support a higher stocking rate. Table 8 will be important in the Discussion to provide intuition for our results.

Table 8 – Average yearly stocking rate in SBPPRL and natural pastures (NG and FNG) in the sites of Projects Agro 87 (Carneiro *et al.*, 2005).

Year **	Average stocking rate ($LU.yr^{-1}$)	
	SBPPRL	Natural *
2001-2002	0.73	0.39
2002-2003	1.13	0.44
2003-2004	1.22	0.43
2004-2005	0.36	0.14

*** "Natural" refers to both NG and FNG, since both were managed with the same stocking rate.**
**** From October of one year to September of the next year.**

LU – Livestock Unit; NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; One LU is the equivalent of one adult cow (a steer corresponds to 0.6 LU and an ewe to 0.15 LU).

2.3.2 SOM data

SOM determination begins with collection of soil samples. One composite sample was collected in each plot. Each composite sample was obtained from the mix of a variable number os sub-samples collected throughout each homogeneous plot, in order to be representative of the average SOM in the plot. The samples were collected, and then analysed, by Laboratório Químico-Agrícola Rebelo da Silva (LQARS), which is the Portuguese Government's official soil laboratory.

In the preparation of the sample for analysis, any living organisms, as well as gross animal and plant material, are removed from the sample. Therefore, from the general composition of SOM indicated in section 2.1.2 (Bot and Benites, 2005), results in this section only refer to the 60-80% which are humus and some minor organic material.

Sample laboratorial analysis begins with the entire sample being spread on a tray and dried overnight, at 35-37ºC. The sample was crumbled mechanically and passed through a 2mm stainless steel sieve. The sieved material is the 'fine soil' subject to analysis. Samples were then analysed for several parameters, such as pH, nitrogen, phosphate and potassium levels, lime requirement and SOM concentration. It was the latter parameter that we used in this work.

From 2001 to 2004, results for SOM concentration were obtained by the wet oxidation method. This method consists in the digestion of organic carbon by sodium dichromate, followed by colorimetric determination on a molecular absorption spectrophotometer at 640 nm (Carter, 1993). From 2005 on, a dry combustion method was used (including for samples collected in 2005). It consists of the determination of total carbon by dry combustion, according to ISO Standard 10694, using a CNS elemental analyzer (Rodeghiero *et al.*, 2009). Organic carbon is determined indirectly after correction of the total carbon concentration for the carbonates present in the soil sample. Both methods measure SOC concentration, which is them multiplied by 1.724 (assuming that 58% of organic matter is carbon) to obtain the corresponding SOM concentration. Since two different methods were used, an internal (unpublished) study was conducted by LQARS to obtain an equivalence factor between results. This study guarantees that the results are consistent. Final results for SOM are presented in mass percentage (%) units, equal to grams of SOM per 100 grams of soil.

There are few results available for 2002, in a set obtained from 2001 to 2005. Since our regression models (whose equations we show in the next section) use pairs of points $(SOM_{i,t-1}, SOM_{i,t})$, two pairs are almost always missing: (2001, 2002) and (2002, 2003). If these values are filled-in, we end up doubling the number of observations. Furthermore, FNG are missing more pairs of points than SBPPRL and NG. Therefore, whenever there is a missing value between two others, we calculate the geometric average (the "growth rate" of SOM) of the two observations. For example, assuming that SOM_{t-1} is missing, while SOM_{t-2} and SOM_t are not, we calculate the missing value as:

$$SOM_{t-1} = SOM_{t-2} \cdot \left(\frac{SOM_t}{SOM_{t-2}} \right)^{1/2}. \tag{1.3}$$

2.3.3 SOM dynamic model

As the thorough review done by Pete Falloon and Pete Smith (2009) shows, other models in the literature intend to explain inter and intra annual variability in SOM. To explain such variability, they are required to use environmental variables, such as climate and soil type. Out of the 33 models reviewed and assessed by Falloon and Smith, only one (O'Brien, 1984) had a yearly time step and no meteorological and management variables. But the O'Brien model considered interactions with plants, and had a completely different objective than ours. However, in our case, it is the SOM trend we wish to estimate and not the interannual variation of SOM levels. Our objective is to calibrate a time series to capture the trend of SOM dynamics in the three grassland systems.

Therefore, we used a simple mass balance model for SOM dynamics, calibrated using field data. The model states that the mass percent balance of SOM is the difference between input and mineralization:

$$\frac{dSOM_t}{dt} = K - \alpha \cdot SOM_t , \tag{1.4}$$

where SOM is the SOM concentration (percentage points, equal to $g_{SOM}.100\ g_{soil}^{-1}$) at time t, K is the SOM input, and α is the organic matter mineralization rate.

We solve Equation (1.4) by integrating it between $t-\Delta t$ and t:

$$SOM_t = \frac{K}{\alpha}\left(1-e^{-\alpha\Delta t}\right)+e^{-\alpha\Delta t}SOM_{t-\Delta t} . \tag{1.5}$$

Therefore, the general solution for Equation (1.4) has an asymptotic exponential form. This means that SOM accumulation is limited by an upper bound. In pastures, if there are no land use conversions or other management activities, disregarding climate effects, SOM reaches a long-term equilibrium.

As the inspection of the SOM analysis results in Table 11 will show, farms with high initial SOM still increased their SOM concentration by a relatively high percentage, regardless of the pasture type. To capture this effect, we separate K in a fixed term $K^{'}$ (which is a function of grassland system and not of representative local conditions), and a variable part (which is a linear function of the initial SOM concentration), being the proportionality parameter a):

$$K = K' + a \cdot SOM_0 . \tag{1.6}$$

Equation (1.6) shows that we use initial SOM as a proxy for representative conditions of the location. This approach is justified by the fact that natural soil and climate conditions, as well as the history of the field, determine the initial SOM concentration. This slightly changes the model. Substituting Equation (1.6) in Equation (1.5), we obtain the general expression of the model:

$$SOM_t = \frac{K'}{\alpha}\left(1-e^{-\alpha\Delta t}\right)+\frac{a}{\alpha}\left(1-e^{-\alpha\Delta t}\right)SOM_0+e^{-\alpha\Delta t}SOM_{t-\Delta t} . \tag{1.7}$$

There are now two alternative approaches to estimate the parameters in Equations (1.5) and (1.7):

1. In the "specific-data model", we consider that all parameters are a function of pasture type ($K_i^{'}, \alpha_i$ and a_i, where $i=\{SBPPRL, FNG, NG\}$). It is necessary to estimate one model per grassland system, obtaining three sets of three parameters each one specific of the grassland type.
2. In the "pooled-data model", we consider that only specific SOM input is a function of pasture type ($K_i^{'}$). It is possible to estimate one single model using all data for all types of pastures, obtaining one set of five parameters. In this

case, we are assuming that the mineralization rate is equal for the three grassland systems.

The difference in statistical procedure is summed up in Table 9. Both models are calibrated estimating a regression equation in which SOM_t is the dependent variable and SOM_{t-1} and SOM_0 are the independent variables. In the specific-data model, we use data from all locations but separately for each grassland system *i*. There are three different regressions, and the nine dynamic parameters are calculated from the nine regression constants. In the pooled-data model, only the constant term has a parameter that depends on the grassland system. We thus consider that the constant term is the sum of three dummy variables, one for each grassland system. The dummy $d_i = 1$ if observation regards grassland system *i* and $d_i = 0$ otherwise[29].

Table 9 – Presentation of the two different approaches to estimate the model for SOM dynamics.

<table>
<tr><td colspan="2">Parameters</td></tr>
<tr><td>“Specific-data model”</td><td>“Pooled-data model”</td></tr>
<tr><td>$K_i^{'} = f(i)$
$\alpha_i = f(i)$
$a_i = f(i)$
$i = \{SBPPRL, FNG, NG\}$</td><td>$K_i^{'} = f(i)$
α
a
$i = \{SBPPRL, FNG, NG\}$</td></tr>
<tr><td colspan="2">Model</td></tr>
<tr><td>$SOM_{i,t} = \frac{K_i^{'}}{\alpha_i}\left(1-e^{-\alpha_i \Delta t}\right) + \\ + \frac{a_i}{\alpha_i}\left(1-e^{-\alpha_i \Delta t}\right) SOM_{i,0} + \\ + e^{-\alpha_i \Delta t} SOM_{i,t-\Delta t}$</td><td>$SOM_{i,t} = \frac{K_i^{'}}{\alpha}\left(1-e^{-\alpha \Delta t}\right) + \\ + \frac{a}{\alpha}\left(1-e^{-\alpha \Delta t}\right) SOM_{i,0} + \\ + e^{-\alpha \Delta t} SOM_{i,t-1}$</td></tr>
<tr><td colspan="2">Regression Equations</td></tr>
<tr><td>$SOM_{i,t} = C_{i,1} + C_{i,2} \cdot SOM_{i,0} + C_{i,3} \cdot SOM_{i,t-1}$
(3 separate sets of data and 3 separate regressions, one for each *i*)</td><td>$SOM_t = \sum_i \theta_i d_i + C_2 \cdot SOM_0 + C_3 \cdot SOM_{t-1}$
(1 set of data and 1 single regression, for all *i*)</td></tr>
<tr><td colspan="2">Obtaining parameters from regression constants</td></tr>
<tr><td>$\begin{cases} \alpha_i = \frac{-ln\left(C_{i,3}\right)}{\Delta t} \\ K_i^{'} = \frac{C_{i,1} \cdot \alpha_i}{1-e^{-\alpha_i \Delta t}} \\ a_i = \frac{C_{i,2} \cdot \alpha_i}{1-e^{-\alpha_i \Delta t}} \end{cases}$ (9 parameters)</td><td>$\begin{cases} \alpha = \frac{-ln\left(C_3\right)}{\Delta t} \\ K_i^{'} = \frac{\theta_i \cdot \alpha}{1-e^{-\alpha \Delta t}} \\ a = \frac{C_2 \cdot \alpha}{1-e^{-\alpha \Delta t}} \end{cases}$ (5 parameters)</td></tr>
</table>

SOM – Soil Organic Matter; $K^{'}$ - specific SOM input; α - mineralization rate; a – contribution of initial plot conditions for SOM increase; C – regression constant; i – grassland system; NG –

[29] This version of the model, and corresponding results, were published in Teixeira *et al.* (2008a, 2010a).

Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.

SBPPRL plots are tilled in the first year, and thus there is increased SOM mineralization for t = 1. Besides, in the first year plants blossom only from the seeds which were sown. It is only from the second year on that a seed bank is created (much larger than the amount of seeds initially sown) from which the pasture permanently blossoms every year. Less seeds imply a lower quantity of biomass produced in the first year, unlike in the following years.

Therefore, we also test the hypothesis that SOM dynamics is different in the first year. In order to do so, we estimate the same regression in Equation (1.7), but consider one more dummy variable, namely to separate results from SBPPRL in the first year. For simplicity in the calculation, we introduce the dummy in the parameter *K'*. The regression equation for the specific-data model for SBPPRL, in Table 9, thus becomes:

$$SOM_t = \sum_j \theta_j d_j + C_2 \cdot SOM_{t-1} + C_3 \cdot SOM_0, \qquad (1.8)$$

where $j = \{SBPPRL(t=1), SBPPRL(t>1)\}$. Note that, for FNG and NG, the model is the same as before, and is still represented by the regression equation in Table 9. The pooled-data model for all grassland systems in Table 9 is now represented for all $k = \{SBPPRL(t=1), SBPPRL(t>1), FNG, NG\}$ by

$$SOM_t = \sum_k \theta_k d_k + C_2 \cdot SOM_{t-1} + C_3 \cdot SOM_0, \qquad (1.9)$$

where d is a dummy variable equal to one for observations of grassland system k and zero otherwise, and θ is the respective regression coefficient.

We estimate the parameters of each model using software SPSS Statistics 17.0 with an Ordinary Least Squares (OLS) method (Verbeek, 2001).

We considered other specifications of the model, namely:

- Logarithmic filling-in of data;
- Using soil and precipitation variables instead of the initial SOM concentration;
- Linear approximation of the model.

All of these options were rejected, but results are shown in Appendix I – Alternative estimations of the SOM model.

2.3.4 Application and validation of the SOM model

The procedures referred in this section are a simplified approach intended only for validation of some parameters obtained.

To determine the grassland system in which the increases in SOM was highest, we calculate SOM increases in all systems starting from the same arbitrary initial SOM. Since we need a scenario for initial SOM, we assumed a starting hypothetical concentration of 0.87%, which is believed to be representative of Portuguese soils in the

Alentejo region[30]. We also apply the model to the initial SOM concentrations measured in each farm. This procedure tests the validity of the model by comparing measured and calculated results.

All results in this section are shown in percentage points, equal to $g_{SOM}.100\ g_{soil}^{-1}$. However, for comparison purposes, we need to determine the carbon equivalent to SOM increases. Stated in another way, we find the equivalent to 1% SOM in terms of t $C.ha^{-1}$. We begin by using the Adams (1973) equation to correct mineral soil density:

$$BD = \frac{100}{\frac{SOM}{0.244\,g.cm^{-3}} + \frac{100 - SOM}{MBD}}, \qquad (1.10)$$

where BD is soil bulk density ($g.cm^{-3}$), MBD is soil mineral bulk density ($g.cm^{-3}$), SOM is SOM concentration (percentage point, equal to $g_{SOM}.100\ g_{soil}^{-1}$), and 0.244 $g.cm^{-3}$ is the MBD for which MBD = BD regardless of the SOM concentration.

Based on Rawls and Brakensiek (1985), MBD of mineral soils varies between 1.20 and 1.69 $g.cm^{-3}$. The indicative MBD in Portugal is 1.25 $g.cm^{-3}$, but soils which are not tilled are more compact, and thus MBD is around 1.40 $g.cm^{-3}$ (Mário Carvalho, personal communication). In the following calculations we will use both.

Results are shown in Table 10. Starting from the MBD values, we consider 1% SOM and calculate BD. Since the soil samples were collected at up to 10 cm, the SOM mass per unit is subsequently determined per unit area. Final values are obtained by converting to tons per hectare.

Table 10 - Carbon sequestration equivalent to the increase in SOM of 1 pp in 10 cm.

MBD ($g.cm^{-3}$)	SOM (pp)	BD ($g.cm^{-3}$)	g(SOM).cm^{-3}	Depth (cm)	g(SOM).cm^{-2}	t (SOM).ha^{-1}	t C.ha^{-1}
1.25	1	1.20	0.0120	10	0.120	12.0	6.96
1.40	1	1.34	0.0134	10	0.134	13.4	7.77

MBD – Mineral Bulk Density, SOM – Soil Organic Matter, BD – Bulk Density, pp – percentage point.

We use Table 10 to convert *K'* from % (in mass) per unit of time into t(SOM).ha^{-1}, which is then converted to equivalent plant production. In order to do so, we assume that only humus is captured in SOM analysis, which is at most 80% of belowground biomass in pastures (Bot and Benites, 2005). Furthermore, the IPCC (1997) indicates 2.8 as the default root to shoot ratio (R:S) for semi-arid grasslands. This value is consistent with the R:S of 0.5 to 4.8 in grazed pastures, which is the range of the comprehensive data for several regions gathered by Coupland (1976). Dividing *K'* by 80% and then by R:S, we obtain an estimate of aboveground production.

The final values in Table 10 may also be used to determine the equivalent of SOM increases in terms of carbon. In order to do so, we had to assume a conversion factor between SOM and soil organic carbon (SOC). Since approximately 58% of SOM is SOC (IPCC, 1997 and 2003), and both are measured as $g.100\ g_{soil}^{-1}$, then the mass of SOC is also 58% the mass of SOM. Therefore, 1% of SOM increase corresponds to the

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[30] This value was obtained as the average SOM concentration in the pasture plots of a systematic grid where soil samples were collected (Fátima Calouro, personal communication). Results are unpublished.

sequestration of 6.96 – 7.77 t C.ha^{-1}. These values are used when comparing our results to other studies.

2.4 Results of the calibration of the SOM model

2.4.1 Results from soil analyses

Results from soil analyses for SOM concentration are shown in Table 11 (Carneiro *et al.*, 2005). Considering the difference between the first and the last year, the minimum SOM increase for SBPPRL was obtained in Coruche (Farm #5), where the pasture blooming after the first year establishment was poor. The highest increases for SBPPRL were obtained in the more productive gneiss soil in Vaiamonte and the schist soils of Herdade de Refróias. In Herdade de Refróias, SOM at the beginning was already 3%. Farms #7 and #8 increased SOM concentration in SBPPRL by 0.35 pp and 0.43 pp per year. Table 11 also shows filled-in data underlined. Direct comparison with natural (non-fertilized) grasslands at each site shows that increases are usually higher for SBPPRL.

Table 11 – SOM concentration in each grassland system for experimental sites (0-10 cm).

Farm No.	Grassland system	SOM (%)				
		2001	2002	2003	2004	2005
1	SBPPRL	1.55	2.17	3.05	3.60	3.80
1	FNG	1.30	1.84	2.60	3.40	3.00
2	SBPPRL	1.75	2.15	2.65	2.70	5.40
2	FNG	1.95	2.42	3.00	4.50	3.50
2	NG	1.95	2.29	2.70	4.00	4.00
3	SBPPRL	0.45	0.73	1.20	1.63	1.60
3	FNG	0.68	0.86	1.10	1.40	2.00
3	NG	0.92	1.01	1.10	1.20	1.15
4	SBPPRL	3.40	3.08	5.10	4.60	5.60
4	FNG	3.80	4.23	4.70	5.40	5.60
4	NG	3.80	4.23	4.70	5.60	-
5	SBPPRL	0.65	0.81	1.00	1.28	1.50
5	FNG	0.55	0.78	1.10	1.15	1.25
5	NG	0.55	0.61	0.68	0.75	0.55
6	SBPPRL	1.82	2.09	2.40	2.18	2.70
6	FNG	1.75	2.25	2.90	2.70	2.70
6	NG	1.75	2.33	3.10	2.40	-
7	SBPPRL	0.55	0.83	1.14	1.60	-
7	NG	1.10	1.20	1.20	1.33	-
8	SBPPRL	0.80	1.40	1.54	2.08	-
8	NG	0.84	1.06	1.10	1.45	-

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; missing data was filled in using the geometric average of the increase rates, and is shown underlined; when result is "-", data could not be filled-in.

Table 12 shows averages for SOM concentration in each grassland system and each year (using only original data). In 2005, FNG have SOM concentrations similar to SBPPRL, but they also start from higher initial SOM concentrations. We can also see that even though the drought in 2005 is shown in stocking rate averages, SOM did not decrease significantly. This means that the lack of water decreased plant production, and therefore also SOM input, but it also decreased SOM mineralization.

Table 12 – Average SOM concentration in each grassland system and year (samples taken in Autumn).

Grassland system	SOM concentration per year and grassland system (%)				
	2001	2002	2003	2004	2005
SBPPRL	1.50	1.51	2.26	2.46	3.43
FNG	1.87	-	2.57	3.09	3.01
NG	1.67	1.13	2.32	2.39	1.90

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

All plots were natural pastures before the beginning of the project. Results for SOM in 2001 are previous to the installation of SBPPRL and to the fertilization of FNG making them representative of the initial soil conditions. Therefore, SOM initial value in each location is equal to SOM in the first year, 2001:

$$SOM_0 = SOM_{2001}. \tag{1.11}$$

The use of SOM_0 as a proxy for representative local conditions (soil parameters, climate conditions, former management), and the use of the first year as the initial SOM concentration, are both justified by visual inspection of results in Table 11 and by statistical results shown in Table 13. SOM_{2001} changes more between farms than between grassland systems within the same farm.

Table 13 shows that the average of all observations for SOM_0, considering all farms and grassland systems (calculated without the filled-in values), is 1.66 %, with a standard deviation of 1.06 pp (n = 18). If we first calculate the average SOM_0 in each of the seven farms, and only then calculate the overall average, the result will be the same, namely 1.66% (average weighted by the number of observations in each farm, which are either 2 or 3). But the standard deviation is much lower, 0.14 pp.

This means that in each farm (average of results for all grassland systems) the standard deviation of the average SOM_0 varies much less than in the overall sample (average of all observations for SOM_0). But when we calculate the average $SOM_{i,0}$ for each grassland system *i*, then we obtain standard deviations similar to that of the whole sample. This fact supports the assessment made from the results in Table 11: the initial SOM concentration is correlated with the farm but not with the type of pasture. It is, therefore, a logical choice for proxy for the specific conditions of a plot in the model.

Table 13 – Average and standard deviation for initial SOM concentration in each farm and grassland system.

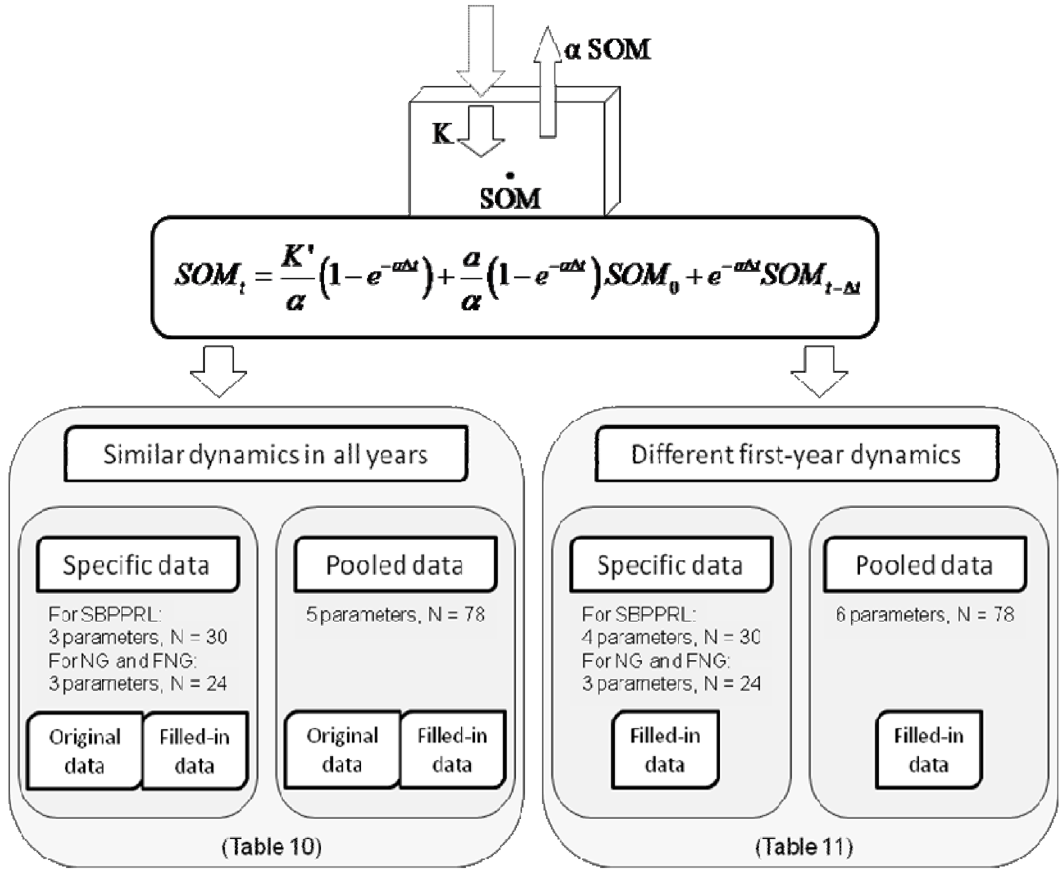
Farm	Grassland system	Initial SOM concentration (%)	Average (%)	Standard deviation (pp)	N (#)
1	SBPPRL	1.55	1.43	0.18	2
	FNG	1.30			
2	SBPPRL	1.75	1.88	0.12	3
	FNG	1.95			
	NG	1.95			
4	SBPPRL	3.40	3.67	0.23	3
	FNG	3.80			
	NG	3.80			
5	SBPPRL	0.65	0.58	0.06	3
	FNG	0.55			
	NG	0.55			
6	SBPPRL	1.82	1.77	0.04	3
	FNG	1.75			
	NG	1.75			
7	SBPPRL	0.55	0.83	0.39	2
	NG	1.10			
8	SBPPRL	0.80	0.82	0.03	2
	NG	0.84			
Average of farms		-	1.66	0.14	7
All observations		-	1.66	1.06	18
Average for SBPPRL		-	1.50	0.99	7
Average for FNG		-	1.87	1.21	5
Average for NG		-	1.67	1.17	6

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

2.4.2 Regression results

Absolute values used in the regression to calibrate the parameters of the model were those in Table 11 (percentage values, equal to $g_{SOM}.100\ g_{soil}^{-1}$). Regression results are shown in Table 14 and in Table 15. The difference between the two tables is the estimation procedure of the model, which is resumed in Figure 13.

Figure 13 – Different estimation procedures used for the SOM dynamic model.



***K’*, *K*, α, *a* – Parameters in Equation (5); N – number of observations.**

2.4.2.1. <u>Results assuming similar dynamics in the first and following years</u>

We begin our analysis by Table 14. Results show that the statistical fit of the models to base data, measured with the adjusted-R^2, is higher for the pooled-data model. The root Mean Standard Error (rMSE), however, is sometimes lower for specific-data models.

Regarding the relative order of parameters for each grassland system, inspection of Table 14 shows lower *K*’ and lower α for NG. A low α is consistent with the fact that the previous management in NG was maintained, and therefore the SOM pool is stabilized. Note however that negative *K’* in the specific-data model for NG, obtained using the initial SOM concentration, does not denote a negative SOM input. The total SOM input is *K*, equal to the fixed term *K’* plus the variable term which depends on the initial SOM concentration. Therefore, the sum of both is positive for initial SOM higher than 0.11%, which is the case in every test plot (and in virtually all Portuguese agricultural land).

Table 14 also shows parameters *K’* and *a* are higher for FNG as compared to SBPPRL in equivalent specific-data models. But field evidence on stocking rates in Table 8 states otherwise. Since stocking rates are always higher for SBPPRL plots, *K* should be higher for SBPPRL (assuming that there is a direct relation between stocking rate and biomass production). Note that the overall SOM balance is still favourable to SBPPRL because the mineralization rate is also higher for FNG.

Table 14 also shows that the use of filled-in data slightly improves the statistical fit of the models. Its conclusions, however, are qualitatively similar to those obtained using only original data. This indicates that filling-in does not bias results. This is due to the fact that increases between consecutive years in the method chosen for filling-in are calculated in a way which is qualitatively similar to the model (decreasing growth rate).

Therefore, we can use filled-in data to calculate values using the first-year dummy for SBPPRL. The results are shown in Table 15. We would not be able to use this dummy variable using only original data, since in 2002, which corresponds to SOM changes in the first year of settlement, there are only three plots (out of eight) with measured data for SBPPRL (Table 11).

2.4.2.2. Results assuming different dynamics in the first year for SBPPRL

Comparing Table 15 and Table 14, we see that the use of the first-year dummy significantly improves the adjusted R^2 of the estimation for the SBPPRL specific-data model (0.951 instead of 0.794). The adjusted R^2 of the estimations are equal for both models (0.970 and 0.969), while the rMSE for the pooled-data model is slightly lower (0.477 against 0.489).

Table 15 also shows a higher SOM input in SBPPRL than in FNG in specific-data models, using the first-year dummy. Parameters *K'* and *a* are higher for SBPPRL with $t > 1$. Unlike Table 14, this result is now consistent with field observations in Table 8, which has the stocking rates in each plot: SBPPRL produce more biomass, and thus support a stocking rate which is systematically twice or more that of natural pastures. However, while *K'* is significantly higher for SBPPRL than for FNG, *a* is very similar in the two systems, and lower for NG. This means that SOM pools respond to fertilization in a similar way (results for *a*), and the effect of sowing is only captured by the specific SOM input (*K'*). This result is supported by the fact that the seed mixes were designed specifically for each plot, and thus maximize SOM input regardless of baseline natural conditions. The difference in effect between both forms of input is another validation of the use of the initial SOM concentration as a variable in the model.

Regarding the absolute values of $K_i^{'}$ in Table 15, we may notice, for example, that $K_i^{'}$ in the pooled-data model ($t > 1$) is equal to 0.604 $\%.yr^{-1}$, which using Table 10 is equivalent to an input of 7.25-8.09 $t.ha^{-1}$ of SOM. Now, we need to transform this value of *K* into equivalent production. Dividing *K* by 80% and then by the average R:S of 2.8, we find that aboveground production is 3.2 – 3.6 $t.ha^{-1}$. Using two extreme R:S, aboveground production would be 1.9 – 2.1 $t.ha^{-1}$ or 18.1 – 20.2 $t.ha^{-1}$ The average production falls within the range of dry matter productivity of SBPPRL of 2 to 9 $t.ha^{-1}$ of dry matter (Carneiro *et al*. 2005).

The mineralization rates obtained in specific-data models for SBPPRL and FNG are similar, and are both higher than for NG. Increased SOM inputs due to the installation of SBPPRL and fertilization in FNG provide the soil labile forms of SOM, which are easily mineralized. Since there was no land use change or management activity during the sampling period in NG, their SOM pool remains stable.

Table 14 – Results of the estimation for pooled-data and specific-data models for grassland systems *i*.

Model	Grassland system	Using SOM_0?	Original data							Data with filled-in missing values						
			Adj. R^2	rMSE	*K'*			α	*a*	Adj. R^2	rMSE	*K'*			α	*a*
					SBPPRL	FNG	NG					SBPPRL	FNG	NG		
Pooled-data	All	No	0.952	0.677	0.422	0.171	0.136	-0.017		0.966	0.554	0.370	0.248	0.165	-0.034	
		Yes	0.956	0.659	0.500	0.303	0.128	0.403	0.630	0.969	0.489	0.415	0.289	0.109	0.267	0.430
Specific-data	SBPPRL	No	0.760	0.745	0.413			-0.020		0.794	0.649	0.353			-0.042	
		Yes	0.731	0.781	0.531			0.237	0.348	0.794	0.585	0.379			0.151	0.276
	FNG	No	0.810	0.622		0.428		0.071		0.886	0.470		0.442		0.043	
		Yes	0.841	0.603		1.083		1.105	1.434	0.912	0.397		0.508		0.443	0.566
	NG	No	0.899	0.615			-0.034	-0.105		0.935	0.487			0.011	-0.113	
		Yes	0.920	0.549			-0.282	0.512	1.073	0.943	0.438			-0.048	0.190	0.432

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; *K'*, α, *a* – Parameters in Equation (1.7); rMSE – root Mean Squared Error.

Table 15 – Results of the estimation for pooled-data and specific-data models for grassland systems *i*, including a first-year dummy in SBPPRL.

Model	Grassland system	Using SOM_0?	Filled data							
			Adj. R^2	rMSE	*K'*				α	*a*
					SBPPRL		FNG	NG		
					t=1	t>1				
Pooled-data	All	Yes	0.970	0.477	0.097	0.604	0.349	0.130	0.365	0.544
Specific-data	SBPPRL		0.951	0.610	0.135	0.697			0.429	0.588
	FNG		0.912	0.397			0.508		0.443	0.566
	NG		0.943	0.438				-0.048	0.190	0.432

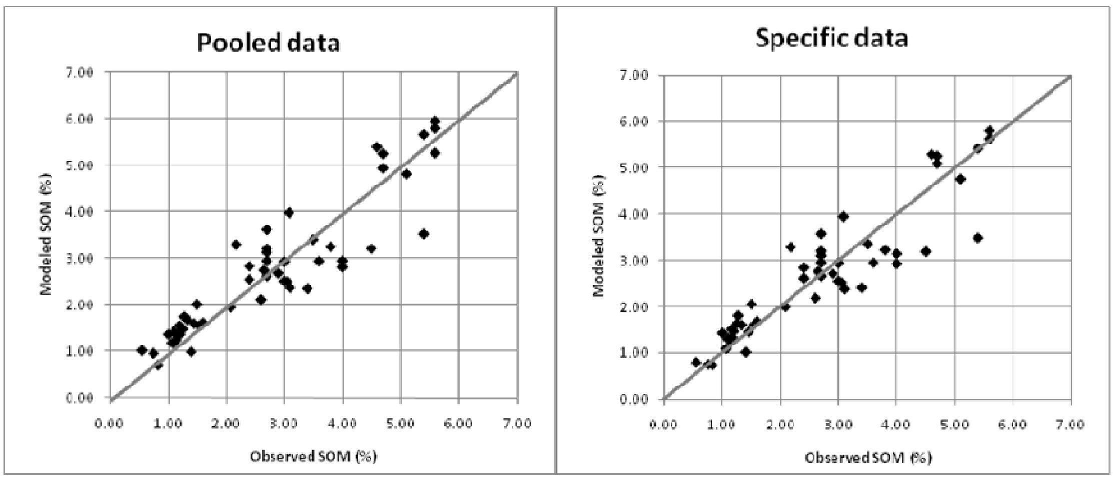
NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; *K'*, α, *a* – Parameters in Equation (1.7); rMSE – root Mean Squared Error.

2.4.3 Assessment of model quality

In order to verify the adjustment to the original data provided by the model, we applied it to each farm. We used parameters indicated in Table 16, and adjusted a model to each plot in each farm, using the plot-specific initial SOM concentration. The initial SOM concentration in each plot corresponds to the first column of data in Table 11.

Results are shown in Figure 14, which plots all observed and modelled results, except points corresponding to initial SOM concentration (which are by construction over the grey line) and except points which were filled-in. The closer that the points are to the grey 45º line, the better the fit.

Figure 14 – Observed and simulated SOM concentration for all farms and grassland systems, using an analytical model (on the left) and the linear approximation (on the right).



SOM – Soil Organic Matter.

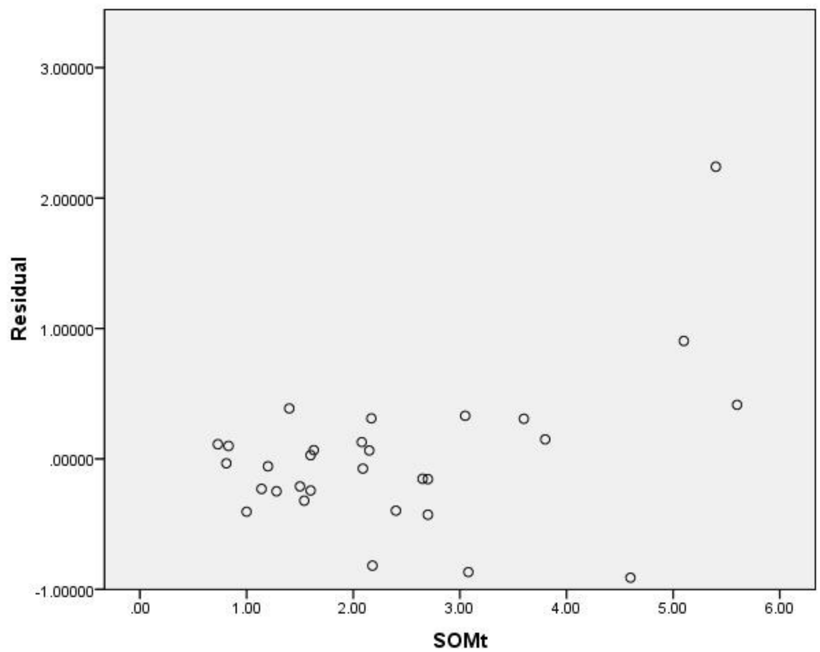
Visual inspection of Figure 14 does not show significant differences between the two models. However, in both cases, even though there seems to be no overall systematic bias, for SOM lower than 2.0 %, models seem to overestimate the values observed. In order to verify these hypotheses, namely that (1) there is no overall model bias and (2) there is a slight bias for SOM lower than 2.0%, we obtained two series, one for the results obtained with pooled data, and another for specific data. The series is calculated as the difference between modelled SOM and observed SOM.

A paired samples t-test for means of the two series (one for the results of each model) shows no evidence to reject the null hypothesis of equal means ($p < 0.05$). Therefore, the two models are equivalent, even though there is a slight tendency for the specific-data model to overestimate results. A paired samples t-tests comparing the respective series for each model with a series where all observations are zero also indicates that the means of both series cannot be rejected to be equal to zero ($p < 0.05$), and therefore there is no overall bias of any model.

To show that the bias does exist for SOM lower than 2.0%, we truncated the series data, separating each series into two more series: one for observations with SOM lower than 2.0%, and another one for SOM higher than 2.0%. Results were now different. The conclusions for observations with SOM higher than 2.0% are equal to the overall conclusions, and there is no bias ($p < 0.05$). However, in any model, we can reject that the series with observations with SOM lower than 2.0% has a zero mean value ($p > 0.05$). Models systematically overestimate lower SOM. This result, which is true for all pasture systems, means that the initial dynamics is not well captured by the asymptotic curve that we model. However, the deviation is relatively small. In fact, even though there is no bias, as we can see in Figure 14, variance is higher for higher SOM.

An alternative representation to Figure 14 may be found in Figure 15, which depicts the series of residuals (difference between estimated and observed values) as a function of SOM. For illustration purposes, only results for the specific model for SBPPRL are shown. A Phillips-Perron Test rejects the null hypothesis of a unit root ($p < 0.05$). This means that the series of residuals is stationary.

Figure 15 – Series of residuals as a function of SOM.



SOM – Soil Organic Matter.

Furthermore, a one series Kolmogorov-Smirnov Test of normality to the series of residuals shows that the null hypothesis of the normal distribution cannot be rejected (K-S Z value 0.928, asymptotic sigma 0.356). Therefore, there is no evidence of a spurious regression due to a bias in residuals.

2.4.4 Projections of average SOM increases and carbon flows

For illustration purposes, and due to the results stated before, we will use the parameters from the pooled and specific-data models, including SOM_0, the complete filled-in data table and using a first-year dummy for SBPPRL.

Table 16 shows results for K, which is a measure of the total input per year. In both formulations of the model, K is higher for SBPPRL than for FNG, which translates to higher SOM increases since the fraction of existing SOM in one year which is mineralized in the next, $1-e^{-\alpha}$, is constant in the pooled-data model and approximately equal in the specific-data model. Table 16 also shows that SBPPRL in the first-year have a much lower (but still positive) specific input than in the following years, but their total SOM input K is approximately equal to that of NG. Therefore, even in the first year, SBPPRL produce at least as much biomass as NG.

Table 16 – Models parameters for each grassland system.

Grassland system	Parameters in analytical and linear models									
	Pooled-data model					Specific-data model				
	K' pp.yr^{-1}	a yr^{-1}	K (SOM_0= 0.87%) pp.yr^{-1}	α yr^{-1}	$1-e^{-\alpha}$ -	K' pp.yr^{-1}	a yr^{-1}	K (SOM_0= 0.87%) pp.yr^{-1}	α yr^{-1}	$1-e^{-\alpha}$ -
SBPPRL (t=1)	0.097	0.544	0.570	0.365	0.306	0.135	0.544	0.609	0.429	0.349
SBPPRL (t>1)	0.604		1.078			0.697		1.171		
FNG	0.289		0.763			0.508	0.566	1.000	0.443	0.358
NG	0.109		0.583			-0.048	0.432	0.328	0.190	0.173

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; *K'*, *K*, α, *a* – Parameters in Equation (1.7).

Using the parameters in Table 16, we determined the average SOM increase in 10 years from each grassland type shown in Table 17 and depicted in Figure 16 and Figure 17. Results show that SBPPRL on average increase their SOM concentration by 0.19-0.20 pp.yr^{-1}, which is equivalent to 1.33-1.40 t C.ha^{-1}.yr^{-1} or 1.48-1.56 t C.ha^{-1}.yr^{-1}, depending on the soil density, using Table 10. This increase is higher than for FNG (0.13-0.14 pp.yr^{-1}, equivalent to 0.94-0.95 t C.ha^{-1}.yr^{-1} or 1.05-1.07 t C.ha^{-1}.yr^{-1}, depending of soil density) and NG (0.07-0.08 pp.yr^{-1}, equivalent to 0.51-0.53 t C.ha^{-1}.yr^{-1} or 0.57-0.59 t C.ha^{-1}.yr^{-1}, depending on soil density). The difference of the average increase in 10 years between the pooled-data and the specific-data models is lower than 0.01 pp per year or 0.08 t C.ha^{-1}.yr^{-1} for all grassland systems, which shows that both models are very similar in the results obtained. Note that all plots were previously NG. Plots remaining NG were only tilled in the first year, and so results for NG must be read as post-tillage dynamics.

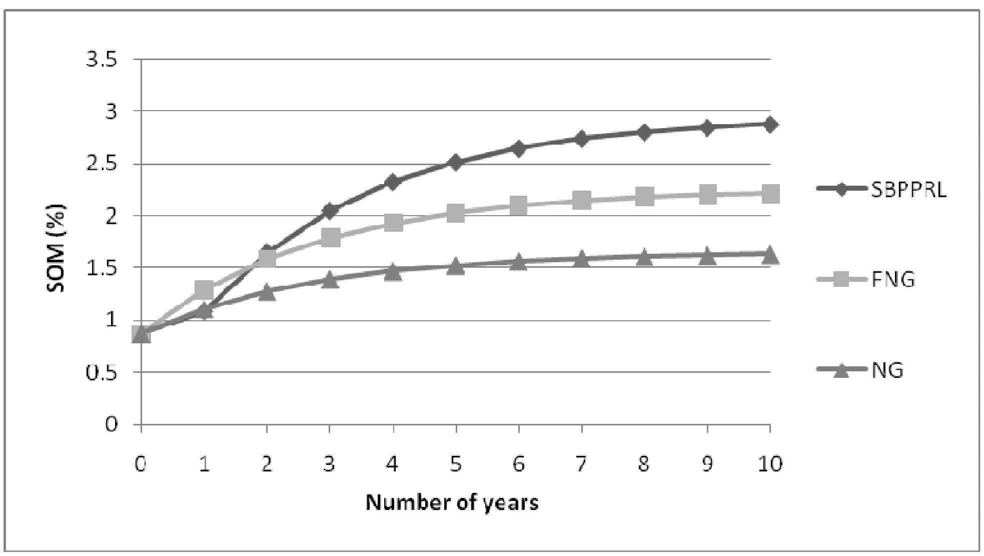
The intersection of the SOM increase curves for SBPPRL and FNG happens due to the first-year loss imposed by the dummy variable. Part of this loss is due to tillage, and so happens in the beginning (year 0). However, tillage has an effect on the mineralization rate. We could not estimate an α for the first year, since that would imply the use of a specific-data model for first-year observations, which are only eight. The dummy we use for the first-year in SBPPRL only affects the coefficient for *K'*, which correctly depicts the decreased productivity due to the existence of less seeds, and therefore less biomass production, in the plot, which happens throughout the first year. Therefore, the first-year dummy also has to capture the effect of tillage, which should be instantaneous and only seen in α. The only way to do so is by spreading the SOM loss throughout the first-year.

Table 17 – Estimated SOM concentration per year, starting from $SOM_0 = 0.87\%$.

Year			Estimated SOM concentration (%)					
			SBPPRL		FNG		NG	
			Pooled-data model	Specific-data model	Pooled-data model	Specific-data model	Pooled-data model	Specific-data model
0			0.87	0.87	0.87	0.87	0.87	0.87
1			1.08	1.29	1.11	1.09	1.37	1.02
2			1.65	1.59	1.28	1.69	1.69	1.14
3			2.05	1.79	1.39	2.09	1.89	1.24
4			2.33	1.93	1.47	2.34	2.02	1.33
5			2.52	2.03	1.53	2.51	2.11	1.40
6			2.65	2.10	1.57	2.61	2.16	1.46
7			2.74	2.14	1.59	2.68	2.20	1.50
8			2.81	2.18	1.61	2.73	2.22	1.54
9			2.85	2.20	1.62	2.76	2.23	1.58
10			2.88	2.21	1.63	2.78	2.24	1.60
Average increase in 10 years	**$pp.yr^{-1}$**		**0.20**	**0.13**	**0.08**	**0.19**	**0.14**	**0.07**
	$t\ C.ha^{-1}.yr^{-1}$	**MBD = 1.25 $g.cm^{-3}$**	**1.40**	**0.94**	**0.53**	**1.33**	**0.95**	**0.51**
		MBD = 1.40 $g.cm^{-3}$	**1.56**	**1.05**	**0.59**	**1.48**	**1.07**	**0.57**

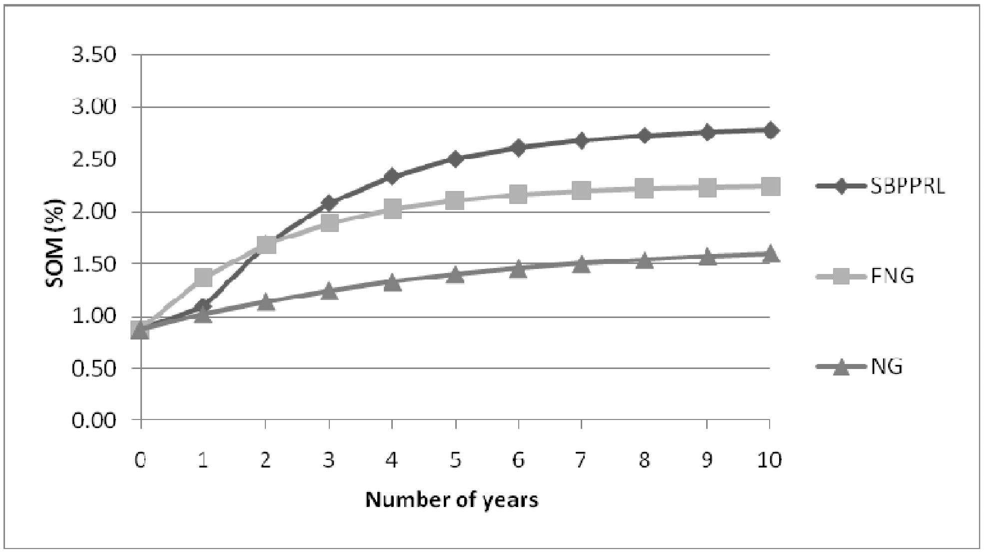
NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; pp – percentage points; MBD – Mineral Bulk Density.

Figure 16 – Simulated SOM concentration in each year, as estimated by the pooled-data model, using data filled-in using geometric averages, starting from 0.87%.



NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter concentration.

Figure 17 – Simulated SOM concentration in each year, as estimated by a specific-data model, using data filled-in using geometric averages, starting from 0.87%.



NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

2.5 Synthesis of results and discussion

Since SOM is the key parameter that determines other environmental services of pastures, in this Chapter we studied SOM dynamics in sown biodiverse and natural types of rainfed grasslands: SOM concentration is important for many agronomic and environmental reasons. We defined a model to calculate average SOM increases as a balance between accumulation of organic material in soils and mineralization of the available SOM pool. In time, as SOM increases, it eventually reaches an upper bound. Therefore, in a grassland system with no change in management, the entry and the mineralization eventually cancel each other. Other studies in literature indicate that SOM also increases asymptotically after land use changes (Sollins *et al.,* 1996; West and Six, 2007), as well as in the response to exogenous inputs of carbon or fertilizers (Six *et al.*, 2002; Six *et al.*, 2004).

Our model was calibrated using five years of soil analyses from eight locations. In each location there were two or three plots with two or three of the different grassland systems considered. We use statistical calibration to adjust an asymptotic curve to the data and obtain the model parameters. The parameters in our model were statistically determined using two methods which yielded a significant ($p < 0.10$) fit to the data. This is an important result, since we only had soil analyses for the first five years after installation (for one of them, 2002, there is practically no data). Considering that SOM saturation in soils is assessed in the long term (Stewart *et al.*, 2007), the fact that the asymptotic pattern is already picked up in the data is a strong conclusion.

We find that the expected steady-state long term SOM concentration in undisturbed SBPPRL is higher than in NG and FNG. In 10 years, there is an average increase of 0.19-0.20 percentage points per year in SBPPRL (equivalent to 1.33-1.40 t $C.ha^{-1}.yr^{-1}$ or 1.48-1.56 t $C.ha^{-1}.yr^{-1}$, depending on the soil density). In turn, SOM increases in FNG and NG are respectively 0.13-0.14 (0.94-0.95 t $C.ha^{-1}.yr^{-1}$ or 1.05-1.07 t $C.ha^{-1}.yr^{-1}$) and 0.07-0.08 percentage points per year (0.51-0.53 t $C.ha^{-1}.yr^{-1}$ or 0.57-0.59 t $C.ha^{-1}.yr^{-1}$). SBPPRL induces SOM increase due to sowing and fertilization, but mineralization rates are equal to those from FNG. NG have stable SOM pools, with lower input and mineralization rates.

Using a pooled-data model involves the estimation of fewer parameters, but requires one more assumption: that the mineralization rate is approximately independent of the grassland type. This hypothesis was validated for SBPPRL and FNG due to the fact that using a specific-data model also results in equal mineralization rates for both systems. Since there was no land use change in NG, SOM pools are already stabilized, and thus both the input of SOM and the mineralization rate were lower. The adjustments to the data of both the pooled and specific-data models are also equally good. Therefore, results are consistent despite the use of alternative estimation methods.

The parameter α is negative for the pooled-data model and in the specific-data models for SBPPRL and NG, when the initial SOM concentration is not used. There are three alternative explanations for why the model in these situations is adjusting a non-saturating exponential curve (positive second derivative) to the data.

The first explanation is that the parameter K is also a function of SOM concentration. The base model in Equation (1.4) would then be, for example:

$$\frac{dSOM}{dt} = K^{'} + kSOM - \alpha SOM \,. \tag{1.12}$$

In this case, the parameter related to the SOM term, which is then estimated in the regressions, is not the mineralization rate: it is the mineralization rate minus the coefficient for the variable SOM input. Therefore, when $\alpha - k < 0$, the model would be merely stating that the variable SOM input is higher than the mineralization rate. Note that it would not be possible to include two variables in the model instead of just α, because then we would not be able to estimate the parameters in the regression.

The second explanation is that, in SBPPRL, SOM increases with the consolidation of the installation of the pasture. However, since in this model SOM always increases with time, it becomes a proxy for the number of years since the installation of the pasture. Therefore, there is a time-dependent effect on K, which is indistinguishable in practice from the one mentioned before, and so Equation (1.12) also states this approach.

However, if these hypotheses were correct, then it would be an effect that happens mainly (in the first case) or exclusively (in the second) in SBPPRL. The results, however, do not confirm it. For the specific-data models, α is always lower (and sometimes negative) in SBPPRL, but there is also a negative α for NG.

Therefore, the explanation is most likely that the model requires the initial SOM concentration. Table 14 shows that when SOM_0 is considered α is always positive. If more productive soils (those that begin with a higher initial SOM concentration) will increase their SOM level more regardless of the grassland system, then our model cannot cope with that exogenous fact unless it considers a negative mineralization rate. This third explanation is the one which led us to use the initial SOM concentration as an independent variable.

In fact, even if the first two explanations are true, SOM_0 is in fact capturing some of the effect mentioned. We can also see in Table 14 that the introduction of the initial SOM concentration as an independent variable in the model increases the values of *K* and α. This is due to the fact that part of the supposed variable effect of *K* (which we know from the first and the second explanations) is captured by the initial SOM concentration, thus increasing α. The increase in α is then balanced by the increase in the fixed part of *K*, so that the same long term equilibrium is met.

The use of the initial SOM concentration slightly increases the adjusted R^2, when taking the increased number of degrees of freedom into account. In the short term, and given the rationale stated above, a negative α could only be stating a transitory compensation of mineralization by the specific input. But in the long run this effect eventually plays out and a steady state is reached. Since we are using data obtained during five years, but wish to extrapolate for a longer period, the use of the initial SOM concentration stabilizes the model, translating the long term dynamics.The main objective of the model was to obtain a long-term trend for SOM dynamics in the different grassland systems. We did not wish to obtain a model which predicts year-by-year SOM concentration, but rather to show the average SOM accumulation potential of each system. Therefore, the main conclusions of this study are that SBPPRL provide soils an increased organic matter pool. Starting from an arbitrary 0.87% SOM initial value, in 10 years there is an average increase of 0.19-0.20 $pp.yr^{-1}$ in SBPPRL. SBPPRL increase their SOM concentration more than other grassland systems. SOM increases in FNG and NG are respectively 0.13-0.14 and 0.07-0.08 $pp.yr^{1}$.

The increased input in SBPPRL is due to two factors. First, production responds to fertilization, and this response seems to depend only on soil intrinsic quality, and so it is a similar effect for SBPPRL and FNG. Second, production responds to the improved seed bank independently of soil characteristics. It is important to notice that SBPPRL in the first-year were tilled and have a small seed bank (the seeds sown). This productivity loss effect in the first-year is needed to interpret results.

The actual difference between SBPPRL and natural pastures may be even greater than we show here. FNG and NG may be overestimated due to the fact that plots were contiguous, since Carneiro *et al.* (2005) explain that there was some contamination of natural grasslands by sown species. Note also that, even though we used 0.87% as the arbitrary starting SOM, Table 11 shows that few farms had such low initial SOM concentrations. If initial SOM concentration is higher, and since *a* is positive (and higher for SBPPRL), then yearly increases are higher as well.

To our knowledge, there are no other internationally published studies on SBPPRL in Portugal or elsewhere. We find that our results are similar to those found in other preliminary Portuguese studies. Some early results hint that in 10 years SBPPRL increase SOM from 1 to 3% (Crespo, 2004). At Herdade dos Esquerdos, in Vaiamonte (Portalegre, Portugal), following a programme of SBPPRL installation, SOM concentration across the farm increased from between 0.7% and 1.2% in 1979 to between 1.45% and 4.40% in 2003 (Crespo *et al.*, 2004; Crespo, 2006a,b). This SOM increase is higher than that of any natural grassland under any form of management found in the literature.

In a related study, Aires *et al.* (2008) measured carbon fluxes over a pasture in Southern Portugal, similar to natural pastures in our study. They found that, in 2004-2005 (drought year), pastures emitted 0.49 t $C.ha^{-1}.yr^{-1}$, while in 2005-2006 (normal precipitation year) they sequestered 1.91 t $C.ha^{-1}.yr^{-1}$. This high intra-annual variability is also captured by our results, since yearly measurements of SOM concentration oscillates around a medium/long-term increasing trend (the trend is given by our model).

Regarding the first year in Aires *et al.*'s (2008) study, the drought year, it was also a sampling year in our study. Table 11 shows that in FNG and NG there was also a decrease in SOM from 2004 to 2005, which is the period for which Aires *et al.* (2008) concluded that soils had been emitters. FNG lost, on average, 0.08 pp, and NG lost 0.49 pp. These values are equivalent to a loss of 0.56 – 0.62 t $C.ha^{-1}.yr^{-1}$, and 3.41 – 3.81 t $C.ha^{-1}.yr^{-1}$, for FNG and NG respectively. When comparing these results to the results of Aires *et al*. (2008), we see that our approach does not under-estimate emissions.

Still regarding the same period, it is interesting to notice that SBPPRL increased SOM concentration even in the drought year by 0.97 pp on average (Table 11). This indicates that SBPPRL have one further advantage, which is increased resilience in drought years.

Regarding the second sampling year in Aires *et al*. (2008), it was a normal precipitation year. Table 15 shows that the lowest average SOM increase in our study is for NG (specific-data model), and is 0.07 pp per year, and the highest result is 0.20 pp per year for SBPPRL (specific-data model). These values correspond, respectively, to 0.48 – 0.54 t $C.ha^{-1}.yr^{-1}$, and 1.39 – 1.54 t $C.ha^{-1}.yr^{-1}$ (two values provided, depending on soil density).

The similarity in results between our work and Aires *et al.*'s (2008) work also indicates consistency between our method, using soil samples, and theirs, using flux measurements.

In the next chapter, we turn to the quantification of the environmental services provided by SBPPRL.

3. Quantifying the environmental services provided by SBPPRL

Chapter 3 deals with the quantification of some environmental effects mentioned in Chapter 1, starting from the quantification of SOM dynamics. We namely determine the carbon balance of NG and SBPPRL systems, including the emissions from animals, legumes and liming. We compare rainfed SBPPRL to other agricultural land uses and practices, to determine which has the highest potential to sequester carbon. We also compare how rainfed SBPPRL compare to their alternative, which in this case is not NG but maize used for biofuel production.

Then, we perform an LCA of the systems, and finally we calculate soil loss in both systems, and refer studies on biodiversity. By doing so, we determine whether SBPPRL are worse that NG in any other environmental impact theme.

3.1 From SOM accumulation to carbon sequestration

3.1.1 Equivalency factors

In Section 2.3.4, we calculated the C sequestration equivalent to an increase in 1% SOM. To convert those increases from C to CO_2, we multiply that value for the molecular weight of CO_2 (44) and divide it by the atomic weight of carbon (12).

Table 18 shows final results for carbon sequestration in t $CO_2.ha^{-1}$ equivalent to 1% increase in SOM (Teixeira, 2008, 2010b).

Table 18 - Carbon sequestration equivalent to the increase in SOM of 1 pp in 10, 20 and 30 cm.

Depth (cm)	MBD ($g.cm^{-3}$)	t $C.ha^{-1}$	t $CO_2.ha^{-1}$
10	1.25	6.96	25.5
10	1.40	7.75	28.4
20	1.25	13.9	51.0
20	1.40	15.5	56.8
30	1.25	20.9	76.6
30	1.40	23.3	85.3

MBD – Mineral Bulk Density; C – carbon; CO_2 – carbon dioxide; SOM – Soil Organic Matter; pp – percentage points.

3.1.2 Which value to use?

Based on these values, and since soil samples for pastures were collected at 10 cm, the SOM increase of 0.20 percent points per year in SBPPRL estimated in the last chapter corresponds to the sequestration of 5.10 or 5.68 t $CO_2.ha^{-1}.yr^{-1}$, depending on the soil density we consider.

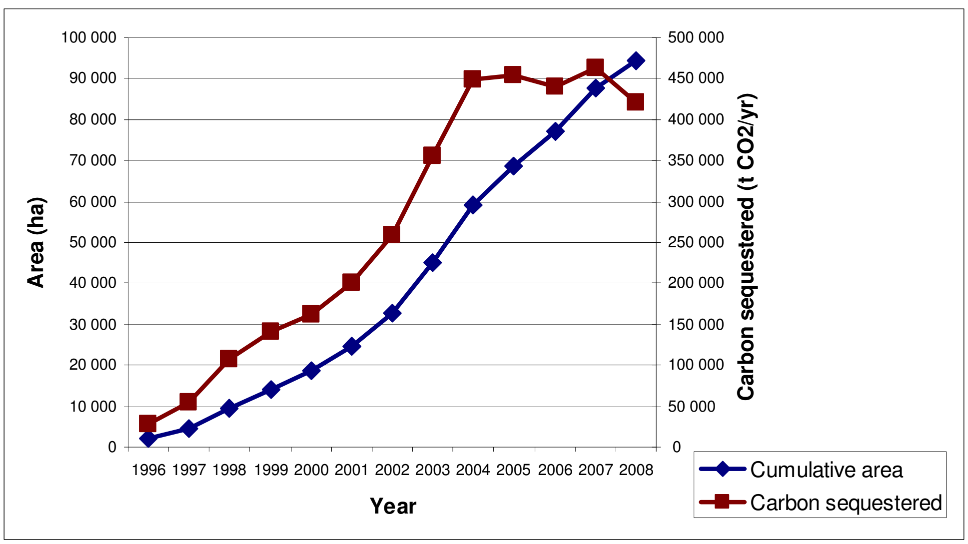
Therefore, the question arises: which soil density should we consider if no information is available? If pastures were sown using tillage, then it is very likely that during the first years after installation the MBD is close to the national average. However, we are considering as the relevant soil layer only the first 10 cm of topsoil. This is the layer which is more affected by interactions with the aerial system, and the one whose density varies the most easily. Therefore, as time goes by, livestock stomping and the lack of mobilization will undoubtfully create compactation in this upper layer of soil, increasing its MBD. If pastures are sown using no-tillage, this effect is even more dramatic.

Whenever we need to use a value throughout this thesis, we opted to use the round number of 5.0 t $CO_2.ha^{-1}.yr^{-1}$. This number is merely indicative; it is not the average of sequestration for the range of possible density values, or even an choice regarding the density we believe to be representative in the country.

3.1.3 How much carbon have SBPPRL sequestered lately?

Using sales data from Fertiprado, shown in Section 1.4.4 (João Paulo Crespo, personal communication), we calculated how the service of carbon sequestration has evolved in the past years. The rate of sales had been increasing since the beginning of the firm, but then started decreasing since 2004. Since SBPPRL sequester more carbon in the first years after the installation, and the area of first-year SBPPRL has decreased, the carbon sequestration service has remained relatively constant since 2004, as shown in Figure 18. To obtain Figure 18, we considered the specific yearly increases for each installation year. Overall, SBPPRL have sequestered around 3.5 Mt CO_2 from 1996 to 2008[31].

Figure 18 – Evolution of the environmental service of carbon sequestration provided by SBPPRL.



[31] Note that, in these calculations, we used the whole area sold by Fertiprado, which is around 70 000 ha. PNAC (2007) has a more modest estimate of 70 000 ha of SBPPRL in Portugal.

3.2 Calculating the carbon balances of pastures

3.2.1 Can we really expect high SOM sequestration in pastures?

To answer to that preliminary question, we turn to the literature. The use of grassland management as a carbon sink is well documented. Net Primary Productivity (NPP) and Net Ecosystem Productivity (NEP) in pastures are positive saturating functions of plant species and functional diversity (Catovsky *et al.*, 2002). NEP is measured by above and belowground production. Belowground production (roots), when incorporated into soils, corresponds to carbon storage. Therefore, some kinds of pasture may have high sequestration potentials. Several studies, such as Soussana *et al.* (2007) and Byrne *et al.* (2007), have tried to determine a correct number for the carbon sequestration potential via grassland management worldwide. Table 19 is a review of available data.

Freibauer *et al.* (2004) reviewed the potential for European soils to sequester carbon. They also evaluated suitable land for carbon farming. In their survey, they do not refer improved grasslands, but they indicate global grassland potential (with grazing management) as 0.8-2.6 t $CO_2.ha^{-1}.yr^{-1}$. Frank (2002) measured CO_2 flux over a grazed wheatgrass pasture during growing season as 2.18 t $CO_2.ha^{-1}.yr^{-1}$. Higher values were found by Smith (2004) – 4.4-6.2 t $CO_2.ha^{-1}.yr^{-1}$ – and Tschakert (2004) – 5.35 t $CO_2.ha^{-1}.yr^{-1}$. Both studies refer to the conversion of cropland to grassland, but the first study results from a survey, and the second from the application of a biogeochemical model.

As for the effect of increased grazing, Reeder and Schuman (2002) used 12 year data for mixed grass and 56 year data for short grass rangeland to conclude that grazed land has higher soil organic carbon than non-grazed areas. Their conclusions are supported by the bulk of available literature. For example, Freibauer *et al.* (2004) state a value for the increase of carbon sequestration by grazing management of 0.8-2.6 t $CO_2.ha^{-1}.yr^{-1}$.

Table 19 – Literature review for the potential of cropland and grassland soils to sequester carbon

Land Use	CO_2 sequestration (t CO_2 .$ha^{-1}.yr^{-1}$)	Method	Reference
Grassland	0.4-11.1	Survey	Conant *et al.*, 2001
Organic input on arable land	1.0-3.0	Survey	ECCP, 2003
Revegetation of set-aside land and introduction of perennials	2.0-7.0	Survey	ECCP, 2003
Promote organic farming	0-2.0	Survey	ECCP, 2003
Water table in peatland	5.0-15.0	Survey	ECCP, 2003
Temperate grassland	2.2 - 2.8	Flux Measurement	Frank, 2002
Eliminate bare fallow	0.6-2.8	Survey	Freibauer *et al.*, 2004
Grassland grazing management	0.8-2.6	Survey	Freibauer *et al.*, 2004
Grassland and pastures	0.18-0.37	Survey	Freibauer *et al.*, 2004
Grassland	0.62	Measurement	McLauchlan *et al.*, 2006
Semi-natural grassland	0.7 - 1.0	Modelling	Sindhoj *et al.*, 2006
Convert cropland to grassland	4.4 – 6.2	Survey	Smith, 2004
Convert cropland to grassland	5.4	Modelling	Tschakert, 2004
Enhancing rotation complexity	0.07 +- 0.04	Survey	West and Post, 2002

However, and regarding the SBPPRL, there were hints that suggested the possibility that the potential for carbon sequestration was even higher.

To understand what the balance means, we must first model them.

3.2.2 Overall C and N models of pastures

As noted before, we used field data collected from 2001 to 2005 (Table 11, location in Figure 12) in several locations in Portugal, during Projects AGRO 87 and AGRO 71. We will, then, focus on results from those projects, subjected to the experimental setting.

The model of carbon (C) and nitrogen (N) cycle processes contributing to the GHG balance in grasslands can be read as follows. Legumes and grasses grow in consociation using atmospheric CO_2. Symbiotic associations of legumes and microorganisms, namely *Rhizobium* (Bot and Benitez, 2005), fix atmospheric nitrogen as well. Belowground, a complex set of reactions between plant (roots), soil mineral particles, microorganisms and macrofauna (earthworms, etc.) takes place (Ostle *et al.*, 2009).

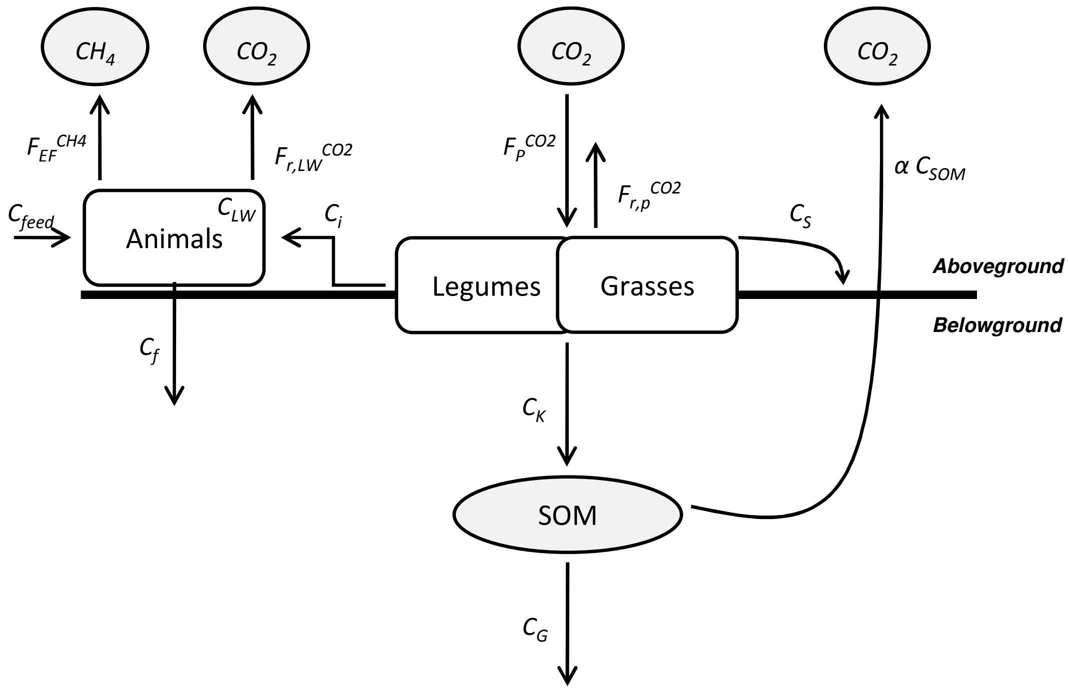
Three important outputs, in terms of GHG come out of the sum of these processes: SOM accumulation in soils, leaching of SOM particles and dissolved N, and emission of N_2O to the atmosphere. The aboveground part of the plants is eaten (on site or off-site) by animals, and be complemented by feeds (C and N import). Either through the process of enteric fermentation or from the degradation of wastes, there is some emission of CH_4 and N_2O. Finally, there is application of limestone to increase pH, and there is a corresponding emission of CO_2.

In these grassland systems there is no off-site digestion by animals (only direct grazing), and there are no applications of organic fertilizers (for a discussion on the implications of both fluxes on the global balance, see Soussana *et al.*, 2007).

3.2.2.1. Carbon balance

The carbon balance of pastures can thus be defined as the overall balance from several sub-systems, namely the soil system, the plant system and the animal system. All of these exchange greenhouse gases (GHG) with the atmosphere. The overall carbon balance is shown in Figure 19.

Figure 19 – Carbon balance in the grassland systems.



C – Carbon; CH_4 – methane; CO_2 – Carbon dioxide; F – flux; SOM – Soil Organic Matter; P – Photosynthesis; r – respiration; S – Senescence; SOM – Soil Organic Matter; LW – Live weight; G – Groundwater; α – mineralization rate; f – faeces or manure; i – ingestion; K – organic matter input to soil from roots.

The carbon balance of the plant system (all variables in in kg C ha^{-1} yr^{-1}) is:

$$F_{P}^{CO_2} - F_{r,plant}^{CO_2} = P_{a}^{C} + P_{b}^{C}\ . \tag{3.1}$$

This means that the flux of CO_2 sequestered by the plant through photosynthesis (F_P^C) minus the plant respiration ($F_{r,plant}^{CO_2}$) will incorporate the total plant biomass production (P_t^C), which is equal to the sum of above (P_a^C) and belowground (P_b^C) biomass. Total production is separated in above and belowground according to the root to shoot (R:S) ratio of pasture plants (equal to P_b^C / P_a^C). Therefore, the aboveground production is equal to:

$$\frac{1}{1+R:S}\ P_t^C = C_S + C_i\ , \tag{3.2}$$

where C_S is input (output from plant) of biomass to the soil by leave senescence, and C_i is ingestion by livestock. Under correctly managed grazing, around 30% of plant biomass escapes the intake by animals (Mazzanti *et al.,* 1994, cit. in Sanaulluah *et al.* 2009), and therefore $C_S = 0.30\ P_a^C$, which means that $1/R:S\ P_t^C = (1\text{-}0.30)^{-1}\ C_i$. Intake by animals is, on average, C_i = 4.8 kg C.LU^{-1}.day^{-1} (Thornley, 1998, cit. in Soussana *et al.*, 2007). Assuming that livestock grazes half of the year (during Spring and Autumn, even though the intake is highly variable), then C_i = 0.88 t C.LU^{-1}.yr^{-1}, or converting to equivalent CO_2, $C_i = 3.2$ t CO_2e.LU^{-1}.yr^{-1}. Furthermore, the IPCC (1997) indicates 2.8 as the default root to shoot ratio (R:S) for semi-arid grasslands. This value is consistent with the R:S of 0.5 to 4.8 in grazed pastures, which is the range of the comprehensive data for several regions gathered by Coupland (1976).

Belowground production will be equal to:

$$\frac{R{:}S}{1+R{:}S} P_t^C = K_{roots} + C_G , \tag{3.3}$$

where K_{roots} is SOM input due to soil processes of humus formation from roots, and C_G is emissions from leached organic matter to groundwater involved in leaching. In this section, we consider $C_G = 0$.

Therefore, the soil stock will in turn be:

$$C_K - \alpha\, C_{SOM} = K_{roots} + C_S + C_f , \tag{3.4}$$

where $C_K - \alpha\, C_{SOM}$ is the balance between the total carbon input (from roots, leave senescence and faeces) and the mineralized fraction of the existing SOM pool, and C_f is input from livestock faeces. According to APA (2009), daily excretion by cows is 2.79 $kg_{dry\ matter}$.LU^{-1}.day^{-1}. Considering that livestock grazes half of the year, and assuming that 58% of the excreted dry matter if C, then $C_f = 1.8$ t CO_2e.LU^{-1}.yr^{-1}.

Regarding the livestock balance, it may be put as:

$$C_i + C_{feed} = F_{r,LW}^{CO_2} + C_{LW} + F_{EF}^{CH_4} + C_f , \tag{3.5}$$

where C_{feed} is concentrated feed intake, $F_{r,LW}^{CO_2}$ is livestock respiration (which, according to Soussana *et al.*, 2007, is around 1 t CO_2.LU^{-1}.yr^{-1}), C_{LW} is net C incorporated in the animal live weight exported from the field, and $F_{EF}^{CH_4}$ are carbon losses from CH_4 emissions from enteric fermentation. We assume that C_{feed} is negligible and do not consider it here (as for the N balance).

Breeding cows are kept in pastures but each cow has one steer per year, which is typically removed at birth or after six months. In the second case, each 6 month-old steer weighs around 250 kg. 0 - 20% of this weight is carbon, and therefore the C exported, translated into CO_2e, is at most $C_{LW} = 0.18$ t CO_2e.LU^{-1}.yr^{-1}. We considered this figure negligible, and therefore removed it from calculations.

The overall balance, from the sum of all the sub-systems is:

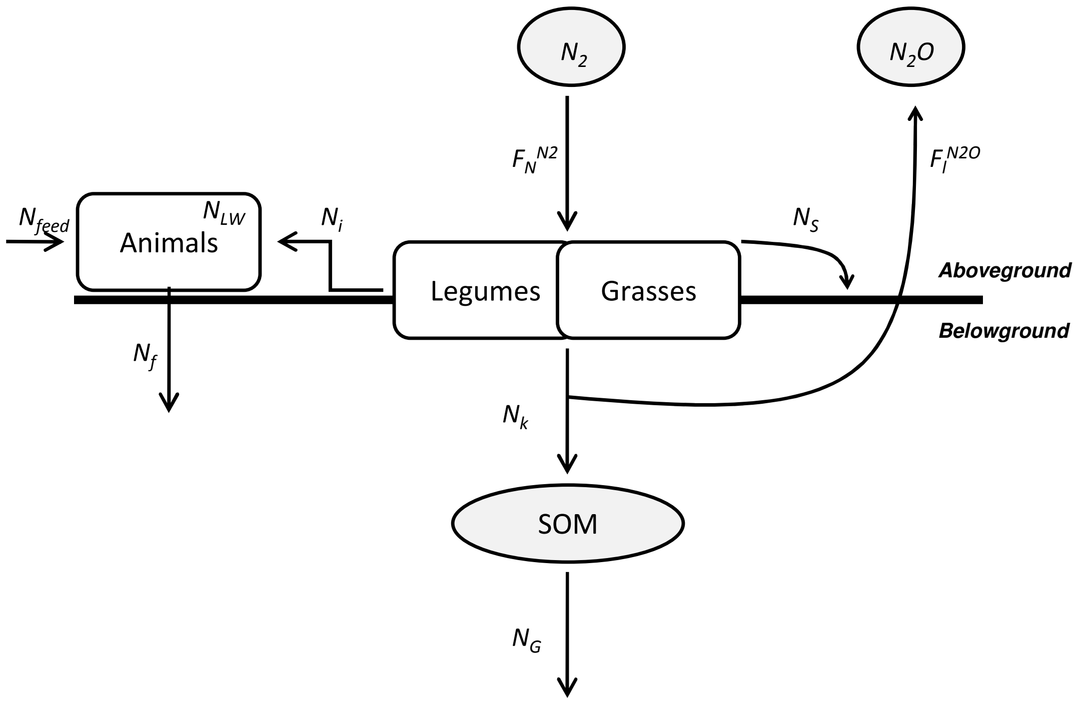
$$F_P^{CO_2} + C_{feed} = C_K - \alpha\, C_{SOM} + C_{LW} + C_G + F_{r,plant}^{CO_2} + F_{r,LW}^{CO_2} + F_{EF}^{CH_4} . \tag{3.6}$$

This means that the overall carbon inputs to the system are sequestration of C during photosynthesis by plants and (possible) introduction via feeds. This carbon is either stored as SOM (balance between entry and mineralized fraction), exported as animal live weight, lost to groundwater, emitted from respiration by plants and livestock, or is emitted by enteric fermentation. Note that $P_t^C = F_P^{CO_2} - F_{r,plant}^{CO_2}$, and therefore the two terms may be estimated together measuring total biomass production.. Note also that C_{feed}, C_{LW} and C_G are equal to zero.

3.2.2.2. Nitrogen balance

The nitrogen balance can also be defined according to the three sub-systems defined for the carbon balance, as shown in Figure 20.

Figure 20 – Nitrogen balance in the grassland systems.



N – Nitrogen; N_2O – nitrous oxide; N2 –Nitrogen gas; F – flux; SOM – Soil Organic Matter; P – Photosynthsis; l – legumes; S – Senescence; SOM – Soil Organic Matter; LW – Live weight; G – Groundwater; α – mineralization rate; f – faeces or manure; i - ingestion; k – organic matter input to soil from roots.

In this case, there is an N_2 input due to legumes (instead of CO_2), and so the plant sub-system may be defined as (all variables in in kg N ha^{-1} yr^{-1}):

$$F_l^{N_2} = P_a^N + P_b^N \,. \tag{3.7}$$

This means that the flux of N_2 sequestered by legumes ($F_l^{N_2}$) will be incorporated either its above (P_a^N) or belowground (P_b^N) biomass, the sum of which is N in the total biomass (P_t^N). As for carbon, total production is separated in above and belowground according to the root to shoot (R:S) ratio of pasture plants. Therefore, the aboveground part is equal to:

$$\frac{1}{1+R:S} P_t^N = N_S + N_i \,, \tag{3.8}$$

where N_S is nitrogen in input (output from plant) of biomass to the soil by leaves' senescence, and N_i is N ingestion by livestock. Analogously to the C balance, $P_a^N = (1\text{-}0.30)^{-1} \; N_i$, due to the relation between production and senescence.

Belowground production will be equal to:

$$\frac{R:S}{1+R:S}\,P_t^N = N_K + N_G\,, \tag{3.9}$$

where N_K is the N incorporated in the SOM stock due to soil processes of humus formation from roots and N_G is N lost from leaching. We consider $N_G \approx 0$ for both grassland systems: for NG, because there are few legumes and no N-fertilization; for SBPPRL, because experience shows that practically no N is leached. An unpublished study (Rodrigues *et al.*, 2010) from the Mountain Research Centre of the Superior Agronomic School in Bragança (Northern Portugal) has determined that soil inorganic-N levels are low in both grassland systems, and there were no significant risks of nitrogen losses (via nitrate leaching and nitrification and NH_3 volatilization).

Therefore, the soil stock will in turn be:

$$\Delta N_{SOM} = N_K + N_S + N_f + F_l^{N_2O}\,, \tag{3.10}$$

where N_{SOM} is N in the SOM pool, N_f is input from livestock faeces and $F_l^{N_2O}$ is the part of soil N which is emitted as N_2O during the complex nitrification/denitrification processes that occur in the soils. Regarding the livestock balance, it may be put as:

$$N_i + N_{feed} = N_{LW} + N_f\,, \tag{3.11}$$

where N_{feed} is the N input in concentrated feeds, and N_{LW} is N exported in the animal live weight. As in the C balance, one steer is exported per year, 0 - 2% of which is N, and therefore N_{LW} = 5 kg $N.LU^{-1}.yr^{-1}$. We also assume that this figure is negligible, and therefore removed it from calculations.

The overall balance, from the sum of all the sub-systems is:

$$F_l^{N_2} + N_{feed} = \Delta N_{SOM} + N_{LW} + N_G + F_l^{N_2O}\,. \tag{3.12}$$

This means that nitrogen fixed by legumes is used by legumes themselves and also by grasses to increase biomass. Some feed N may also enter the system. These N inputs are either incorporated in the soil SOM, exported as live weight, re-emitted to the soil as faeces or emitted as N_2O to the atmosphere. Note that $F_l^{N_2} = P_t^N$, and therefore the flux may be estimated using N in total biomass production.. Note also that N_{feed}, N_{LW} and N_G are equal to zero.

3.2.2.3. Net greenhouse gas balance

The net greenhouse gas balance (NGHGB) is the sum of the contribution from all GHG weighted by their global warming potential (GWP) in a 100-year time horizon (Forster *et al.*, 2007):

$$NGHGB = GWP_{CO_2}F^{CO_2} + GWP_{CH_4}F^{CH_4} + GWP_{N_2O}F^{N_2O}\,, \tag{3.13}$$

where $GWP_{CO_2} = 1$, $GWP_{CH4} = 25$ and $GWP_{N_2O} = 298$. This balance is equal to the sum of the stock and GHG flux terms in the C and N balances.

We consider that CO_2 sequestration is equal to the only stock term, which is SOM increase through carbon sequestration (already shown). Carbon stocked in soil pools is the only immobilized form of carbon in the system, and thus is the only effective sink that can be measured. There is, however, C exportation in steers and CO_2 emissions from livestock. There are also CO_2 emissions that do not enter the overall plant-soil-atmosphere system, namely carbon emitted due to the application of limestone for acidity correction (C_{lime}). Net CO_2 emissions are thus equal to:

$$F^{CO_2} = \Delta C_{SOM} + C_{LW} + F^{CO_2}_{r,LW} + C_{lime}, \tag{3.14}$$

Optimum conditions for legume production require not-too-low pH. When soil pH(H_2O) is lower than 5.3, which happens in 20 to 30% of Portuguese soils (David Crespo, personal communication), $CaCO_3$ or $CaMg(CO_3)_2$, depending on the type of limestone used, are applied to SBPPRL. During this operation, limestone is applied on the surface (except when the pasture is installed). After reacting with the soil particles, the cations are incorporated in the soil structure and CO_2 is emitted. The emission factor, attending the stoichiometry of the substances, is around 0.12 kg CO_2 kg^{-1} limestone applied (IPCC, 2003). We assume that SBPPRL require the application of 2 t $CaCO_3$ in the installation and then every 4 years (year 4 and year 8 after installation).

The only source of CH_4 is livestock, via enteric fermentation. Therefore, $F^{CH_4} = F^{CH_4}_{EF}$. There is one further emission of N_2O from soil processes associated with nitrification/denitrification cycles from bacterial activities related to N fixation by legumes. Therefore, $F^{N_2O} = F^{N_2O}_{l}$.

3.2.2.4. <u>CH_4 emissions</u>

The overall balance of emissions from animals, in a general situation, ($F_{livestock}$ in kg CO_2e ha^{-1} yr^{-1}) is

$$E_{livestock} = GWP_{CH_4} F^{CH_4}_{faeces/manure} + GWP_{CH_4} F^{CH_4}_{EF} + GWP_{N_2O} F^{N_2O}_{faeces/manure}, \tag{3.15}$$

where $F_{faeces/manure,CH_4}$, $F_{faeces/manure,N_2O}$ and F_{EF,CH_4} are, respectively, CH_4 and N_2O emissions from faeces or manure, and CH_4 emissions from the process of enteric fermentation (EF). Each of these emissions are calculated as

$$F_{source,GHG} = k_{GHG}\ SR, \tag{3.16}$$

where k_{GHG} is the emission rate of each GHG (CH_4 or N_2O), measured in kg GHG $head^{-1}$ yr^{-1}; SR is the average annual stocking rate (head ha^{-1}).

Values for k_{GHG} are obtained from the Portuguese National Inventory Report (NIR) for 2007 (APA, 2009), as shown in Table 20. Manure is only produced in stables, where aerating conditions are poor and thus digestion of waste is anaerobic. Grazing livestock produce faeces which are well aerated. Therefore, as noted before, the only CH_4 emissions from livestock are due to enteric fermentation.

Table 20 – Emission factors for livestock sources.

	Emission factor ($kg.head^{-1}.yr^{-1}$)						LU (head ha^{-1})
	EF		Manure		Faeces		
Animal type	CH_4	N_20	CH_4	N_20	CH_4	N_20	
Non-dairy cattle	56.10	0.00	1.60	0.96	0.00	0.00	1.00
Steers	33.66	0.00	0.96	0.58	0.00	0.00	0.60

EF – Enteric Fermentation; CH_4 – methane; N_2O – nitrous oxide; LU – Livestock Unit. Source: APA (2009).

Field results for stocking rates were already shown in Section 2.3.1 (Carneiro *et al.*, 2005). Measured average stocking rates for NG and SBPPRL are, respectively, 0.42 and 1.03 $LU.ha^{-1}$. For simplicity purposes, we assume round figures of 0.5 and 1.0 $LU.ha^{-1}$ in calculations.

3.2.2.5. N_2O emissions

There are basically two ways in the literature to rapidly calculate nitrogen emissions from legumes. The first uses an emission factor per fixed nitrogen unit. The second uses a factor per production of dry matter.

IPCC (1997) considers an N_2O emission factor from legumes of 0.0125 $kgN_2O\text{-}N.kg^{-1}$ fixed N. Carneiro *et al.* (2005) state for several locations values for fixed N ranging from 100 (Coruche) to 300 $kg.ha^{-1}$ (Cercal). The average is about 180 $kg.ha^{-1}$. Therefore, emissions would be 1.25 - 3.75 kg $N_2O\text{-}N.ha^{-1}$, or 0.4 – 1.2 t $CO_2e.ha^{-1}$, considering the GWP of N_2O. The average would be about 0.7 t $CO_2e.ha^{-1}$

An alternative calculation may be done considering that sown grasslands have higher dry matter (DM) productivity. According to Carneiro *et al.* (2005), productivity varies from 2 000 kg $DM.ha^{-1}$ (Coruche, Portugal) to 9 000 kg $DM.ha^{-1}$ (Quinta da França, Portugal). On average, about 60% of such production is due to legumes (Carneiro *et al.,* 2005). Therefore, and considering an emission factor of 0.001 kg $N_2O\text{-}N.kg^{-1}$ DM, emissions would range from 1.2 to 5.4 kg $N_2O\text{-}N.ha^{-1}$, or 0.3 to 1.5 t $CO_2e.ha^{-1}$, with an average of 0.9 t $CO_2e.ha^{-1}$.

In this section we use the average of the two average values, which is 0.8 t $CO_2e.ha^{-1}$.

3.2.2.6. Biomass production in SBPPRL

Results obtained may be validated using field data to confirm that the C and N balances close for each sub-system (verifying the equality in each Equation). The parameters we require are the overall production of SBPPRL and N content, which were obtained by Carneiro *et al.* (2005) and are shown in Table 21. These results are the average from the first 4 years after installation.

Table 21 – Dry matter production and average N content of SBPPRL biomass (Carneiro *et al.*, 2005).

Farm #	Total dry matter ($kg.ha^{-1}$)	Average N content (%)	Average N sequestered ($kg.ha^{-1}$)
2	9 040	2.42	220
3	5 660	2.47	140
4	10 595	2.81	300
5	3 940	2.50	100
6	8 765	2.72	240
7	6 720	2.81	190
8	4 300	2.47	100
Average	**7 000**	**2.60**	**184**

N – nitrogen.

3.2.2.7. Results for the GHG balance

Results from the application of the emission factors in Table 20 to the changes in stocking rate considered are shown in Table 22. The doubling of the stocking rate will double emissions (1.4 instead of 0.7 t $CO_2e.ha^{-1}.yr^{-1}$), since there are no transfers and no changes in feed consumption.

Table 22 – CH_4 and CO_2e emissions from cattle.

	Breeding cows	
GHG	**NG**	**SBPPRL**
t $CH_4.ha^{-1}$	0.028	0.056
t $CO_2e.ha^{-1}$	0.701	1.403

NG – Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; CH_4 – methane; N_2O – nitrous oxide.

Table 23 shows results for NGHGB for NG, and Table 24 for SBPPRL. NGHGB is negative (sink) for both NG and SBPPRL. However, sink in SBPPRL is about three times that of NG, even though there are more emissions in this system than in NG. This is due to the fact that the average ΔC_{SOM} is higher for SBPPRL. Even in the first year, when SBPPRL lose carbon due to tillage and less production, they still increase the SOM pool more than NG. However, once the SBPPRL system becomes an emitter (more emissions from soils and cattle than carbon sequestration), which happens at the 6^{th} or 7^{th} year, it emits more than NG even in the most favourable scenario for SBPPRL.

Table 23 – NGHGB for NG.

Year	ΔC_{SOM}		C_{LW}	$F_{r,LW}^{CO_2}$	C_{lime}	$F_{EF}^{CH_4}$	$F_{l}^{N_2O}$	NGHGB	
	MBD = 1.25 g.cm^{-3}	MBD = 1.40 g.cm^{-3}						MBD = 1.25 g.cm^{-3}	MBD = 1.40 g.cm^{-3}
1	-4.96	-5.54	0.18	0.50	0.00	0.70	0.00	-3.58	-4.15
2	-3.69	-4.12	0.18	0.50	0.00	0.70	0.00	-2.31	-2.74
3	-2.77	-3.10	0.18	0.50	0.00	0.70	0.00	-1.39	-1.72
4	-2.10	-2.34	0.18	0.50	0.00	0.70	0.00	-0.72	-0.96
5	-1.60	-1.79	0.18	0.50	0.00	0.70	0.00	-0.22	-0.40
6	-1.23	-1.37	0.18	0.50	0.00	0.70	0.00	0.15	0.01
7	-0.95	-1.06	0.18	0.50	0.00	0.70	0.00	0.43	0.32
8	-0.74	-0.83	0.18	0.50	0.00	0.70	0.00	0.64	0.55
9	-0.58	-0.65	0.18	0.50	0.00	0.70	0.00	0.80	0.73
10	-0.46	-0.51	0.18	0.50	0.00	0.70	0.00	0.92	0.87
Average	**-1.91**	**-2.13**	**0.18**	**0.50**	**0.00**	**0.70**	**0.00**	**-0.53**	**-0.75**

NGHGB – Net Greenhouse Gas Balance; NG – Natural Grasslands; MBD – Mineral Bulk Density; C – Carbon; SOM – Soil Organic Matter; LW – Live Weight; r – Respiration; EF – Enteric Fermentation; CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

Table 24 – NGHGB for SBPPRL.

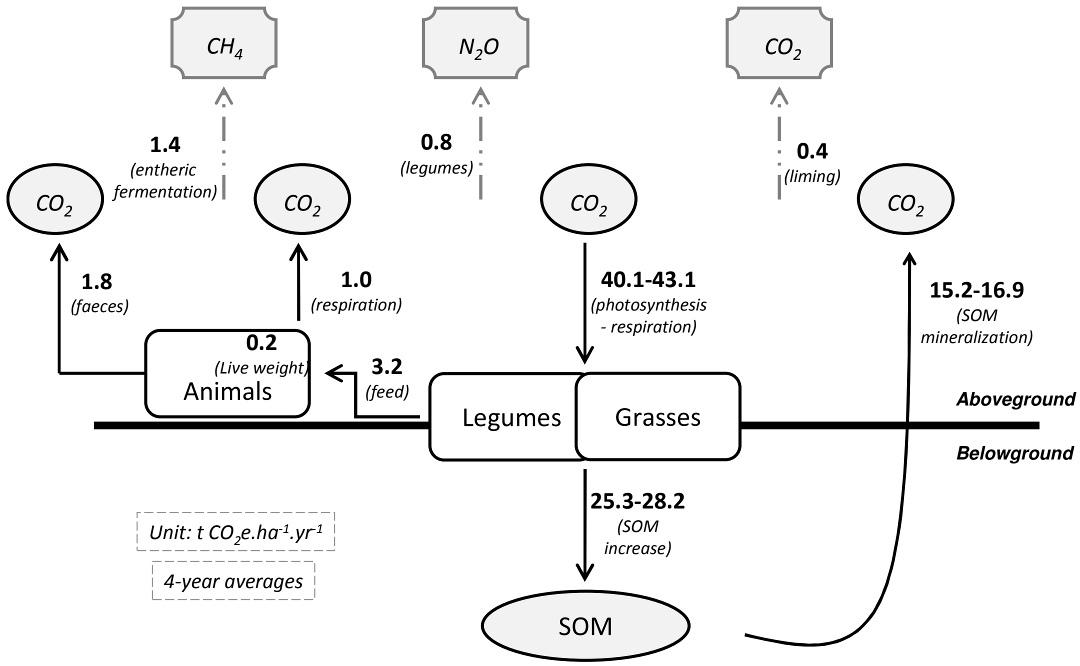
Year	ΔC_{SOM}		C_{LW}	$F_{r,LW}^{CO_2}$	C_{lime}	$F_{EF}^{CH_4}$	$F_{l}^{N_2O}$	NGHGB	
	MBD = 1.25 g.cm^{-3}	MBD = 1.40 g.cm^{-3}						MBD = 1.25 g.cm^{-3}	MBD = 1.40 g.cm^{-3}
1	-5.53	-6.18	0.18	1.00	0.24	1.40	0.80	-1.91	-2.56
2	-14.97	-16.72	0.18	1.00	0.00	1.40	0.80	-11.59	-13.33
3	-10.06	-11.23	0.18	1.00	0.00	1.40	0.80	-6.68	-7.85
4	-6.77	-7.56	0.18	1.00	0.24	1.40	0.80	-3.15	-3.93
5	-4.56	-5.09	0.18	1.00	0.00	1.40	0.80	-1.17	-1.70
6	-3.07	-3.43	0.18	1.00	0.00	1.40	0.80	0.31	-0.05
7	-2.07	-2.31	0.18	1.00	0.00	1.40	0.80	1.31	1.07
8	-1.40	-1.56	0.18	1.00	0.24	1.40	0.80	2.22	2.06
9	-0.95	-1.06	0.18	1.00	0.00	1.40	0.80	2.44	2.33
10	-0.64	-0.71	0.18	1.00	0.00	1.40	0.80	2.74	2.67
Average	**-5.00**	**-5.58**	**0.18**	**1.00**	**0.07**	**1.40**	**0.80**	**-1.55**	**-2.13**

NGHGB – Net Greenhouse Gas Balance; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; MBD – Mineral Bulk Density; C – Carbon; SOM – Soil Organic Matter; LW – Live Weight; r – Respiration; EF – Enteric Fermentation; CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

These results are divided between the several sources as shown in Figure 21. Aires *et al.* (2008) found a global primary production for natural grasslands in a normal precipitation year equal to 46.2 t CO_2e.ha^{-1}.yr^{-1}. This figure is similar to the production of SBPPRL shown in Figure 21. If we repeat calculations for natural pastures (using the model for NG obtained in section 1), we obtain a production of 19.1 – 20.4 t CO_2e.ha^{-}

[1].yr[-1]. This figure is approximately half of the one obtained in Aires *et al.*'s (2008) measurements.

Figure 21 – NGHGB of the SBPPRL system.



NGHGB – Net Greenhouse Gas Balance; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

3.2.2.8. Closure of C and N balances

The C and N balances of SBPPRL and NG are quantified in Table 25. On average, the simplified C balance states that the overall plant production has to be equal to the SOM balance in soils, livestock respiration and emissions from enteric fermentation:

$$P_t^{CO_2} = C_K - \alpha\, C_{SOM} + F_{r,LW}^{CO_2} + F_{EF}^{CH_4}\,. \tag{3.17}$$

For example, for SBPPRL, summing the three terms (notice that SOM is the average from the first four years), we obtain an estimated balance of production of 11.73 and 12.82 t CO_2e.ha^{-1}.yr^{-1}. Aboveground production, according to Table 21, was on average 7 t.ha^{-1}.yr^{-1}. Considering that 58 % of production is carbon, and multiplying by 44 and dividing by 12 (respectively the molecular and atomic weights of CO_2 and C), we obtain 14.89 t CO_2e.ha^{-1}.yr^{-1}. Note that production is equal to the difference between photosynthesis and respiration. Therefore, the balance closes with 79 or 86 % of accuracy. This means that production is higher than the sum of the outputs. For NG, we did not have production figures, and therefore we estimate that NG produce half of the biomass as SBPPRL, since the stocking rate is also half. Balance closure is lower for NG, at around 62 – 68 %.

The simplified version of the nitrogen balance is

$$P_t^N = \Delta N_{SOM} + F_l^{N_2O}. \quad (3.18)$$

According to Rodrigues *et al.* (2010), the C:N content of SOM in SBPPRL is around 10. The average SOM_N in the first four years is then 255 or 284 kg $N.ha^{-1}.yr^{-1}$. N emissions from legumes are 0.8 t $CO_2e.ha^{-1}$, which is equal to 3 kg N_2O-N. The sum of both is about 258 or 284 kg $N.ha^{-1}.yr^{-1}$. According to Table 21, the average N in SBPPRL plants is 184 kg $N.ha^{-1}.yr^{-1}$. This means only 44 or 60 % accuracy. For NG, we used the same assumption for half of the production with the same N content. The percentage of closure is higher, at 88 – 100 %. However, for both grassland systems, output is higher than estimated N in biomass.

Table 25 – C and N balances for SBPPRL and NG.

C balance (t $CO_2e.ha^{-1}.year^{-1}$)			N balance (kg $N.ha^{-1}.year^{-1}$)		
Term	**SBPPRL**	**NG**	**Term**	**SBPPRL**	**NG**
$C_K - \alpha C_{SOM}$ (first 4 years)	9.33 - 10.42	3.38 - 3.77	ΔN_{SOM} (first 4 years)	255 - 284	92 - 103
$F_{EF}^{CH_4}$	1.40	0.70	$F_l^{N_2O}$	3	0
$F_{r,LW}^{CO_2}$	1.00	0.50			
$C_K - \alpha C_{SOM} + F_{r,LW}^{CO_2} + F_{EF}^{CH_4}$	11.73 – 12.82	4.58 - 4.97	$N_k + N_G + N_{LW} + F_l^{N_2O}$	258 - 284	92 – 103
$P_t^{CO_2} = F_P^{CO_2} - F_{r,plant}^{CO_2}$	14.89	7.44	P_t^N	184	92
% closure	**79 – 86 %**	**62 – 67%**	**% closure**	**(-) 44 – 60%**	**(-) 88 - 100%**

C – Carbon; N – Nitrogen; CO_2 – Carbon dioxide; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; NG – Natural Grasslands; SOM – Soil Organic Matter; LW – Live Weight; r – Respiration; EF – Enteric Fermentation; G – Groundwater; l – Legumes; CH_4 – methane; N_2O – nitrous oxide; (-) output higher than production.

We can also calculate the balance for each sub-system and see where the closure error comes from. We only show results for SBPPRL.

The carbon balance for aboveground biomass states that production ($P_a^{CO_2}$) is equal to the sum of carbon loss via leaf senescence (C_s) and ingestion by animals (C_i), which in turn is equal to 1/0.7 C_i. Average aboveground production is equal to total production divided by 1 + R:S. We obtain 3.9 t $CO_2e.ha^{-1}.yr^{-1}$. C_i is equal to 3.2 t $CO_2e.ha^{-1}.yr^{-1}$ (82 % balance closure). This indicates that average production in SBPPRL may not be completely consumed by grazing animals (and thus a higher sustainable stocking rate is possible), or that the intake is higher than estimated in SBPPRL.

The nitrogen balance for aboveground biomass states that aboveground N is equal to N lost by leaves senescence and ingestion by animals. Assuming a C:N of biomass also equal to 25 (standard for grasses and legumes), then the sum of both outputs is equal to 183 kg $N.ha^{-1}.yr^{-1}$, while the average N in SBPPRL plants is 184 kg $N.ha^{-1}.yr^{-1}$ (100% closure).

The carbon balance for aboveground biomass states that production ($P_b^{CO_2}$) is equal to SOM accumulation in soils due to incorporation of roots (K_{roots}). Belowground production is equal to aboveground production multiplied by the R:S, and therefore is 10.9 t $CO_2e.ha^{-1}.yr^{-1}$. However, K_{roots} is not obtained directly, and can only be calculated using the balance of the soil sub-system in Equation (3.4), which states that K_{roots} is equal to the overall SOM yearly accumulation minus the C entry from leave senescence and faeces. In that case, K_{roots} = 5.43 – 6.52 t CO_2e.ha^{-1}.yr^{-1} (58 – 63 % closure). This indicates that belowground production may be overestimated (R:S used is too high), or that SOM storage is underestimated.

The nitrogen balance for belowground biomass also requires N_K, the N output to soil, to be determined using the balance to soils, as the N stock is SOM minus the N entering soils from senescence, faeces and the N emitted as N_2O. The balance of these terms is 125 – 157 kg N.ha^{-1}.yr^{-1}. In the belowground balance, this value must be equal to N in belowground production, which is obtained as N in aboveground production times the R:S, and therefore is 516 kg N.ha^{-1}.yr^{-1} (24 – 30 % closure). Note that this balance assumes that all N is stocked in SOM, since we did not account any more gases leaving the system.

Regarding the balance from animals, removing the null terms, it states that the ingested C is either respirated, emitted as CH_4 or expelled as faeces. As indicated before, C_i is equal to 3.2 t CO_2e.ha^{-1}.yr^{-1}. The sum of the other terms is equal to 4.2 t CO_2e.ha^{-1}.yr^{-1} (76% closure), which means that either ingestion is underestimated, or emissions are overestimated.

Finally, the N balance from livestock states that N ingestion is equal to N in faeces. Since N_i = 183 kg N.ha^{-1}.yr^{-1}, and N_f = 72 kg N.ha^{-1}.yr^{-1}, there is only 39 % closure. This means that not all terms were accounted, and ingested N is emitted or expelled from the animals in other forms.

3.2.2.9. Discussion of results for the carbon balance

In this section we calculated the carbon balance of two grassland systems in Portugal, namely NG and SBPPRL. For SBPPRL, we obtained an average carbon sink (in 10 years) of 1.55 – 2.13 t CO_2.ha^{-1}.yr^{-1}, depending on the mineral bulk density (1.25 or 1.40 g.cm^{-3}). NG also have a significant sink potential of 0.53 – 0.75 t CO_2.ha^{-1}.yr^{-1}. The fertilized NG system was not consider here, since it is an intermediate situation between NG and SBPPRL.

The sink potential is mainly due to carbon sequestration and storage as SOM. The yearly sink may be as high as around 15 t CO_2.ha^{-1} for SBPPRL. During this stage, NBP is negative, which means that more production is stored than exported. However, the sink effect phases out as SOM pools stabilize, and the NBP becomes positive. After around 8 years, pastures become emitters. Since the stocking rate is higher and legumes emit more, after eight years NG have a lower carbon balance than SBPPRL. Still, it is important to notice that we used one hectare as the functional unit in the calculation, since it is also the base for Kyoto reporting. If we had used kilogram of meat produced, quilogram of protein in feed, or Euro of meat sold, results could change. SBPPRL are richer in legumes than NG, and provide a better quality feed to livestock.

Even though the functional unit used here is conservative, after 10 years SBPPRL are almost at 3% SOM. Soils with such high SOM concentration are more productive and farms benefit from overall environmental effects (decreased soil erosion, increased water holding retention). From that point on, those other agronomic, economic and environmental benefits of the systems may justify is maintenance for a longer term.

We obtain two sets of values, one for each soil MBD. The lowest MBD (1.25 $g.cm^{-3}$) is the indicative in Portugal (Maria de Fátima Calouro, personal communication), but soils which are not tilled are more compact, and thus MBD is likely higher (1.40 $g.cm^{-3}$) (Mário Carvalho, personal communication). Unpublished results (Carlos Aguiar, personal communication) from tests in one of the SBPPRL plots in the sample, namely Quinta da França, show an average MBD of 1.55 $g.cm^{-3}$, with a standard deviance of 0.09 $g.cm^{-3}$ (results from samples ranging from 1.37-1.82 $g.cm^{-3}$). More compact soils are usually problematic, particularly in the upper layer due to livestock stomping (Greenwood *et al.*, 2001). In those conditions, a rigid layer is created that imposes a barrier for plant growth and water infiltration (Bot and Benitez, 2005). However, in these grasslands, plants are annual, and the root system renews every year, leaving aeration channels which maintain a macropore structure in soils crucial for production (Drewry *et al.*, 2007). Furthermore, field work done in the context of Project Extensity showed that SBPPRL increase soil biodiversity, particularly macrofauna (Henriques *et al.*, 2006). Earthworms and insects create holes from which water and other organic matter penetrates the soil, which balances the effect of increased MBD (Bot and Benitez, 2005).

The calculations made for the C and N balances of the overall system and of each sub-system (aboveground and belowground plant, soil and animal), showed that the percentage of balance closure is usually higher than 50 % but significantly lower than 100 %, and higher for C than for N balances. This is mainly due to the fact that we did not include all N emissions to the atmosphere. We also did not use a grassland-specific R:S, turning instead to the literature. But the R:S used (2.8) seems to be too high, which was the cause for some of the balances not closing. In general, the limitations in our approach seem to be conservative for SBPPRL, since results suggest that the sustainable stocking rate can be higher or animals can eat more, and emissions are overestimated. Results are summed up in Table 26.

Table 26 – % of closure for C and N balances for SBPPRL, and main explanations.

Balance (SBPPRL)	Closure (%)	Likely explanation(s)
C – overall	79 – 86	Production overestimated, or either SOM storage or emissions underestimated
N – overall	(-) 44 – 60	N in biomass underestimated
C – aboveground	82	Not all production is consumed, and thus a higher stocking rate is possible, or C intake is underestimated
N – aboveground	100	-
C – belowground	58 – 63	Belowground production overestimated (R:S too high), or SOM storage underestimated
N – belowground	24 – 30	There are many more emissions from soil (NH_3, N_2, etc.) which were not included
C – soil	-	Cannot be calculated (no measured K_{roots})
N – soil	-	Cannot be calculated (no measured N_K)
C – animal	(-) 76	Ingestion is underestimated, or emissions are overestimated
N – animal	39	Ingested N is emitted or expelled from the animals in other forms

C – Carbon; N – Nitrogen; CO_2 – Carbon dioxide; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; (-) output higher than production.

Regarding the unaccounted effects in the balance, two parameters were left out of our study. First, even though there are no harvests for forage, in a natural system, there is some invasion by shrubs, which are intermediate stages in natural succession. Tillage is the most widely used system of shrub control in NG (Pinheiro *et al.*, 2008). During tillage, all shrub biomass in incorporated in soils, thus never leaving the plot. Therefore, for NG, $F_{harvest} \approx 0$ (but SOM accumulation is soils is also minimized).

However, in well managed SBPPRL, shrubs never grow, and therefore there is a loss in organic matter which corresponds to shrub production. We consider that $F_{harvest}$ in SBPPRL is equal to the aboveground production (since belowground production in NG is captured by SOM increase) of woody biomass removed during tillage in NG. Ehleringer and Mooney (1983) estimated aboveground net primary productivity of shrubs as 1.1 – 1.3 $t.ha^{-1}.yr^{-1}$, while Castro and Freitas (2008) find values around 1.9 – 2.7 $t.ha^{-1}.yr^{-1}$. As an intermediate value, we may consider $F_{harvest} = 2\ t.ha^{-1}.yr^{-1}$.

The second effect regards losses that may occur in NG. We considered NG as unmanaged areas. However, two important disturbance events may occur in NG: tillage for shrub control, or forest fires. Tillage would mineralize most of the SOM accumulated. If it did not occur, forest fires could take place. During a forest fire, large scale emission of greenhouse gases occurs. This is likely in NG, but not in SBPPRL, because, as stated before, shrub growth is limited. Therefore, two effects deter fires in SBPPRL, namely:

- The grazed agro-forestry mosaic landscape interrupts large areas of forest monocultures prone to fire;
- Unlike in NG, increased grazing pressure in SBPPRL stops invasion by shrubs, which are also good fuel for forest fires.

We also did a farm scale analysis, which did not take into account life cycle effects. One important effect regards feeds. Instead of increasing stocking rates, farmers who install SBPPRL may opt to transfer steers from intensive to extensive production, thus decreasing stable periods and feed consumption. Van der Werf *et al.* (2005) indicate an average emission for pig feed of 528 kg $CO_2e.t^{-1}$ feed. Their value is close to the 498 kg $CO_2e.t^{-1}$ feed found by Blonk *et al.* (1997) and the 591 kg $CO_2e.t^{-1}$ feed found by Carlsson-Kanyama (1998), also for pig feed. Casey and Holden (2005) consider the emission of 1156 kg $CO_2e.t^{-1}$ feed for lactating cows, which need more protein-intensive ingredients (thus produced with more fertilizers). Therefore, emissions from feeds are not neglectible, and are the reason for one of the next Sections of this Chapter, in which we determine the life cycle impacts of pastures.

One important assumption regards N_2O emissions. These emissions from soils change as legume percentage changes, as well as soil C:N ratios (Bot and Benitez, 2005; Ostle *et al.*, 2009). Legume percentage is known to be higher in the first years in SBPPRL, but C:N ratios decrease in pastures (Rodrigues *et al.*, 2010). These two effects have contradictory results in N_2O emissions, and the overall balance is unknown. Besides, calculations made for N balances suggest that N_2O emissions may be overestimated.

Some authors have already claimed that N_2O emissions, estimated using typical emission factors from IPCC (1997), could be overestimated. Li *et al.* (2005), modelled carbon sequestration enhancement strategies, and found that carbon dynamics influences nitrogen dynamics. They estimate that, when SOM increases, the increase in N_2O emissions diminishes or even eliminates the carbon sequestration. However, empirical studies like Crews and Peoples (2004), Kammann *et al.* (1998), Ledgard (2001) or Rochette and Janzen (2005) state that emission factors are systematically overestimated, and find values close to the lowest extreme of the interval presented here. These studies refer to mixes of grass and legumes, and therefore are a valid approximation of the system studied here. In equilibrium, it is very likely that fixed nitrogen is fully consumed by grasses.

But even if emissions were of the magnitude of Li's (Li *et al.*, 2005), these emissions may be offset by their alternative. Nitrogen fixed by legumes is a direct substitute of synthetic nitrogen in fertilizers. To achieve close productivities without legumes, pastures would require nitrogen fertilization. The other option would be to use commercial feed for cattle, but then fertilized crop production would be required. Therefore, we may compare the emissions due to biological N fixation to the emissions due to fertilizer production and use.

Considering the emission factor in IPCC (1997) stated above, each fixed N unit corresponds to 3.88 kg $CO_2e.ha^{-1}$ emitted. We found on the database of the Life Cycle Assessment software SimaPro 6.0 the CO_2 emissions for which fertilizers are responsible during their production. We found that for each kg of N in the fertilizer urea 3 kg of CO_2e are emitted, which is about the same as what is emitted by legumes. But for each kg of N in the fertilizer ammonium nitrate, 7.75 kg of CO_2e are emitted, and this is about the double of the emissions due to biological fixation. Therefore, each unit of nitrogen is used more efficiently if it is biological rather than synthetic.

It is important to remark that, for Kyoto accounting purposes, the effects mentioned here contribute to different items. Carbon sequestration is accounted under the optional Article 3.4, "Grassland Management". Emissions from legumes, liming and livestock are reported for the agricultural sector, while feed and fertilizer consumption are reported under industrial activities. Furthermore, some effects deal with international transfers. For instance, if the increase in stocking rate would substitute meat imports and decrease the Portuguese meat deficit, the corresponding emissions would now be reported by Portugal, even if substituted meat was produced intensively. This is because Kyoto agreements place the entire burden of emissions on producers. It is crucial in future climate negotiations to account for implicit emissions in trading and develop a fair indicator of the allocation of CO_2 emissions between producers and consumers. Preliminary work on what a fair indicator would be has been done, for example, by Rodrigues *et al.* (2006).

A final word goes to emissions from livestock. In the previous results, we were subjected to the experimental setting of the AGRO projects. Since the AGRO projects were set at a plot scale, there was an effective increase in stocking rate. But it is not plausible that it is so when we move to bigger scales. At the country scale, change from NG to SBPPRL implies maintenance in stocking rate, and decrease in feed consumption by cows. This scenario is the most likely. Table 27 shows that practically all breeding cows in Portugal are subsidized, and so, if animal quotas are maintained, then there can be almost no global stock rate increase.

Table 27 - Registered (INE[32]) and supported (IFADAP/INGA, 2004) breeding cows in Portugal.

Year	Registered animals	Supported animals	% of Supported Animals
1998	341 000	321 948	94.4
1999	342 000	303 700	88.8
2000	342 000	307 093	89.8
2001	351 000	307 731	87.7
2002	359 000	321 978	89.7
2003	371 000	332 243	89.6
2004	384 000	365 050	95.1

Our results would also change if, instead of using the NIR emission factors, we used factors from the IPCC (1997). Unlike the NIR, the IPCC considers that faeces also emit CH_4 and N_2O. Therefore, in our first scenario, we start from a degraded grassland with a stocking rate of 0.5 LU, and then introduce 0.5 LU of breeding cows after the grassland is sown. Corresponding emissions would rise by 1.1 t CO_2 eq.ha^{-1}.yr^{-1}, as shown in Table 28.

Table 28 – Emissions from breeding cows in pastures.

Gas	Emission factors		Stocking rate	Emissions	
	Enteric fermentation	Faeces[33]		Enteric fermentation	Faeces
	kg.head^{-1}.yr^{-1}	kg.head^{-1}.yr^{-1}	head.ha^{-1}	kg.ha^{-1}.yr^{-1}	kg.ha^{-1}.yr^{-1}
CH_4	73[34]	2.156[35]	0.5	36.5	1.078
N_2O	0[36]	1.927[37]	0.5	0	0.964
CO_2e				766.5	321.478

CH_4 – Methane; N_2O – Nitrous oxide; CO_2e – Carbon dioxide equivalent.

For Portugal, this is the worst scenario possible of increased animal stocking rate, and it represents only about 20% of total carbon sequestration. Therefore, global carbon balance would still be very favourable, and SBPPRL would still be carbon sinks. Considering that meat demand in Portugal does not change by implementing such policies, this would represent a transfer: instead of buying meat produced elsewhere, Portugal would produce it locally. Therefore, in terms of global world emissions, they would remain the same.

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[32] http://www.ine.pt/prodserv/quadros/quadro.asp

[33] Figures in this column have excessive precision. However, such are the exact figures provided by APA, and so we opted to maintain them as such.

[34] Enteric fermentation CH_4 emission factor for breeding cows in grasslands, in 2004.

[35] Faeces CH_4 emission factor for breeding cows in grasslands, in 2004.

[36] IPCC (who establishes the Kyoto accounting method) does not consider N_2O emissions from enteric fermentation.

[37] Faeces N_2O emission factor for breeding cows in grasslands, in 2004.

Finally, a third option would be to assume that there is a transition from a stocking rate of 0.5 $LU.ha^{-1}$, composed only by breeding cows, to a stocking rate of 1.0 $LU.ha^{-1}$ where for each cow, a steer is being fed and finished during a year (from 6 to 18 months). There are three major effects:

1. Breeding cows' population increases from 0.5 $LU.ha^{-1}$ to 0.625 $LU.ha^{-1}$.
2. Number of steers being fed in the pasture increases from 0 $LU.ha^{-1}$ to 0.375 $LU.ha^{-1}$, corresponding to an increase of 0.625 $steer.ha^{-1}$ (since 1 steer represents 0.6 LU).
3. Steers are withdrawn from intensive feeding, and so emissions corresponding to 0.625 $steer.ha^{-1}$ are avoided.

Effects 1, 2 and 3 are quantified in Table 29. Table 29 shows that global balance is 176 kg $CO_2e.ha^{-1}$ emitted. This value is minor when compared to carbon sequestration by SBPPRL. Detailed calculations are shown in Appendix II – Estimation of CO_2e emissions from livestock.

Table 29 – Effect on greenhouse gases' emissions of the stocking rate increase

Gas	Effect	Emission factors		Stocking rate	Emissions	
		Enteric fermentation	Faeces/ Manure		Enteric fermentation	Faeces/ Manure
		$kg.head^{-1}.yr^{-1}$	$kg.head^{-1}.yr^{-1}$	$head.ha^{-1}$	$kg.ha^{-1}.yr^{-1}$	$kg.ha^{-1}.yr^{-1}$
CH_4	1	73	2.156	0.125	9.125	0.270
	2	50.2	0.679	0.625	31.375	0.424
	3	-50.2	-1.156	0.625	-31.375	-0.723
N_2O	1	0	1.927	0.125	0	0.241
	2	0	0.659	0.625	0	0.412
	3	0	-1.122	0.625	0	-0.701
CO_2e	1			0.125	191.625	80.331
	2			0.625	658.875	136.593
	3			0.625	-658.875	-232.560
Total (kg CO_2e)					191.625	-15.636

CH_4 – Methane; N_2O – Nitrous oxide; CO_2e – Carbon dioxide equivalent.

In this case, Portugal would have a benefit from switching from an intensive to an extensive production system. In terms of global effects, since transfers would remain domestic, there would also not be a significant effect. But the overall conclusion remains the same: SBPPRL are still a carbon sink.

As we have shown, the whole SBPPRL system is a sink. But is it a more powerful sink than other possible land uses? We now answer this question when we refer available data for carbon sequestration in uses such as agriculture and forests.

3.3 Comparison with other agricultural land uses for carbon sequestration[38]

Portugal will have to report, in the context of Article 3.4 of the KP, the carbon balance of all land uses. One methodological distinction will be made between areas under specific management activities, and unmanaged areas. Two specific activities will be reported. The first one is SBPPRL, under the grassland management item, and the second one is no-tillage, under the cropland management item.

3.3.1 No-tillage

Like SBPPRL, the use of no-tillage as a specific management activity has been pinpointed in the literature as having a substantial potential for carbon sequestration. No-tillage is particularly important for annual crops (Basch, 2002).

To assess the potential of no-tillage for carbon sequestration, we start by presenting next a small literature review of available studies, to frame the figures obtained next. We then estimate the potential for carbon sequestration using the (default) IPCC method. Finally, national data is used to conclude the final figures to be used for national accounting.

3.3.1.1. Literature review

Table 30 shows a short literature review on carbon sequestration from no-tillage in annual crops. Figures found in these studies range from 0 to 3 t $CO_2.ha^{-1}$ (ECCP, 2003).

Table 30 – Literature review of available studies on carbon sequestration from no-tillage.

Reference	Study method	Carbon sequestration (t $CO_2 \cdot ha^{-1} \cdot yr^{-1}$)
ECCP, 2003	Survey	0-3.0
Six *et al.*, 2004	Survey	0.8
Six *et al.*, 2004	Survey	0.4
Smith, 2004	Survey	1.4
Cambardella and Elliott, 1992	Medição	1.22
Six *et al.*, 2002	Survey	1.2 +- 0.4
West and Post, 2002	Survey	0.21 +- 0.05
Freibauer *et al.*, 2004	Survey	0.6
Bernacchi *et al.*, 2005	Measurements (*Eddy covariance*)	2.2
Marland *et al.*, 2003	Survey	0.34
Marland *et al.*, 2004	Survey	0.57 +- 0.14

3.3.1.2. Calculations using the IPCC default method

The IPCC (2006) has a method to calculate the carbon sequestration potential of no-tillage using the expression

[38] In this sub-section, we do not consider carbon sequestration from forest management. Carbon storage in aboveground biomass is important for forests, but not for the rest of the agricultural land uses, which makes them hardly comparable. However, and as a curiosity, according to Pereira *et al.* (2009a), cork oak *montados* sequester around 1-5 t $CO_2.ha^{-1}.yr^{-1}$, pine forests sequester around 15-26 t $CO_2.ha^{-1}.yr^{-1}$, and eucalyptus forests sequester around 15-32 t $CO_2.ha^{-1}.yr^{-1}$.

$$\Delta C_i = \Delta C_{ABi} + \Delta C_{BBi} + \Delta C_{DWi} + \Delta C_{LLi} + \Delta C_{SOi} + \Delta C_{HWPi}, \qquad (3.24)$$

where ΔC_i is the change in carbon stock for soil use *I*, and the subscripts have the following meanings:

AB – Aboveground biomass, assumed zero for non-woody crops;

BB – Belowground biomass, assumed zero for non-woody crops;

DW – Dead woody biomass, assumed zero for non-woody crops;

LI – Litter, which is biomass encompassed in soils by the death of plants and management actions;

SO – Soil carbon;

HWP – Biomass removed as woody products, assumed to be zero for non-woody crops.

Therefore, for no-tillage of annual (non-woody) crops, Equation (3.24) becomes

$$\Delta C_{NT} = \Delta C_{SO_{NT}}, \qquad (3.25)$$

where the soil carbon balance is calculated by the following expression

$$\Delta C_{SO} = \frac{C_t - C_0}{D}. \qquad (3.26)$$

In Equation (3.26) *C* represents soil organic carbon (SOC), *t* is the inventory year (under no-tillage), and *0* represents the baseline year (conventional tillage). *D* is the reference time period for the change in SOC from C_0 to C_t. *C* is calculated using

$$C = C_{REF} \bullet F_{LU} \bullet F_{MG} \bullet F_I, \qquad (3.27)$$

where:

C_{REF} – reference carbon stock in managed soils; for no-tillage, we assumed the value 18.8 t $C.ha^{-1}$ (Fátima Calouro, comunicação pessoal).

F_{LU} – stock change factor for land-use systems or sub-system for a particular land-use, dimensionless; for annual crops, the figure is 0.80 (IPCC, 2006);

F_{MG} – stock change factor for management regime, dimensionless; for conventional tillage, the value is 1.00, and for no-tillage 1.10 (IPCC, 2006);

F_I – stock change factor for input of organic matter, dimensionless; we considered that when crop residues are left on the field, this parameter is *medium* (1.00) or *high* (1.37), and it is *low* (0.95) otherwise.

Note that all equations mentioned before are applied on a hectare basis.

We thus estimate, from Equations (3.24) to (3.27), carbon sequestration from no-tillage as 0.26 t $CO_2.ha^{-1}.yr^{-1}$, in case no crop residues are left on the field. If residues are left, the value rises to 0.28 or 0.38 t $CO_2.ha^{-1}.yr^{-1}$ (medium or high input of SOM).

3.3.1.3. Calculations using data from the Évora University

However, an analysis of existing data for Portugal shows that this value may be underestimated. Most studies were conducted in research projects by the Évora University, and all refer to rainfed crops.

Data for Herdade da Revilheira in Évora, Portugal, (Carvalho and Basch, 1995) shows that in no-tilled crops, leaving residues on the field, SOM increases by of about 0.47 percent points in the first 30 cm and in five years (Table 31). The test took place in a plot field of 2.8 ha in a four year rotation. The rotation was sunflower – wheat – forage – triticale. Sunflower was replaced in some plots during the test, due to the fact that it was highly attacked by birds (Mário Carvalho, personal communication).

Table 31 – SOM change (1999-2004) in no-tilled luvisoil areas in Herdade da Revilheira, leaving residues on the field (Carvalho and Basch, 1995).

Depth (cm)	Increase (%)
0-10	0.86
10-20	0.42
20-30	0.15
0-30 (average)	0.47

SOM – Soil Organic Matter.

Other results are shown in Table 32, and confirm that SOM increases by about 0.1 percent points per year in 30 cm in no-tilled plots, when compared with conventional tillage. Increases below that depth occur due to Summer cracks, which allow residues to fall to the interior of the layer.

Table 32 - SOM concentration and soil respiration (mineralization) in a cromic vertisol (Almocreva, Barros de Beja, Portugal) after 8 years under different tillage systems.

Depth (cm)	SOM concentration (%)		Soil respiration ($mgCO_2.10g^{-1}$ soil)	
	NT	CT	NT	CT
10	2,53	1,91	3,17	0,69
20	2,15	1,67	3,87	2,88
30	2,25	1,62	5,80	3,70
40	2,22	1,33	-	-

SOM – Soil Organic Matter; CO2 – Carbon dioxide; NT – No-Tillage, CT – Conventional Tillage. Source: Carvalho and Basch (1995).

Soils in which all crop residues are removed have expectedly lower SOM concentrations, since without cover soil respiration (another term used for the SOM mineralization rate) increases. Available studies corroborate this statement, as shown in Table 33. In four years, no-tilled soils show higher SOM concentrations of about 0.1 percent points (or 0.025 percentage points per year) in relation to other tillage methods. Note that SOM concentration is higher in the 20-30 cm layer for conventional tillage due to the burying of organic components by the plow. Note however that, in the case of these trials, the rotation did not include a forage stage.

Table 33 - SOM concentration and distribution in a luvisoil area, after four years of tillage.

Depth (cm)	SOM concentration (%)			
	NT	R	Chi	CT
0-10	1.70	1.50	1.60	1.30
10-20	1.30	1.40	1.20	1.30
20-30	1.00	0.90	0.90	1.10
0-30 (average)	1.33	1.27	1.23	1.23

SOM – Soil Organic Matter; CO2 – Carbon dioxide; NT – No-tillage; R – Ripper; Chi – Chisel; CT – Conventional tillage.
Source: Carvalho, unpublished.

For winter crops, Carvalho *et al.* (2002) also determined the SOM concentration in soils under no-tillage for three years, in the 0-30 cm layer, and four different quantities of straw left on the fields. Data obtained was used to calibrate a statistical model to relate SOM concentration and the quantity of residues left (*R*, measured in tons), which is,

$$SOM\,(\%) = 0{,}89 + 0{,}13R(t)\,. \qquad (3.28)$$

From this expression, we may extrapolate that in a situation without residues, SOM concentration is 0.89%. For each ton left, SOM increases by 0.13 percent points.

We may assume that, even when all straw is removed, 1 $t.ha^{-1}$ of residues always stay on the field (Mário Carvalho, personal communication). The average straw produced for oats, wheat and triticale is around 2.2 $t.ha^{-1}$ (average straw productivity from the GPP (2001) database). Therefore, the difference between leaving straw on the field is 1.2 $t.ha^{-1}$. From Equation (3.28), this represents a 0.156 percent points difference in SOM concentration. Since the parameters in the equation were calculated for a three year period, this is equivalent to 0.052 percent points per year.

Considering the equivalences in Section 3.1, and assuming the 30 cm depth, the yearly increase of 0.025 percent points which corresponds to no-tillage without residues left on the field is equivalent to the sequestration of 1.9-2.1 t $CO_2.ha^{-1}$. Similarly, the increase of 0.1 percent points from no-tillage with residues is equivalent to 7.7-8.5 t $CO_2.ha^{-1}$. The difference between these two situations is, then, of 5.8-6.4 t $CO_2.ha^{-1}$. All these figures are much higher than those obtained from the IPCC (2006) method shown before.

We use results from Carvalho *et al.* (2002) to confirm the previous ones. As shown before, the difference between the case with and without straw was 0.052 percent points per year, which is equivalent to 4.0-4.4 t $CO_2.ha^{-1}$, which is similar to, but still 30 lower than, the value calculated above. Note that this previous value is, however, subjected to higher uncertainty, since we had to look at a different source to determine typical straw production.

3.3.1.4. Results and discussion

The analysis done has allowed us to obtain the following conclusions regarding the carbon sequestration potential of no-tillage:

1. Without residues left on the field – 1.9-2.1 t $CO_2.ha^{-1}.yr^{-1}$;
2. Leaving residues on the field – 7.7-8.5 t $CO_2.ha^{-1}.yr^{-1}$.

We again note that these values are yearly averages of a four year rotation where, in the second case, soil cover in one year was a forage crop, and therefore no crop residues are produced. Hence, the situation in that year is similar to the one where no residues are left on the field.

Regarding the sequestration factor to use for each crop, there are several possibilities, which result from combining the following conditions:

- Whether it is a Spring/Summer crop, a Fall/Winter crop, or a fallow;
- Whether it is a rainfed or an irrigated crop.

Typical Fall/Winter crops are wheat and triticale. The majority of farmers sell straws from these crops, and therefore only low-quality residues remain on the field (Mário Carvalho, personal communication). Carbon wise, we are in situation 1 above.

The main rainfed Spring/Summer crop is sunflower, and the main irrigated Spring/Summer crop is maize. Grain maize residues always remain on the field, since grain maize straw has no commercial value. Carbon wise, we are in situation 2 above. Silage maize, however, is a different story, since the whole plant is harvested, and so we are in situation 1.

Since all test results shown before were obtained in rainfed plots, the question arises of which sequestration factor to use for irrigated crops. There are two reasons why we can believe that irrigated and rainfed crops are different:

- Irrigated areas are more productive, with more root formation, and hence more organic matter enters the soil;
- Since there is water available in the hot seasons, the conditions for higher soil respiration are gathered, hence decreasing SOM accumulation.

Since there are no know systematic studies of irrigated crops, we considered that the first effect is predominant (Mário Carvalho, personal communication). Therefore, the use of the rainfed sequestration potential is a conservative option for irrigated areas.

Therefore, Table 34 shows a summary of which factores to use in each combination of cases.

Table 34 - Carbon sequestration factors for no-tillage.

Crop	Sequestration factor (t CO_2.ha^{-1}.yr^{-1})			
	Spring/Summer crop	Fall/Winter crop with straw	Fall/Winter crop without straw	Fallow
Rainfed	1.9-2.1	7.7-8.5	1.9-2.1	1.9-2.1
Irrigatd	7.7-8.5	7.7-8.5	1.9-2.1	1.9-2.1

Note also that the studies indicated here were done on luvisoils and vertisols. We can admit that, in other soil types, results will be different. However, a few considerations are in order:

- In soils with higher clay content, the use of a luvisoil sequestration factor may be conservative, since these are usually more productive and therefore expectedly will have higher SOM increases.

- In sandy soils, the use of a carbon sequestration potential from luvisoils may be an overestimation. However, due to their frequent low productivity, hardly any crops are produced in sandy soils in Portugal.

3.3.2 Other land uses

For other land uses, we studied whether there is SOC stock change for cropland remaining cropland, for each crop. Since other crops are either (1) permanent or (2) annual with soil mobilization, it may be argued that there is no significant carbon stock change. In the first case, there is no SOC accumulation because the long-term SOM equilibrium is already reached. In case (2), there is also no accumulation because periodic soil mobilization creates the conditions for SOC inputs to deep layers of soil to be mineralized quickly.

To assess if these statements are true, we used two methods: a Paired Samples T Test, and a time series model to determine the cointegration order of all sample series. We explain both briefly next.

3.3.2.1. Method

We intended to study whether SOC remains constant in Portuguese croplands that remain croplands. For some available trials there were only two sampling years, and for others there were as many as twelve. Two methods were used:

1. For samples consisting on two or more years, we study if the average soil organic matter in all sampling years is equal (or, to be accurate, if we cannot statistically reject the hypothesis that averages vary significantly amongst sampling years);
2. For samples consisting on more than two years, determine if the time series of measured SOC is stationary (which means that there is no upward or downward trend).

In the first case, we used a Paired Samples T Test. This test consists on determining the difference between the average values of two series, and determining if that difference is statistically different from zero (given the standard deviation of each sample). Therefore, we compute the average SOC for any two years of a given sample; if both average values are equal, then we assume that there was no change in soil carbon stocks between the two years.

This test is only valid if both series are normally distributed. Therefore, we started by performing a Kolmogorov-Smirnov test on all series. This tests for a normal distribution.

The Paired Samples T Test is especially helpful when there are only two sampling years, since it is used to compare the average value of any two series. When there are more than two sampling years, we ran the test coupling all years in pairs. Therefore, we tested all possible combinations.

However, the Paired Samples T Test has some serious limitations. First of all, it requires series to have the same number of observations (since the difference between observations is calculated first, and only then the average of the differences). When the number of observations is different, some must be discarded. Therefore, not all available information is used.

Besides, this test is too restrictive. All years must be compared in pairs, and so it tests if averages are constant for all pairs (perfectly stationary series). However, SOC may vary significantly between particular years and, in the long term, the times series may still be stationary.

Therefore, we used a second method whenever there was a time series of more than two sampling years. We used the Box-Ljung statistics to determine the integration level of all series. To say that a series is stationary (statistically constant in time) is the same as to say that it is "zero order integrated". A "first order integrated" series has a linear trend, as so forth.

However, we needed a time series with all sampling years between the first and the last. Therefore, we linearly interpolated all missing values.

We used the statistical software SPSS 16.0.

3.3.2.2. Data

Data used were the result of two decades of soil sampling and analysis in several test sites. All trials were made by the former Laboratório Químico Agrícola Rebelo da Silva (LQARS). Soil samples were collected at two depths: 0-20 cm and 20-40 cm (for annual crops) or 20-50 cm (for permanent crops).

Data for SOC are structured as an incomplete time series for each crop in each site. Most series are incomplete since some sample years are missing. Soil use and available sample years for each test site are described in Table 35.

Table 35 – Trials done by LQARS, and available sample years for each type of land use.

Land Use	Species	Trial location	Trial #	Available sample years
Olive	"Verdeal Transmontana"	Mirandela	1	1987, 1990, 1993 to 1997
	"Cobrançosa"	Mirandela	2	1987, 1990, 1995
	"Galega"	Castelo Branco	3	1998 to 2002
	"Picual"	Santarém	4	1995, 1998
	"Cobrançosa"	Santarém	5	1987, 1989
	"Blanqueta"	Santarém	6	1989, 1995
	"Blanqueta"	Elvas	7	1988, 1991, 1998
	"Carrasquenha"	Elvas	8	1989 to 1999
Vineyard	"Bical"	Bairrada	9	1987, 1989, 1996
	"Castelão (Camarate)"	Bairrada	10	1988, 1990, 2000
	"Periquita (Castelão)"	Palmela	11	1988, 1989, 1992, 1996, 2000
	"Loureiro"	Vinhos Verdes	12	1987, 1990, 1992, 1996, 2000, 2004
Fall/Winter grains	Triticale	Viseu	13	1986, 1988 to 1990
	Triticale	Évora	14	1986 to 1988, 1990,1992
	Wheat	Coimbra	15	1985 to 1987
	Wheat	Serpa	16	1986, 1988 to 1992
Maize and rice	Rice	Coimbra	17	1986, 1990
	Maize	Estarreja	18	1984 to 1985, 1987 to 1991
	Maize	Aveiro	19	2002 to 2004
	Maize	Viseu	20	1980, 1982 e 1983, 1985
	Maize	Pegões	21	1988 to 1990
	Maize	Beira Interior	22	1981, 1983 to 1985
	Maize	V. Franca Xira	23	1987,1988,1989, 1990, 1991
	Maize	Chamusca	24	1985, 1987, 1989
	Maize	Elvas	25	1981, 1983, 1985, 1987

3.3.2.3. Results and discussion

Paired samples T-test

Results for the Paired Samples T Test are shown in Table 36. A test is inconclusive if there are more than two sampling years, and results differ for each pairing. They show downward or upward trends if, for available years, the average soil carbon levels decreased or increased in all pairings.

Results for annual crops (trials 13-25) mostly support that SOC does not change between years or, if so, there is an upward trend. Conclusions for perennial crops (trials 1-13) cannot be drawn from Table 36.

Table 36 – Conclusions on the stationarity of the soil carbon time series according to the Paired Samples T Test, per trial.

Trial #	Number of sampling years	Constant average?		If not, trend?		Conclusion
		0-20 cm	20-40/50 cm	0-20 cm	20-40/50 cm	
1	12	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
2	4	No	Inconclusive	Inconclusive	Downward	Non stationary
3	4	No	No data	Inconclusive	No data	Inconclusive
4	2	No	No	Downward	Downward	Downward trend
5	2	No	No	Downward	Downward	Downward trend
6	2	No	No data	Downward	No data	Downward trend
7	2	No	No	Upward	Upward	Upward trend
8	10	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
9	3	Inconclusive	No	Inconclusive	Downward	Inconclusive
10	3	No	No	Inconclusive	Inconclusive	Non stationary
11	5	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
12	6	Inconclusive	Inconclusive	Inconclusive	Inconclusive	Inconclusive
13	4	Inconclusive	No data	Inconclusive	No data	Inconclusive
14	5	No data	No data	No data	No data	Inconclusive
15	2	No	No data	Downward	No data	Downward trend
16	6	Inconclusive	No data	Inconclusive	No data	Inconclusive
17	2	No	No data	Upward	No data	Upward trend
18	7 (0-20 cm); 2 (20-40 cm)	Inconclusive	No	Inconclusive	Upward	Inconclusive
19	3	No	No data	Upward	No data	Upward trend
20	4	Inconclusive	No data	Inconclusive	No data	Inconclusive
21	3	Inconclusive	No data	Inconclusive	No data	Inconclusive
22	4	Yes	No data	-	No data	Stationary
23	4	No	No data	Upward	No data	Upward trend
24	3	Inconclusive	No data	Inconclusive	No data	Inconclusive
25	4	Yes	No data	-	No data	Stationary

Time series model

Results for time series modelling are shown in Table 37. A series is considered stationary if it is zero order integrated. If it is first order integrated, then it is possible to determine if the trend is up or downward.

Results confirm the claim that SOC does not change between years for annual crops, but are again fuzzy for perennial crops.

Table 37 – Conclusions on the stationarity of the soil carbon time series according to time series modelling, per trial.

Trial #	# sampling years	# interpolated years	Stationary series?		If not, trend?		Conclusion
			0-20 cm	20-50 cm	0-20 cm	20-50 cm	
1	12	1	Yes	Yes	-	-	Stationary
2	4	10	No	No	Downward	Downward	Downward trend
3	4	1	Yes	Yes	-	-	Stationary
4	0	0	No data	No data	No data	No data	Inconclusive
5	0	0	No data	No data	No data	No data	Inconclusive
6	0	0	No data	No data	No data	No data	Inconclusive
7	0	0	No data	No data	No data	No data	Inconclusive
8	10	1	Yes	Yes	-	-	Stationary
9	3	7	No	No	Upward	Downward	Not stationary
10	3	10	No	No	Downward	Downward	Downward trend
11	5	8	Yes	Yes	-	-	Stationary
12	6	12	No	No	Upward	Upward	Upward trend
13	4	1	Yes	No data	-	No data	Stationary
14	5	2	Yes	No data	-	No data	Stationary
15	0	0	No data	No data	No data	No data	Inconclusive
16	6	1	Yes	No data	-	No data	Stationary
17	0	0	No data	No data	No data	No data	Inconclusive
18	7	1	Yes	No data	-	No data	Stationary
19	3	0	Yes	No data	-	No data	Stationary
20	4	2	Yes	No data	-	No data	Stationary
21	3	0	Yes	No data	-	No data	Stationary
22	4	1	Yes	No data	-	No data	Stationary
23	4	0	Yes	No data	-	No data	Stationary
24	3	2	Yes	No data	-	No data	Stationary
25	4	3	Yes	No data	-	No data	Stationary

Comparison between approaches

Final results are shown in Table 38. The combination of results from both methods yields conclusive classifications for each trial. Then, conclusions for each crop are drawn from generally appreciating results for corresponding trials.

Table 38 – Conclusions on the stationarity of soil carbon in Portuguese soils, per crop.

<table>
<tr><th rowspan="2">Trial #</th><th colspan="2">Conclusion</th><th rowspan="2">Final conclusion, by trial</th><th rowspan="2">Crop</th><th rowspan="2">Final conclusion, by crop</th></tr>
<tr><th>T Test</th><th>Time Series</th></tr>
<tr><td>1</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td><td rowspan="8">Olive</td><td rowspan="8">Depends on the trial
Probably: Stationary</td></tr>
<tr><td>2</td><td>Not stationary</td><td>Downward trend</td><td>Downward trend</td></tr>
<tr><td>3</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>4</td><td>Downward trend</td><td>Inconclusive</td><td>Downward trend</td></tr>
<tr><td>5</td><td>Downward trend</td><td>Inconclusive</td><td>Downward trend</td></tr>
<tr><td>6</td><td>Downward trend</td><td>Inconclusive</td><td>Downward trend</td></tr>
<tr><td>7</td><td>Upward trend</td><td>Inconclusive</td><td>Upward trend</td></tr>
<tr><td>8</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>9</td><td>Inconclusive</td><td>Not stationary</td><td>Not stationary</td><td rowspan="4">Vineyard</td><td rowspan="4">Depends on the trial</td></tr>
<tr><td>10</td><td>Not stationary</td><td>Downward trend</td><td>Downward trend</td></tr>
<tr><td>11</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>12</td><td>Inconclusive</td><td>Upward trend</td><td>Upward trend</td></tr>
<tr><td>13</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td><td rowspan="4">Fall/Winter grains</td><td rowspan="4">Stationary</td></tr>
<tr><td>14</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>15</td><td>Downward trend</td><td>Inconclusive</td><td>Downward trend</td></tr>
<tr><td>16</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>17</td><td>Upward trend</td><td>Inconclusive</td><td>Upward trend</td><td rowspan="9">Maize and rice</td><td rowspan="9">Stationary/upward trend
Probably: Stationary</td></tr>
<tr><td>18</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>19</td><td>Upward trend</td><td>Stationary</td><td>Stationary/Upward trend</td></tr>
<tr><td>20</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>21</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>22</td><td>Stationary</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>23</td><td>Upward trend</td><td>Stationary</td><td>Stationary/Upward trend</td></tr>
<tr><td>24</td><td>Inconclusive</td><td>Stationary</td><td>Stationary</td></tr>
<tr><td>25</td><td>Stationary</td><td>Stationary</td><td>Stationary</td></tr>
</table>

Conclusions - olive

Conclusions are that, for olive, the stationarity of SOC is dependent on the trial. Note however, that trials that point to a downward trend in soil carbon stocks are:

- Trial 2 – the conclusion was obtained from 4 sampling years and 10 interpolated years. Since so many values were interpolated, results must be taken with caution;
- Trials 4, 5 and 6 – there are only 2 sampling years in each trial, and therefore conclusions must be influenced by the fact that we are comparing only two specific years. However, trial 7 yields an upward trend only based on 2 sampling years as well.

Therefore, the two most reliable trials are numbers 1 and 8. Both of them yield stationarity as the most likely conclusion.

Final conclusion: SOC is probably stationary, but results depend on the trial and number of sampling years.

Conclusions - olive

Conclusions are that, for vineyard, the stationarity of SOC is dependent on the trial. In this case, results that point for no stationarity cannot be dismissed easily, since they are corroborated by both methods.

Final conclusion: No conclusion may be drawn, since results depend on the trial.

Conclusions - Fall/Winter grains

In the case of Fall/Winter grains, there is only one trial (trial number 15) that yields a downward trend for SOC dynamics. However, this trial consists only of two sampling years, and therefore its conclusion is relatively weak and may be dismissed. All other trials point to stationarity.

Final conclusion: SOC is stationary.

Conclusions - maize and rice

In the case of maize and rice, all trials point to either stationarity of SOC accumulation. However, since stationarity is the conclusion for most trials, we consider that as the most likely result.

Final conclusion: SOC is either stationary (which is the most likely result) or increasing.

Average carbon values per crop

We have previously determined that the average values for olive, Fall/Winter crops and maize/rice are constant, regardless of sampling year. This means that SOC is constant for croplands remaining croplands with each type of crop.

Therefore, the average soil carbon stock for each crop may be obtained by calculating the average of all values for each crop. Results are shown in Table 39. Maize and rice have the highest SOC stocks, followed by olive. Fall/Winter grains have the lowest stocks.

Table 39 – Average Soil Organic Carbon stock per crop.

Crop	Depth (cm)	Soil organic carbon (%)
Olive	0-20	0.93
	20-50	0.64
Vineyard[39]	0-20	0.98
	20-50	0.92
Fall/Winter grains	0-20	0.54
	20-40	0.58
Maize and rice	0-20	1.54
	20-40	1.71

[39] Note that averages for vineyard have no statistical meaning, since we could not prove that soil organic carbon stocks are stationary.

3.4 Complementing the assessment – the case of irrigated pastures

All studies conducted in this thesis refer to rainfed pastures[40]. Even though those are the majority, as shown in Table 1 for the case study of Project Extensity, irrigated pastures gain a particular importance as an alternative to crop production. In a parallel study (Valada *et al.*, 2008, 2010) done during this thesis, the life cycles of two alternative land uses were studied:

1. Irrigated land is used for maize production. Maize is then used as a raw material for the production of ethanol. DDG, a dry distilled grain, is a by-product that can be used to feed cattle as a substitute for soybeans in concentrated feeds. Because an area is occupied with maize, it cannot be grazed by cattle, which remain in stables. Bioethanol is then used in cars' motor combustion. We consider that the CO_2 released was previously sequestered by maize.
2. Sown Irrigated biodiverse permanent Pastures, or SIP for simplification, are installed. SIP are then grazed by cattle. There is direct substitution of bioethanol and gasoline. Therefore, in this scenario, gasoline remains in use due to the non production of maize. Gasoline is used in car combustion, releasing as major pollutants CO_2, CO, NO_x and CH_4.

Since SIP benefit from having water available all year, they guarantee an increased and regular production throughout the year. Therefore, they are used for steer feed. This means that they bring out all the benefits from transferring steers from intensive to extensive production, as we referred in Section 3.2.2.9.

Those scenarios were evaluated in an LCA conducted with software SimaPro 6.0. From all the impact categories that were analysed, we present results here for two: greenhouse gas emissions and energy resources.

The functional unit chosen for the first scenario was 1 t of ethanol, which requires an area of 0.27 ha of maize production. The directly comparable functional unit for the second scenario is the quantity of gasoline that directly substitutes 1 t of bioethanol. That quantity is 0.72 t of gasoline. The same area of 0.27 ha is considered in the second scenario to be used for SIP. Both conventional tillage and no-tillage were considered for maize production. For no-tillage maize and for SIP, carbon sequestration in soils was considered.

Results for greenhouse gas (GHG) emissions are shown in Figure 22 (Valada *et al.*, 2008). If carbon sequestration is not included, the bioethanol production scenario is responsible for only slightly less GHG emissions than the gasoline scenario. However, bioethanol fuels benefit from a tax reduction. If that tax reduction was only motivated by the principle that bioethanol is responsible for low CO_{2e} emissions, at the very least our results show that it would be a very expensive policy. Each ton of CO_{2e} gained from the substitution of gasoline would cost about € 100 to the Portuguese state. Furthermore, if carbon sequestration is included, the SIP scenario gains advantage. Therefore, we concluded that SIP is a better land use for GHG emissions.

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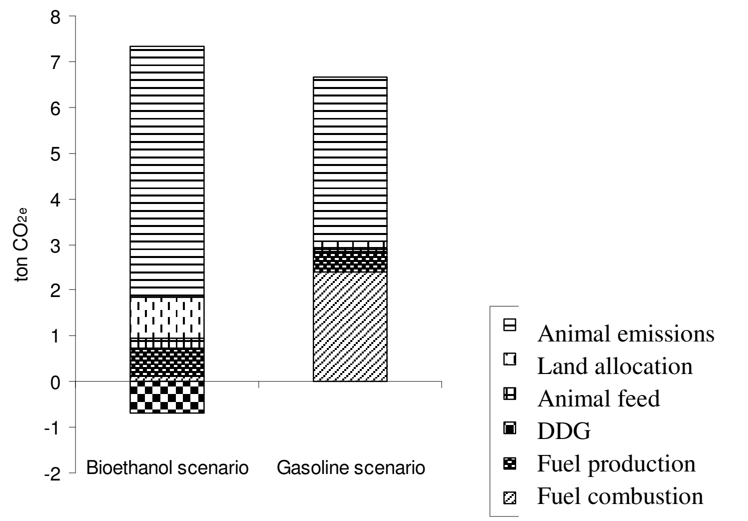
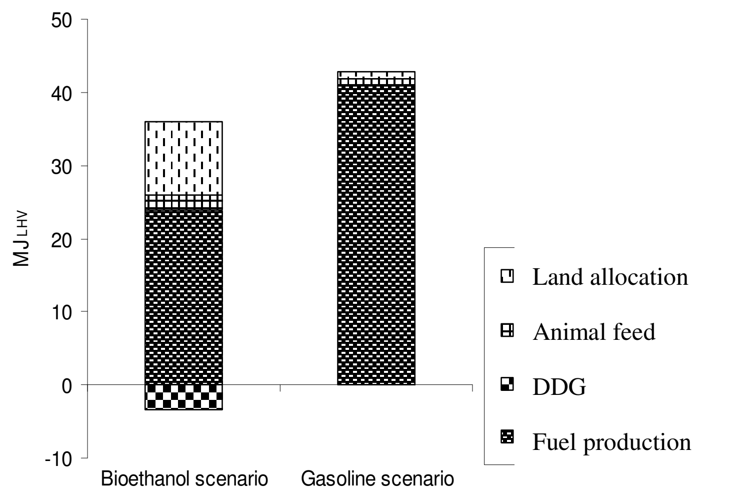
[40] For details on this study, see Valada *et al.* (2008).

Figure 22 – Results for GHG emissions of the bioethanol scenario (maize production) and the gasoline scenario (sown irrigated pastures).



Regarding energy resources (Figure 23), more energy is always necessary in the life cycle of gasoline production than in the bioethanol scenario. Our results (Valada *et al.*, 2008) show that, as an energy policy, it is favourable to produce maize for bioethanol.

Figure 23 – Results for energy resources of the bioethanol scenario (maize production) and the gasoline scenario (sown irrigated pastures).



These results complement the ones shown in this thesis for rainfed pastures. We compared rainfed pastures with the alternative land use, which are natural pastures. Irrigated pastures have to be compared with their alternative land use, which at the time of the study was maize for bioethanol production. And when we do so, we see that, at least in some impact categories, and considering both direct and indirect impacts[41] that result from the land use choice, SIP are also a positive option.

This case of SIP raises the question of whether the study of SBPRL should remain only concerning direct effects, or whether the whole life cycle would yield different conclusions regarding their positive environmental effects. In the next section, we perform the life cycle assessment of environmental impacts of SBPPRL and NG.

3.5 Life cycle assessment of pastures

One method particularly suited for the environmental assessment of indirect impacts is Life Cycle Assessment (LCA). The evaluation framework most commonly applied in LCA involves the steps shown in Table 40 (Tukker, 2000).

Table 40 – Theoretical approach of Life Cycle Assessment.

Step	Theorical approach
Goal and scope definition	Definition of: • the purpose of the study; • the functional unit.
Inventory	A system is defined that includes all relevant process chains of the product function in question: • manufacture; • use; • waste management. For each process in this chain, the relevant environmental interventions are inventoried in relation to the process' contribution to the central product function. Interventions caused by each process that is part of the system concerned are added by intervention type. The final result is a list of all environmental interventions associated with the product's function. The list is known as the inventory table.
Impact assessment (Classification, characterization, and valuation)	Aggregate the information obtained in the inventory. First, a classification of impact categories is chosen, usually reflecting a common mechanism of environmental threat (e.g., global warming, acidification, and ozone depletion). In the characterization step, the environmental interventions listed in the inventory table are translated into scores on each impact category. The result of this operation is called the impact profile of a functional unit of a product. In principle, the impact categories can be aggregated, by means of weighting, to give a single score for the environmental impact of a product.

Source: Tukker (2000).

The LCA done in this thesis recurred to software commonly used in such studies, namely SimaPro 6.0, which was developed by the National Reuse of Waste Research Programme and Pré Consultants of the Netherlands. The use of software is a precious

[41] Note that some indirect impacts come up from the consideration of causal effects related to the choice only by economic reasons – for example, emissions from livestock in intensive production in the maize scenario. In this sense, the analysis conducted was similar to the one we did in this thesis. It also included the same elements of sustainability assessment explained in the first Chapter.

time saver, since the inventory and impact assessment methods are already included therein. The main advantages of using LCA software such as SimaPro are the speed of assessment and the fact that its data base is very wide-ranging, since all existing inputs for any given activity are considered.

However, many processes are country-specific, and the fact that it uses a foreign database is a strong limitation. Its application must be carefully planned, since SimaPro does not possess a thorough database of impacts, including specific impacts for Portugal. It is always necessary to complement the inventory with country-specific data. Therefore, we decided to use SimaPro as the basis, but we incorporated national information whenever it was available or the impact resulting from the process was significant. Results were calculated iteratively – after a first run, most significant impacts were characterized again using country-specific data.

To perform an assessment of the pastures' life cycle, we divided the work in three stages (Teixeira *et al.*, 2008):

1. LCA of feed ingredients;
2. LCA of commercial feeds;
3. LCA of NG and SBPPRL.

In each level, the LCA done in the level before is included is crucial. The impacts of feed ingredients must be determined to obtain the impacts of whole feeds, and animal feed is required when studying the impacts of the whole pasture system. Each part will be explained in detail next.

3.5.1 SimaPro and the Ecoindicators

SimaPro works according to ISO 14000 environmental management standards. It consists of a data base of inputs and outputs from several processes and production of materials. Therefore, the assessment of environmental impacts consists in the sum of impacts from each step of its life cycle (inventory stage). Impacts are then added by environmental themes. The total impact in each theme is then normalized and aggregated into a single impact indicator, usually using one of two methods: "Ecoindicator 95" (EI95) and "Ecoindicator 99" (EI99) (impact assessment stage). Both methods aggregate impacts into a subjective and abstract unit called "Ecoindicator Point", or Pt. Even though conclusions drawn are often similar (Luo 2001), it is important to use both, as they present different themes and a different conception.

EI95 classifies, characterizes and normalizes the environmental impacts based on their contribution to several themes (Luo, 2001). The environmental aspects related to a given product are first aggregated into a number of effects caused, and those are then characterized according to the degree of damage inflicted on the environment; finally, these results are normalized into a single score, based on subjective evaluation (Goedkoop, 1998). The environmental impact themes in EI95 are: emissions of greenhouse gases, heavy metals and carcinogens; substances causing ozone layer destruction, acidification, eutrophication, winter smog and summer smog; and consumption of energy resources[42].

EI99 is an update and extension of EI95, which emphasises its damage-oriented methodology by considering three areas of environmental damage: human health (measured in DALY – Disability Adjusted Life Years), ecosystem quality (expressed as PAF – Potentially Affected Fraction and PDF - Potentially Disappeared Fraction) and resource depletion (expressed as $MJ.kg^{-1}$) (Luo, 2001; Goedkoop and Spriensma, 2000).

3.5.2 LCA of feed ingredients

The most basic unit that comes into animal feed is the ingredient of a commercial feed. First, we determined which are the main ingredients in commercial feeds. To do so, we spoke to farmers from Project Extensity and searched INE's database and the Portuguese Association of Producers of Commercial Feeds for Animals' year book (IACA, 2004). At the time of the study, we came up with the following list:

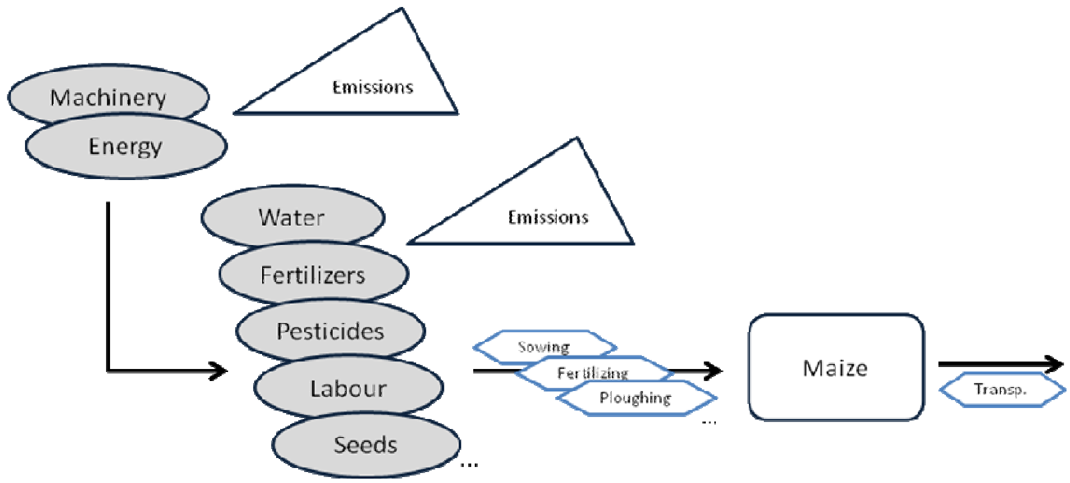
- Forages – wheat straw, silage maize;
- Cereal crops – grain maize, grain wheat, barley;
- Oilseeds – soybeans, sunflower;
- Minor by-products – DDG, corn gluten feed, palm kernel cake.

One LCA was done for each of these ingredients in each location where national consumption is mostly produced. One thorough study (Teixeira *et al.*, 2007) was done to serve as basis for the others. This study was done to compare different locations where maize could be produced, as well as techniques to optimize its environmental impact (such as no-tillage). The study is presented in full in Appendix III – Environmental analysis of maize production. In the context of this thesis, the most important part is to show how results were obtained.

First, we defined the functional unit, which is 1 t of maize, and defined the borders of the life cycle, as shown in Figure 24. We chose the most productive zones and characterized the production systems using the technical coefficients from crop fact sheets in GPP (2001). Then, every product entry (fertilizers, chemicals, …) and every action (sowing, ploughing, …) considered in the fact sheets were simulated using SimaPro 6.0. One international provenience was included, namely Argentina (the main maize selling country to Portugal at the time).

[42] Two more themes are available, namely solid waste and pesticides, but there is rarely enough information to calculate them.

Figure 24 – LCA system studied for maize production.



As mentioned before, the work was done iteratively, and therefore in each run we determined the steps in the life cycle responsible for the most impacts, and redone them using more accurate inventory data. For example, our results show that fertilization is the operation responsible for most of the environmental impact of maize production in all locations (around 70%). Therefore, we had to correct air and water emissions from fertilization operations, using data from van der Werf *et al.* (2005). One particularly crucial parameter was nitrogen leaching. In this case, and not being able to obtain any Portuguese studies on the issue, we turned to studies carried out in relatively similar conditions in Spain.

Engström *et al.* (2007) indicate that the most important environmental themes for Swedish agriculture are eutrophication, global warming and resource use. Our analysis confirms that these themes are important, but indicates some others of interest, like acidification and heavy metals.

The importance of heavy metals is striking. But it may be understood if we follow this thinking line. SimaPro allocates the impact of building the machinery used to the production in which it intervenes. Agriculture is an overcapitalized industry. Unlike other types of machinery (industrial, private transportation vehicles), agricultural machinery is used for a relatively small time frame, and only in a very specific time of year. Therefore, costs and inputs of machinery building and use must always be considered, since its impact is comparable to that of maintenance and fuel consumption. Furthermore, in Portugal, recycling or reuse is not necessarily the final destination of materials, and emissions may be aggravated by lack of adequate final destination. For example, in the case of irrigation, machinery needed stands for 46% of its heavy metal emissions, while electricity consumption stands for 38%.

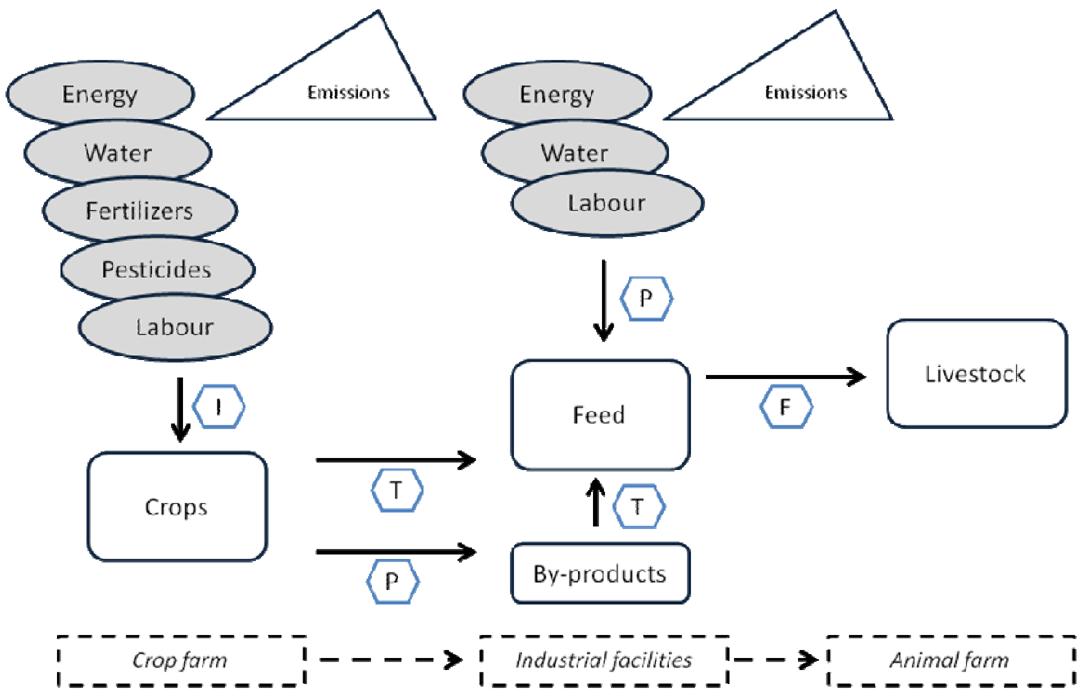
The impact of heavy metals may also be explained by fertilizer use. Fertilizers are currently the main sources of cadmium emissions to the soil, which is an important problem in The Netherlands, where the method was developed. The average European value is 3.8-6.8 $g.ha^{-1}$ in crop land, whereas in The Netherlands values are as high as 7.5-8.5 $g.ha^{-1}$ (Ferrão, 1998). For example, in BI (where the impact on heavy metals is the highest), we found that over 35% of the impact comes from fertilizing. Irrigation also has a very significant part (over 25%).

After calibrating the method for maize produced in many regions, we moved on to all the other ingredients. The last step was to aggregate them all in typical commercial feeds. That's what we did next.

3.5.3 LCA of commercial feeds

For commercial feeds, another specific LCA was conceptualized, as shown in Figure 25. There are four major steps in this life cycle, namely ingredient production (I), transportation of ingredients to an industrial facility (T), feed processing at the facility (P), and transportation of processed feed to the animal farm (F).

Figure 25 – Commercial feed's life cycle scheme.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Therefore, the impact of each feed is:

$$I_{feed} = \sum_{i=1}^{n} I_i + \sum_{i=1}^{n} T_i + P_{feed} + F_{feed}, \tag{3.23}$$

where each capital letter represents the impact of each stage of the life cycle, and i is each ingredient. Four different functional units were tested in this assessment, which were 1 kilogram of:

- feed (total mass);

- crude protein in the feed;
- digestible energy in the feed;
- crude fiber in the feed.

Three feeds were analysed:

1. A grain maize-based feed (Feed 1), which is an average typical formulation for intensive production, since it contains no complement with green forages (only wheat straw);
2. A silage maize-based feed (Feed 2), which is mostly based on silage maize, which is a green forage, mixed at the farm – it is the most "extensive" feed;
3. The average national feed, which is the average of all feed ingredients used in Portugal, and was obtained from the fodder balance (IACA, 2004).

Table 41 gives the composition of each feed. The first type of feed is a conventional maize grain-based feed. It is given to the animal in a quantity of 2.5% of its live weight per day, plus a constant amount of 1.5 kg of wheat straw (Alfredo Sendim; personal communication). The second type of feed is maize silage-based. The animals are fed with 3% of their live weight of this composition per day (Alfredo Sendim; personal communication). Since many feeds are used for each type of animal and age, it is impossible to use a given feed composition as representative (Fernando Anjos; personal communication). The average national feed for finishing calves was obtained from the fodder balance, and so there is a large uncertainty in its composition. This feed should only be considered as a benchmark for the other two, serving as a control.

The animals are fed from the age of 6 to 8 months (180 to 200 kg live weight) to the age of 12 to 14 months[43] (360 to 400 kg live weight), and considering the average value in each of those, it may be shown that the first type of feed, which provides a fixed quantity of straw and 2.5% of the animal's live weight, is given to the animal in larger amounts in the beginning, and smaller in the end, than the second type of feed, which varies the quantity of straw within the base feed (Alfredo Sendim; personal communication).

[43] Note that this is true for organic production. By choosing this specific case, we are using a adopting approach. There are many cases of intensive production in which livestock is not slayed until 24 months of age (Alfredo Sendim; personal communication).

Table 41 – Composition of the feeds studied.

	% (kg ingredient.100 kg^{-1} feed)		
Ingredient	**Feed 1**	**Feed 2**	**Average feed**
Maize (silage)		58.6	
Maize (grain)	20.0	12.5	13.7
Corn Gluten Feed	20.0	6.6	27.0
Wheat (grain)	19.0	6.6	4.9
Barley	10.0		2.1
Soy meal (44% protein)	6.0	9.1	16.9
Manioc	5.0		
DPG	5.0		
Palm kernel cake	5.0		
Sunflower	4.8		
Carbonate	2.1		
Fats	0.9		
Bicarbonate	0.8		
Salt	0.6		
Premix	0.5	0.8	
Urea	0.3		
Others			36.3[44]
Wheat (straw)	(1.5 $kg.day^{-1}$)[45]	5.8	

Here, we considered two intervals of 2.4 months: (1) from 7.2 to 9.6 months, when the animals are fed in 60% by commercial feed and 40% from pasture; (2) from 9.6 to 12 months, animals are confined to stables and completely fed with the commercial feed. Feed 1 and the average feed allow the animals to grow more rapidly, at the rate of 1.5 $kg.day^{-1}$, whereas Feed 2 provides a slower growth rate of 1.2 $kg.day^{-1}$. Therefore, animals fed with Feed 2 end the second period with less weight than those fed in the other cases (Table 42) (Alfredo Sendim; personal communication).

Table 42 – Animal weights when fed with each feed.

Age (months)	Weight 1 and average (kg)	Weight 2 (kg)
7.2	190.0	190.0
9.6	298.0	276.4
12	406.0	362.8

Again, we used SimaPro 6.0 and an iterative approach to results, correcting the inventory for the most significant impacts. For by-products, we also considered two impact allocation methods – mass allocation, and economic value allocation.

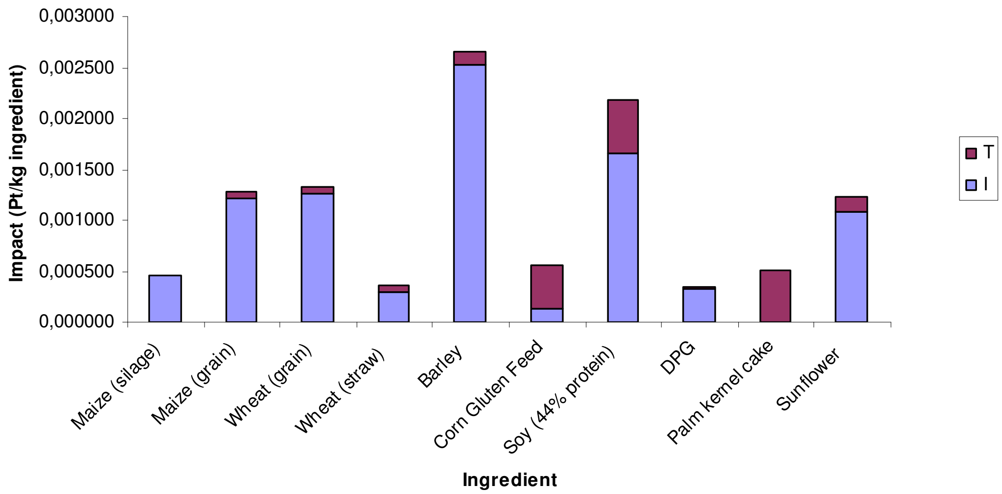
The whole study is presented in Appendix IV – Environmental analysis of concentrated feeds. Returning to individual feed ingredients, results show that by-products have lower environmental impacts. Barley and soybeans are the ones with higher impacts per kilogram, as shown in Figure 26. Barley also has the worst environmental impact per kilogram of crude protein, crude fiber and digestible energy. Soy, on the other hand, is

[44] The "others" are undisclosed cereals and by-products. Due to lack of information, it was not possible to determine what they are.

[45] Straw is given to the animals in a fixed quantity, which does not depend on the quantity of feed also given. It could not be determined whether the average feed contains straw, but since its fibre content is equal to that of Feed 1 (as shown next) it is plausible to assume that it does not (straw is mainly used for fibre).

an efficient provider of protein. Silage maize has a low impact per kilogram and per unit of crude fiber, but is not an efficient source of protein or energy. Wheat straw and by-products are efficient in all functional units.

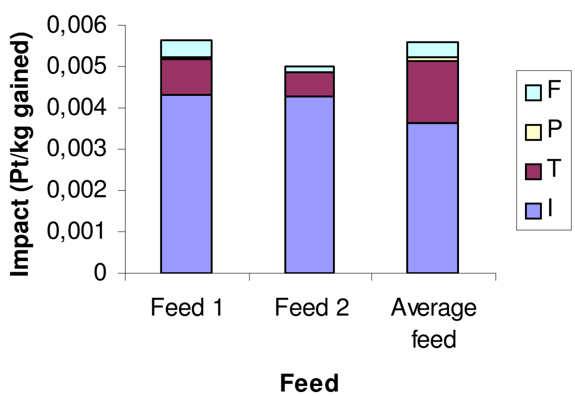
Figure 26 – Unit impact of per kilogram of each of the ingredients (Ecoindicator 95).



I – Impacts of ingredient production; T – impacts of transportation of ingredients.

As for the impact of aggregated feeds, shown in Figure 27, we found that each kilogram of the silage maize-based feed ("extensive" feed) has a lower aggregated environmental impact than the grain maize-based feed ("intensive" feed). Feed 1 has, in fact, about the same impact as the average feed. We can also see from Figure 27 that the production of the ingredient is the stage of the life cycle with higher impacts, but its transportation to the industrial facility cannot be neglected.

Figure 27 – Overall impact of each feed (Pt.kg^{-1} gained), Ecoindicator 95.



I – Impacts of ingredient production; T – impacts of transportation of ingredients; P – impacts of feed processing at the facility; F – transportation of the feed to the animal farm.

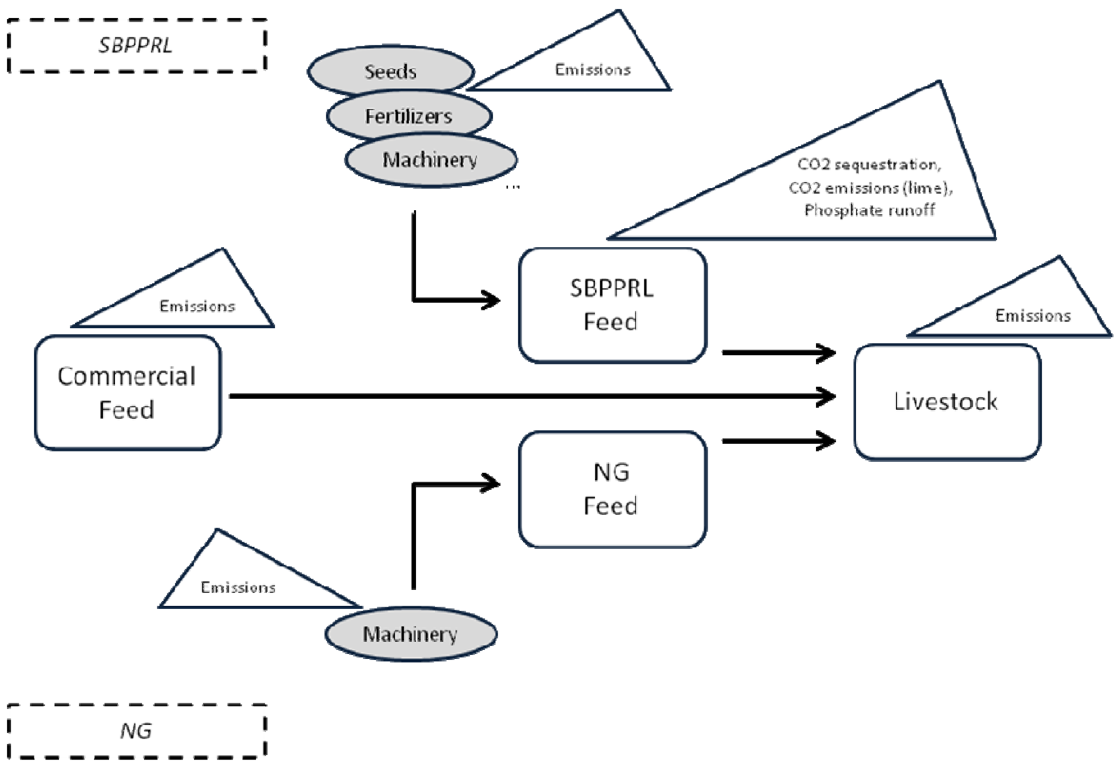
However, each of these feeds do not provide the same amounts of protein, fiber and energy to cattle (although all provide an acceptable minimum of each). Feed 2 is very rich in fiber, while Feed 1 and the average feed are slightly richer in terms of protein and energy. For this reason, all feeds have about the same environmental impact per unit of crude protein and digestible energy, while feed 2 has a lower environmental impact per unit of crude fiber.

Now that we know which feeds are typically used and their impacts, we can turn to the life cycle of pastures as a whole.

3.5.4 LCA of grassland systems

A whole LCA study is required for SBPPRL and NG due to the fact that one is more intensive than the other. SBPPRL require a wider array of inputs than NG: seeds, fertilizers, and machinery. In a way, they are a return to a pre-labor extensive systems situation. Major differences between animal production in sown and natural grasslands are shown in Figure 28.

Figure 28 – Life cycles of animal production in natural (baseline scenario) and sown (proposed scenario) grasslands.



NG – Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; CO_2 – Carbon dioxide.

The impact of 1 t of feed was determined in the previous section. Those feeds, however, are meant for steers. Nevertheless, we may consider that the silage maize-based feed is appropriate for grazing cows. This is the feed with the lower impact from those we studied before, and therefore it is also the most favorable for the NG scenario.

Therefore, we guarantee that we do not bias results towards decreasing the SBPPRL system impacts (where there are less or none commercial feeds).

The impact of 1 ha of SBPPRL and NG can be calculated using software SimaPro 6.0, using the inputs in the life cycle in Figure 28. For SBPPRL, we use the sheets that will be shown in the next Chapter, in Table 59 for installation and Table 60 for maintenance. One tenth of the installation impacts were added to the maintenance impacts, since the expected time frame of SBPPRL is 10 years. The impact of one generic legume seed in SimaPro's database were used for SBPPRL seeds. This is a simplification, since the impact of the production of the seeds is very different from species to species. For NG, only one tillage every five years was considered. Therefore, one fifth of the impact of one tillage operation was calculated to find the impact of NG.

We use two scenarios for SBPPRL: one where 100 kg of phosphate fertilizer (Single Superphosphate) are added yearly for maintenance, and one where 200 kg are added. The 200 kg are the worst-case scenario possible, corresponding to the worst levels of P in soils prior to installation. Typically, less P is needed, and after a while the operation isn't required every year. Therefore, the 100 kg scenario is included for comparison purposes. Even though the same uncertainty in quantities exsists for other inputs and operations, phosphate fertilizer is the main source of impact, as we will show next.

Results are shown in Table 43. Each hectare of SBPPRL has, naturally, higher impacts than each hectare of NG. This is due to the fact that NG only require one tillage operation around every 5 years, while SBPPRL require a wide array of operations and inputs. The only impact category where SBPPRL have a better result than NG is GHG emissions, measured as CO_2e, due to carbon sequestration.

Table 43 – LCA impacts of 1 ha of SBPPRL and NG and 1 t of feed in each impact category.

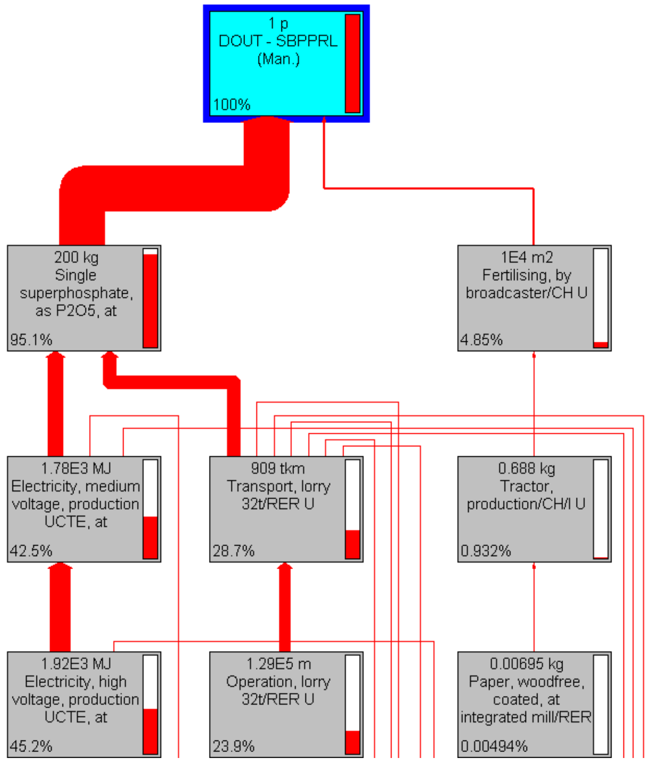
Impact category	[Unit]	$\{I\}^{SBPPRL}$ [Unit].ha^{-1} 100 kg.yr^{-1} phosphote	$\{I\}^{SBPPRL}$ [Unit].ha^{-1} 200 kg.yr^{-1} phosphote	$\{I\}^{NG}$ [Unit].ha^{-1}	$\langle I\rangle^{feed}$ [Unit].t^{-1}
GHG	kg CO2	-3.9×10^{3}	-3.6×10^{3}	1.1×10^{1}	3.5×10^{2}
Ozone layer	kg CFC11	4.7×10^{-5}	7.4×10^{-5}	2.1×10^{-6}	3.5×10^{-5}
Acidification	kg SO2	5.7×10^{0}	9.9×10^{0}	9.2×10^{-2}	4.1×10^{0}
Eutrophication	kg PO4	1.2×10^{0}	1.9×10^{0}	1.6×10^{-2}	1.0×10^{0}
Heavy metals	kg Pb	2.0×10^{-2}	3.4×10^{-2}	4.7×10^{-4}	3.9×10^{-3}
Carcinogens	kg B(a)P	6.6×10^{-5}	9.7×10^{-5}	5.4×10^{-6}	2.8×10^{-5}
Winter smog	kg SPM	4.3×10^{0}	7.7×10^{0}	3.2×10^{-2}	6.8×10^{-1}
Summer smog	kg C2H4	1.5×10^{-1}	2.3×10^{-1}	7.8×10^{-3}	9.5×10^{-2}
Energy resources	MJ LHV	5.5×10^{3}	9.2×10^{3}	7.7×10^{2}	3.1×10^{3}

I – Impact; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; NG – Natural Grasslands; GHG – Greenhouse Gases; LHV – Lower Heating Value; LCA – Life Cycle Assessment.

Almost the totality of the impact of SBPPRL in every category is due to phosphorus fertilizer production, transport and application. While several studies exist on the environmental effects of the application of phosphorus (Blake *et al.*, 2000), our study indicates that it is the life cycle impacts involved in its production that are responsible for the most part of the impacts. This is particularly visible in the acidification and energy resources themes, as shown in Figure 29 for energy resource consumption.

Production of the fertilizer is responsible for 95% of all the energy use for the maintenance of SBPPRL. On many farms, however, fertilizers are applied and farmers continue to grow crops.

Figure 29 – Main sources of energy resorce consumption in the life cycle of SBPPRL, according to SimaPro 6.0.



In most impact categories, each unit of synthetic phosphate has less environmental impacts than, each unit of nitrogen fertilizer (less GHG emissions, for example), except for the themes acidification, eutrophication and winter smog, as shown in Table 44. Therefore, SBPPRL suppress the need for N fertilizers, which is also positive for GHG reduction, but require phosphate, which causes impacts in other themes.

Table 44 – Comparison of LCA impacts between a nitrogen fertilizer and a phosphate fertilizer.

Impact category	[Unit]	Ammonium nitrate [Unit].kg^{-1} N	Urea [Unit].kg^{-1} N	Single superphosphate [Unit].kg^{-1} P_2O_5	Triple superphosphate [Unit].kg^{-1} P_2O_5
GHG	kg CO2	7.8×10^{0}	3.0×10^{0}	2.5×10^{0}	1.9×10^{0}
Ozone layer	kg CFC11	4.1×10^{-7}	4.2×10^{-7}	2.7×10^{-7}	1.9×10^{-7}
Acidification	kg SO2	2.7×10^{-2}	1.3×10^{-2}	4.2×10^{-2}	3.3×10^{-2}
Eutrophication	kg PO4	4.4×10^{-3}	1.9×10^{-3}	6.5×10^{-3}	4.2×10^{-2}
Heavy metals	kg Pb	5.9×10^{-5}	4.0×10^{-5}	1.4×10^{-4}	2.4×10^{-4}
Carcinogens	kg B(a)P	3.4×10^{-7}	3.2×10^{-7}	3.1×10^{-7}	1.9×10^{-7}
Winter smog	kg SPM	6.4×10^{-3}	5.2×10^{-3}	3.4×10^{-2}	2.9×10^{-2}
Summer smog	kg C2H4	5.8×10^{-4}	5.8×10^{-4}	8.2×10^{-4}	4.8×10^{-4}
Energy resources	MJ LHV	5.6×10^{1}	6.4×10^{1}	3.7×10^{1}	2.7×10^{1}

GHG – Greenhouse Gases; LHV – Lower Heating Value; LCA – Life Cycle Assessment.

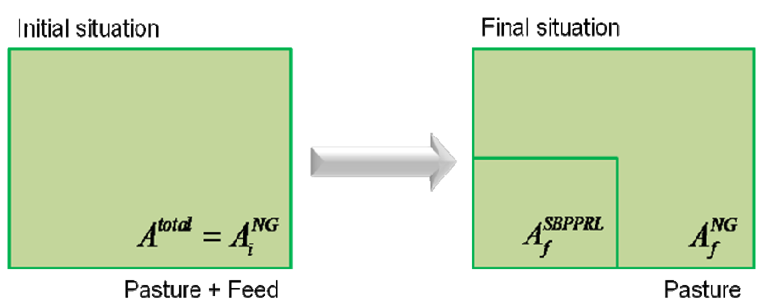
Still, as we have noticed before, the direct comparison between each hectare of SBPPRL and NG is of little interest, since it is not a complete life cycle. The NG system's life cycle is only complete considering feeds. And so we move on to determining the overall impacts of each scenario.

3.5.4.1. <u>How to calculate the LCA impacts with the available data?</u>

However, as may be suggested by Figure 28, it does not suffice to calculate, in one scenario, the impact of 1 ha of SBPPRL, and compare it, in the other scenario, with the impact of 1 ha of NG and corresponding substituted feed. This approach would be wrong because it does not maintain the livestock balance. In such a situation, there is no guarantee that the same number of animals are being fed in the two scenarios. Remember, for example, the case of the field plots used so far, where livestock units in SBPPRL and NG were the same cattle grazes both plots, differing only on the number of days it is kept on each (and therefore the feed is the same).

A more likely approach is to consider two scenarios regarding two moments in time (Figure 30). In the beginning, before installing SBPPRL, all plots in the farm are NG (we hereby denote all quantities regarding the initial simulation with the subscript *i*). Cattle grazes and consumes commercial feeds. Then, farmers starting installing SBPPRL. We assume that they install as many as necessary to suppress the need for concentrated feeds, or the whole area of the farm, whichever is smaller (quantities referring to the final simulation are denoted with the subscript *f*). This assumption is useful for calculations, since the amount of feeds provided by farmers with NG and SBPPRL is unknown.

Figure 30 – Scenario for substitution of feed by increased production in SBPPRL.



A – Area; i – initial; f – final; NG – Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.

In this scenario, several balances may be defined. All calculations are done for the average hectare in the farm. The first one is the area balance, which states that the total area of the farm (A^{total}) is equal to the area of NG in the beginning (A_i^{NG}), and to the sum of the area of NG in the end (A_f^{NG}) plus the area of SBPPRL installed (A_f^{SBPPRL}):

$$A^{total} = A_i^{NG} = A_f^{NG} + A_f^{SBPPRL}. \tag{3.25}$$

The second balance is the livestock balance, which simply states that, since SBPPRL replace cattle needs for commercial feeds, the number of livestock units (LU) in constant (no animals enter or leave the farm):

$$LU^{total} = LU_i = LU_f \,. \tag{3.26}$$

Given the previous condition, the energy requirements in the animal feed in the beginning and in the end are the same, and therefore, there is an energy balance:

$$\begin{aligned} LU\left(\frac{EFU}{LU}\right)^{total} &= A_i^{NG}\left\{EFU\right\}^{NG} + A_i^{NG}\left\{EFU\right\}^{feed} \\ &= A_f^{NG}\left\{EFU\right\}^{NG} + A_f^{SBPPRL}\left\{EFU\right\}^{SBPPRL} \end{aligned} , \tag{3.27}$$

where $\{EFU\}$ are the forage units, measured as digestible energy, per hectare, respectively, of the commercial feed and the feed grazed in pastures by the cattle. Regarding the feed, however, the energy content should be assessed per unit of mass. In order to do so, a conversion is required:

$$\left\{EFU\right\}^{feed} = \frac{\left\{EFU\right\}^{feed}}{\left\{M\right\}^{feed}}\left\{M\right\}^{feed} = \left\langle EFU\right\rangle^{feed}\left\{M\right\}^{feed}, \tag{3.28}$$

where $\{M\}$ is the quantity of feed required yearly per area unit of NG in the initital situation.

The total impact of the initial situation is the sum of the impact of NG and the feed in the area of the farm:

$$\left\{I\right\}_i^{total} = A_i^{NG}\cdot\left(\left\{I\right\}^{NG} + \left\{I\right\}^{feed}\right), \tag{3.29}$$

where $\{I\}$ is the environmental impact per hectare, respectively, of the commercial feed and the feed grazed in pastures by the cattle. As before, the energy content of the feed should be assessed per unit of mass. In order to do so, another conversion is required:

$$\left\{I\right\}^{feed} = \frac{\left\{I\right\}^{feed}}{\left\{M\right\}^{feed}}\left\{M\right\}^{feed} = \left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}. \tag{3.30}$$

The impact of the final situation is thus:

$$\left\{I\right\}_f^{total} = A_f^{NG}\left\{I\right\}^{NG} + A_f^{SBPPRL}\left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}. \tag{3.31}$$

Therefore, the difference in impacts between the two situations is:

$$\left\{I\right\}_f^{total} - \left\{I\right\}_i^{total} = A_f^{NG}\left\{I\right\}^{NG} + A_f^{SBPPRL}\left\{I\right\}^{SBPPRL} - A_i^{NG}\cdot\left(\left\{I\right\}^{NG} + \left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}\right). \tag{3.32}$$

Rearranging this expression, and using the area balance, we find that:

$$\left\{I\right\}_f^{total} - \left\{I\right\}_i^{total} = A_f^{SBPPRL}\left(\left\{I\right\}^{SBPPRL} - \left\{I\right\}^{NG}\right) - A^{total}\left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}. \tag{3.33}$$

Since we want to find the life cycle environmental impact of the installation of 1 ha of SBPPRL, this Equation must be divided by the area of SBPPRL:

$$\frac{\left\{I\right\}_f^{total} - \left\{I\right\}_i^{total}}{A_f^{SBPPRL}} = \left\{I\right\}^{SBPPRL} - \left\{I\right\}^{NG} - \frac{A^{total}}{A_f^{SBPPRL}}\left\langle I\right\rangle^{feed}\left\{M\right\}^{feed}. \tag{3.34}$$

This Equation (3.34) shows that the difference in impact due to the installation of 1 ha of SBPPRL is equal to the impact of SBPPRL per hectare minus the impact of NG per hectare, and minus the impact of the commercial feed multiplied by the inverse of the fraction of the farm where, in the final situation, SBPPRL have been sown.

But in Equation (3.34) there are two missing terms. We do not have data for $\{M\}^{feed}$ (we do not know how much feed is given to cattle per hectare of NG, since there is a large variability and uncertainty in this term). We also do not know the fraction of the total area of the farm which is sown with SBPPRL (the area required to compensate the use of feeds). We define this fraction as x:

$$x \equiv \frac{A_f^{SBPPRL}}{A^{total}}. \tag{3.35}$$

Therefore, Equation (3.34) must be modified, in order to suppress this lack of data. We return to the energy balance in Equation (3.27), and use the area balance to modify it

$$\begin{aligned} &A^{total}\{EFU\}^{NG} + A^{total}\langle EFU\rangle^{feed}\{M\}^{feed} = \\ &= \left(A^{total} - A_f^{SBPPRL}\right)\{EFU\}^{NG} + A_f^{SBPPRL}\{EFU\}^{SBPPRL}. \end{aligned} \tag{3.36}$$

Dividing Equation (3.36) by the total area and using the definition of x, we find that:

$$\{EFU\}^{NG} + \langle EFU\rangle^{feed}\{M\}^{feed} = (1-x)\{EFU\}^{NG} + x\{EFU\}^{SBPPRL}. \tag{3.37}$$

Equation (3.37) may now be divided by the forage units of NG:

$$1 + \frac{\langle EFU\rangle^{feed}}{\{EFU\}^{NG}}\{M\}^{feed} = (1-x) + x\frac{\{EFU\}^{SBPPRL}}{\{EFU\}^{NG}}. \tag{3.38}$$

At this time, we may introduce a new variable, ε, which is the coefficient of forage production between SBPPRL and NG:

$$\varepsilon = \frac{\{EFU\}^{SBPPRL}}{\{EFU\}^{NG}}. \tag{3.39}$$

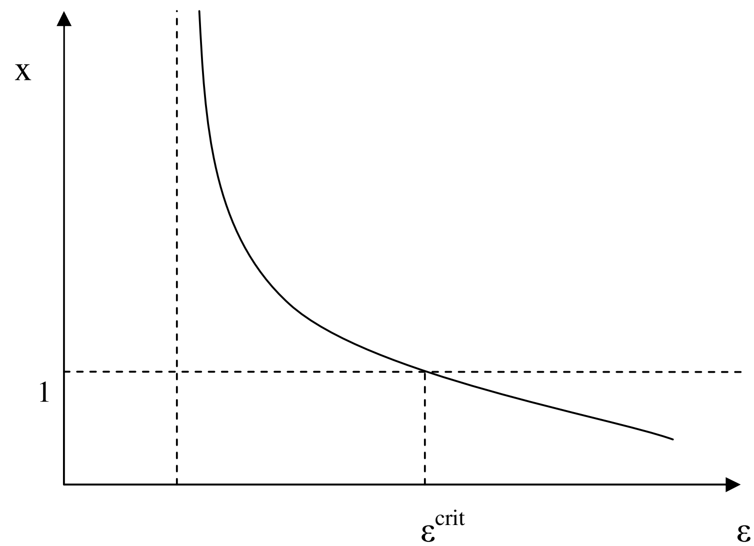
This coefficient is useful because, even though we do not know the energy content of both pastures, we can know their ratio, which is equal to the ratio between stocking rates in SBPPRL and NG plots. Field data was obtained in a situation where cattle grazed more days SBPPRL plots than NG plots, but was given the same quantity of commercial feeds. Therefore, we may assume that the ratio in stocking rates, both measured in grazing days, corresponds to the ratio in forage content of both types of pasture.

Using this coefficient in Equation (3.38), and solving for x:

$$x = \frac{1}{\varepsilon - 1}\frac{\langle EFU\rangle^{feed}}{\{EFU\}^{NG}}\{M\}^{feed}. \tag{3.40}$$

The graphical representation of Equation (3.40) is shown in Figure 31.

Figure 31 – Effect of the nutritional quality of SBPPRL (*ε*) on the fraction of the farm sown with SBPPRL (*x*).



There is an ε^{crit}, which is the value for which all area is converted to SBPPRL, and therefore is defined by x = 1, or, recurring to Equation (3.40):

$$\varepsilon^{crit} = 1 + \frac{\langle EFU \rangle^{feed}}{\{EFU\}^{NG}} \{M\}^{feed}. \tag{3.41}$$

Therefore, according to Equation (3.41), the critical value increases with the fraction of the animal feed that is assured by feeds in the initial case. For $\varepsilon < \varepsilon^{crit}$, there is no valid solution for *x*: the productivity of SBPPRL is not sufficient to completely eliminate the use of feed, within the available area. This means that, in this situation, all available area would be converted to SBPPRL, and feeds would still be required.

Returning to our goal, which was to calculate the difference in impacts between the final and the initial situation per unit of SBPPRL area, we may use the inverse of Equation (3.40) and substitute it in Equation (3.34), obtaining

$$\frac{\{I\}_f^{total} - \{I\}_i^{total}}{A_f^{SBPPRL}} = I^{SBPPRL} - I^{NG} - (\varepsilon - 1)\frac{\{EFU\}^{NG}}{\langle EFU \rangle^{feed}} \langle I \rangle^{feed}. \tag{3.42}$$

Using the definition of ε again, we find the final expression:

$$\frac{\{I\}_f^{total} - \{I\}_i^{total}}{A_f^{SBPPRL}} = \{I\}^{SBPPRL} - \{I\}^{NG} - \left(\frac{\varepsilon - 1}{\varepsilon}\right)\frac{\{EFU\}^{SBPPRL}}{\langle EFU \rangle^{feed}} \langle I \rangle^{feed}. \tag{3.43}$$

We have estimates available for all of the variables in Equation (3.43). We begin with LCA results of the environmental impacts.

3.5.4.2. <u>Determining the impacts of each scenario</u>

We have ε = 2 since, as we have shown before, stocking rate in SBPPRL is approximately double that of NG. The digestible energy content of 1 t of feed, and of 1 ha of SBPPRL, can be obtained in energy equivalency tables for feedstuff (Stanton, 2004). Results for 1 t of the feed we used are shown in Table 45.

Table 45 – Digestible energy content of each ton of feed.

Ingredient	Quantity ($t.t^{-1}$ feed)	DM (%)	DE ($MJ.kg^{-1}$ DM)	DE (MJ)
Maize (silage)	0.586	36	12.7	2 687
Maize (grain)	0.125	89	16.4	1 828
Corn Gluten Feed	0.066	90	15.1	899
Wheat (grain)	0.066	89	16.4	965
Soy meal (44% protein)	0.091	90	16.1	1 315
Premix	0.008	-	-	-
Wheat (straw)	0.058	88	8.1	415
Total	1	-	-	8 175

DM – Dry Matter; DE – Digestible Energy.

As for SBPPRL, since there is no single composition of the pasture, we must arbitrate significant values for *EFU*. We use clover as representative of SBPPRL energy content. The digestible energy in clover varieties may be retrieved from energy equaivalency tables for feedstuff. Table 46 shows the average, minimum and maximum values (Stanton, 2004). The interval is due to the fact that the energy content depends on the state of maturation. Green (early stage) clovers are typically more energetic (corresponding to the maximum values). To obtain the total energy in one hectare, we must multiply the values in Table 46 by the dry matter production of SBPPRL which, according to Table 21, is 7 000 kg.

Table 46 – Digestible energy content of each clover species.

Clover variety	Digestible energy ($MJ.kg^{-1}$ dry matter)		
	Average	Minimum	Maximum
Trifolium alexandrinum	19.5	15.3	22.7
Trifolium meneghianum	20.0	20.0	20.0
Trifolium pratense	21.8	21.8	21.8
Trifolium repens	20.7	20.7	20.7
Trifolium squarrosum	21.1	20.4	21.8
Trifolium subterraneum	25.6	25.6	25.6
All varieties	**20.0**	**15.3**	**25.6**

Results for the difference in impacts are shown in Table 47 for all impact categories studied. A negative value means that, for every hectare of SBPPRL sown, we obtain less impacts than in the prior situation with NG. This is the case for all themes when 100 kg of phosphate fertilizer are used. When 200 kg of phosphate fertilizer are used, the only theme where the scenario with SBPPRL has more impact is winter smog. This is one of the themes where phosphate fertilizers have more impacts than nitrogen fertilizers, and therefore there is a significant loss when substituting feeds for SBPPRL. Results are qualitatively the same if we use the extreme values of digestible energy in Table 46 instead of the average; the only difference is that, when using the maximum value, the scenario with SBPPRL has less impacts in all themes regardless of the quantity of phosphate fertilizer.

Table 47 – Difference in impacts between scenarios per area of SBPPRL sown.

Impact category	[Unit]	ε	$\{EFU\}^{SBPPRL}$ MJ.ha^{-1}	$\langle EFU\rangle^{feed}$ MJ.ton^{-1}	$\frac{\{I\}_f^{total}-\{I\}_i^{total}}{A_f^{SBPPRL}}$ 100 kg.yr^{-1} phosphote	200 kg.yr^{-1} phosphote
GHG	kg CO2	2	139.8 x 10^3	8.2 x 10^3	-6.9 x 10^3	-6.7 x 10^3
Ozone layer	kg CFC11				-2.6 x 10^{-4}	-2.3 x 10^{-4}
Acidification	kg SO2				-2.9 x 10^1	-2.5 x 10^1
Eutrophication	kg PO4				-7.5 x 10^0	-6.9 x 10^0
Heavy metals	kg Pb				-1.4 x 10^{-2}	-1.4 x 10^{-4}
Carcinogens	kg B(a)P				-1.8 x 10^{-4}	-1.5 x 10^{-4}
Winter smog	kg SPM				-1.6 x 10^0	1.8 x 10^0
Summer smog	kg C2H4				-6.7 x 10^{-1}	-5.9 x 10^{-1}
Energy resources	MJ LHV				-2.1 x 10^4	-1.8 x 10^4

I – Impact; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; NG – Natural Grasslands; GHG – Greenhouse Gases; LHV – Lower Heating Value; LCA – Life Cycle Assessment.

3.5.4.3. Consistency analysis – what fraction of a farm is sown at most?

Another interesting analysis is to determine what fraction of farm area is sown by farmers or, in other words, how much area of SBPPRL do farmers have to install in order to eliminate concentrated feeds. In order to do so, we return to Equation (3.34) and to the fact that two variables were unknown: $\{M\}^{feed}$ and *x*. From Equation (3.40), we can see that there is a direct proportionality between both, and that neither one can be determined independently. In other words, there are several combinations of the two variables that yield the same environmental impacts that we determined in Table 47. The relation between the variables may be obtained from Equation (3.34) as:

$$\frac{\{M\}^{feed}}{x}=\frac{\{I\}^{SBPPRL}-\{I\}^{NG}-\dfrac{\{I\}_f^{total}-\{I\}_i^{total}}{A_f^{SBPPRL}}}{\langle I\rangle^{feed}}.$$

From Equation (3.40), this is the equal to

$$\frac{\{M\}^{feed}}{x}=\left(\frac{\varepsilon-1}{\varepsilon}\right)\frac{\{EFU\}^{SBPPRL}}{\langle EFU\rangle^{feed}}.$$

Using the same values already shown in Table 47, we find that the ratio between the two variables is equal to 8.56 kg.ha^{-1} SBPPRL. We can thus build a table that presents, for each x, which would be the feed initialy provided to animals, in the absence of SBPPRL. Then, we can multiply that quantity by the $\langle EFU\rangle^{feed}$ to calculate the energy provided by the feed in each case. We can add that value to the $\{EFU\}^{NG}$ (which we know is half of the value for SBPPRL, since ε = 2), and obtain the total energy provided to animals in the initial situation. Then, to check the consistency of the results, we can calculate the energy provided in the final situation, which is equal to the fraction of SBPPRL times $\{EFU\}^{SBPPRL}$ plus the complementary fraction, which are NG, times $\{EFU\}^{NG}$. The results for each line of the table should be the same. The consistency of results is shown in Table 48.

Table 48 – Initial and final digestible energy for each scenario of SBPPRL area fraction.

x	$\{M\}^{feed}$ (t.ha^{-1})	$\{EFU\}^{feed}$ (GJ.ha^{-1})	$\{EFU\}^{NG}$ (GJ.ha^{-1})	$\{EFU\}_i^{total}$ (GJ.ha^{-1})	$\{EFU\}_f^{total}$ (GJ.ha^{-1})
0.0	0.00	0	89.5	89.5	89.5
0.1	1.09	8.9	89.5	98.4	98.4
0.2	2.19	17.9	89.5	107.4	107.4
0.3	3.28	26.8	89.5	116.3	116.3
0.4	4.38	35.8	89.5	125.3	125.3
0.5	5.47	44.7	89.5	134.2	134.2
0.6	6.57	53.7	89.5	143.2	143.2
0.7	7.66	62.6	89.5	152.1	152.1
0.8	8.76	71.6	89.5	161.1	161.1
0.9	9.85	80.5	89.5	170.0	170.0
1.0	10.95	89.5	89.5	179.0	179.0

x – fraction of SBPPRL area in farm; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; NG – Natural Grasslands; EFU – Energy Forage Units; i – initial situation; f – final situation.

The higher the value of *x*, the higher the quantity of feed that is given to animals in each hectare of NG in the initial situation. Since we considered a constant energy provision by each type of pasture, this also means that more energy is required by livestock. Therefore, if we estimate how much feed is needed in an average case, we can also know *x* in an average case.

If we assume a stocking rate of 0.5 LU.ha^{-1}; that a cow weights around 500 kg; that a cow eats between 0.5 and 1.5% of its live weight per day; then we can assume that feed consumption in the initial situation is 0.5-1.4 t feed.ha^{-1}. According to Table 48, this implies *x* between 0 and 20 %. In our case, the critical value is never reached, and we have reason to believe that NG and SBPPRL will always co-exist in farms. They are complementary, rather than purely competitive, options.

If we now return to the first Chapter, and recall that in Table 1 we showed that 15.6% of the area of pastures in Project Extensity were SBPPRL[46], then we obtain the justification we needed for our prior approach when we clamed that farmers install SBPPRL up until the point when no more commercial feeds are required.

Two environmental effects which are not accounted by LCA are soil loss and biodiversity. We now turn to those.

[46] We did not use this figure in the beginning because, even though it may be used as an indication for consistency check, it is unreliable as a parameter. The overall Extensity area is not equaivalent to a single farmer, and even so there was no indication that all farmers had no intention of sowing more pastures.

3.6 Soil protection and decreased erosion

NG require some form of mechanical operation to control shrub invasion. SBPPRL, on the other hand, do not, since animals control shrubs by themselves. The reduction of these actions, which imply some form of soil tillage, corresponds to less destruction of soil agglomerates. This effect is well documented for croplands where tillage was substituted by no-tillage (Basch *et al.*, 2001). The passage of agricultural machinery creates a crust at the surface of the field which decreases the infiltration of water, increasing water runoff and carriage of soil particles. Another main determinant of soil erosion is SOM. Soils with high SOM concentration are more stable, and less susceptible to erosion.

Soil loss may be estimated using Wischmeier and Smith's (1978) Universal Soil Loss Equation (USLE). This is the most simple and well-known model to this end. The basic formulation of the model is

$$A = R \cdot K \cdot LS \cdot C \cdot P\,, \qquad (3.19)$$

where

A – average soil loss or specific erosion ($t.ha^{-1}.yr^{-1}$);

R – rainfall erosivity factor ($MJ.mm.ha^{-1}.h^{-1}.yr^{-1}$);

For Portugal, and according to Tomás (1993), R may be estimated according to an empirical equation obtained as the average for three stations in the Algarve region:

$$R = -685{,}3 + 3{,}406P\,, \qquad (3.20)$$

where P is total annual precipitation (mm).

K – soil erodibility factor ($t.MJ^{-1}.mm^{-1}$);

LS – slope length-gradient factor, considering slope length and steepness simultaneously (dimensionless)

$$LS = \left(\frac{x}{22.13}\right)^m .\left(0.065 + 0.045s + 0.0065s^2\right) \qquad (3.21)$$

x – average slope length (m)

m – factor determined by the interaction between slope length and steepness, as well as soil proprieties, vegetation type and agricultural practices (dimensionless, 0.2 < m < 0.6)

s – slope steepness (%)

C – crop/vegetation and management factor (dimensionless)

P – support practice factor (dimensionless)

From all these factors, the ones where grassland type has any significant influence are K, C and P. We will analyse next the effect of switching land use from NG to SBPPRL in each of the three coefficients.

3.6.1 Effect on the parameter K

According to Pimenta (1998b), the K factor for permanent pastures is 0.02, which is the same as for areas with spontaneous herbaceous or shrub vegetation. There is no value for SBPPRL. Therefore, we determined the difference between both from the difference in SOM concentration. Chapter 1 of the present thesis is dedicated to comparing SOM dynamics in different grassland types. For now, it suffices to say, for motives we will explain then, that SOM increases in SBPPRL by about 0.20 percent points per year more than NG. We will consider that, in 10 years, a NG is kept for 2 cycles of 5 years with no mobilization, keeping its SOM level in 1% throughout the period. In the same 10 years, SBPPRL are never tilled, and increase their SOM level from 1 to 3%.

To determine K, we resort to Pimenta (1998a), and the expression

$$K = \frac{2.1 \times M^{1,14} \times 10^{-4} \times (12 - SOM) + 3.25 \times (\alpha - 2) + 2.5 \times (\beta - 3)}{759.3}, \tag{3.22}$$

where

M – % of loam and thin sand multiplied by 100% less the percentage of clay (four major texture classes were considered, as shown in Table 49);

Table 49 – Percentage of each soil constituents for major texture classes.

Textura	% clay	% lome	% sand
Clay	60	10	30
Silt	20	25	55
Loam	5	65	30
Sand	5	2,5	92,5

α - factor for soil structure class (1 to 4, as shown in Table 50);

Table 50 – Parameter α to determinate soil erodibility.

Structure	α
Very thin granulate	1
Thin granulate	2
Coarse granulate	3
Compacte	4

β - factor for soil permeability class (1 to 6, where 1 is very slow and 6 is fast).

Table 51 – Parameter β to determinate soil erodibility.

Permeabilidade	β
Very slow	1
Slow	2
Slow to moderate	3
Moderate	4
Moderate to fast	5
Fast	6

Results are shown in Table 52[47]. We estimated soil erodibility for the four major texture classes.

Table 52 – Soil erodibility for the four situations analysed.

Texture	α	β	**K (%SOM = 1)**	**K (%SOM = 3)**
Clay	Very thin granulate	Very slow	0.001	-0.001
Silt	Thin granulate	Moderate	0.021	0.018
Loam	Compacte	Slow to moderate	0.072	0.061
Sand	Coarse granulate	Fast	0.016	0.015

K – Soil erodibility; SOM – Soil Organic Matter.

3.6.2 Effect on the parameter C

According to Pimenta (1998b), factor C for permanent pastures is 0.02, while for zones with spontaneous herbaceous or shrub vegetation is 0.05. While the first factor is our best guess for SBPPRL, the second one must be corrected to better depict the NG system. Also according to Pimenta (1998b), the factor C rises to 0.40 for croplands due to tillage. NG require some tillage operation for shrub control, which we considered to happen every five years (twice every ten years). Therefore, the average factor C is 0.12 (20% x 0.40 + 80% x 0.05). Therefore, the parameter C is 83% lower in SBPPRL than NG.

3.6.3 Effect on the parameter P

Parameter P refers to specific practices aimed at lowering soil erosion. Even though it could be argued that this pasture system is, in itself, a management practice to diminish erosion, there are few values available for this parameter. To our knowledge, only parameters for cross slope cultivation, contour farming and strip-cropping have been estimated. Since these are not comparable with the SBPPRL system, we have to use the default value for P, which is 1, in both scenarios (SBPPRL and NG).

[47] Note that parameters α and β were subjective choices, determined from the correspondence with each texture class.

3.6.4 Combined effect

To understand how much soil loss is avoided by the use of SBPPRL, we must calculate all parameters in the USLE. Therefore, we made some assumptions to calculate the value of A in a general case.

We assumed an average total yearly precipitation of 500 mm. Using Equation (3.20), we obtain an R of 1 017.7 $MJ.mm.ha^{-1}.h^{-1}.yr^{-1}$. We also considered a slope steepness of 5% and a slope length of 50 m, thus obtaining an m of 0.5 (Tomás and Coutinho, 1993), using Equation (3.21). Results for average soil loss, obtained with Equation (3.19), are shown in Table 53.

The absolute results are very dependent on soil erodibility (parameter K), since changes in K will have proportional differences in soil loss (parameter A). Therefore, K controls how much soil is lost. The difference between SBPPRL and NG is also sinfluenced by K, but it is mostly the management factor (parameter C) that is responsible for less soil to be lost in SBPPRL. Therefore, according to the USLE, SOM increases decrease soil loss, but the most important effect is that of no-tillage.

Table 53 – Average soil loss in NG and SBPPRL.

Texture	Average soil loss ($kg.ha^{-1}.yr^{-1}$)		
	NG	SBPPRL	Difference
Clay	8	0	8
Silt	174	25	149
Loam	598	84	514
Sand	133	21	112

NG – Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.

Two additional facts must be stressed regarding soil loss. When topsoil is lost, organic matter and nutrients are also carried away with the mineral materials. Therefore, erosion also causes SOM loss and phosphorus loss. Phosphorus has the opposite electrical charge that mineral soil particles do. Therefore, is becomes adsorbed to the soil, and so phosphorus losses occur through surface soil erosion. Even though adsorbed phosphorus is largely unusable to plants, decreasing soil loss is also a way to decrease nutrient loss.

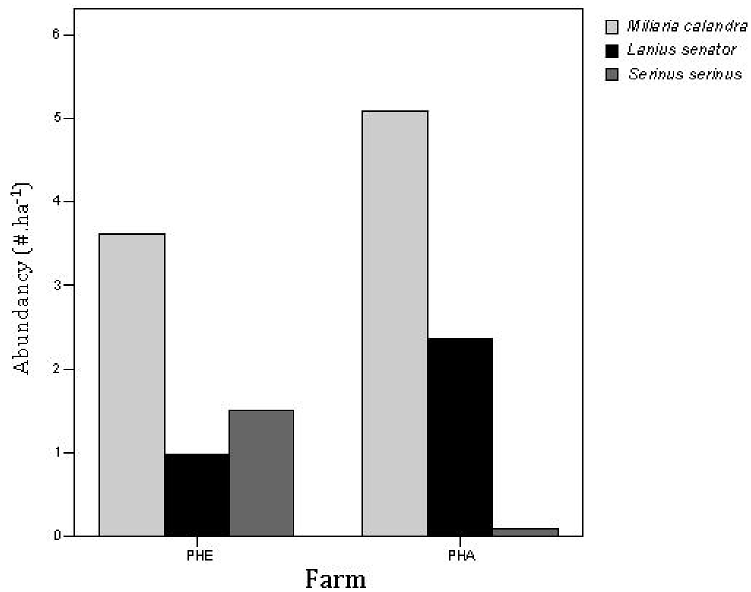
3.7 Effects on biodiversity

The effects of grassland system in wild animal biodiversity were studied in the course of Project Extensity. All results briefly explained next were obtained in studies conducted by Liga para a Protecção da Natureza (LPN).

Two types of bio-indicators were used, namely insects and birds. For coleopterous insects, pitfall traps were placed in adjacent plots of SBPPRL and NG. Birds common in agricultural zones were indentified through expert listening, also in adjacent plots of both types of grassland. This experimental setting was repeated at four regions, namely Lower Alentejo (Castro Verde), Upper Alentejo (Portalegre), Central Alentejo (Montemor-o-Novo) and Cova da Beira (Covilhã). These four locations represent four ecossystem types: pseudo-steppe (Castro Verde), *montado* (Montemor-o-Novo and Portalegre), and mountain (Covilhã).

It could be argued that, since SBPPRL are more productive, and livestock stocking rates may be higher, the quantity and diversity of birds would decrease. However, results obtained did not show significant differences in what respects to birds, as shown in Figure 32. Bird presence in grasslands seems to be more correlated with factors that do not respect to grassland type, such as the presence of trees, places to nidify, fences or other locations to rest. This is not surprising, because birds have a large area of activity, much larger than average pasture plots.

Figure 32 – Bird biodiversity in natural and sown pastures (using three species as indicators).

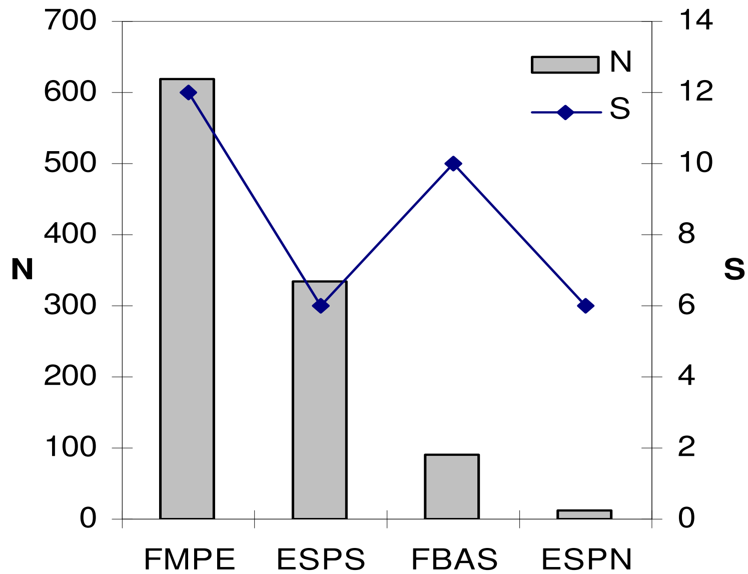


PHE – SBPPRL, Organic, (Northern Alentejo); PHA – Natural pastures, conventional (Northern Alentejo, adjacent farms)
Source: Henriques *et al.* (2008).

Regarding insects, Figure 33 shows that SBPPRL have a statistically significant higher quantity (N) than NG. The sown pasture in the sample ESPS is comparable with the natural pasture ESPN, since they were in the same region. While the specific richness (S) is similar in both farms, the number of insects is much lower in ESPN. Part of this result is also due to the fact that organic farming is practiced in ESPS. The NG in the organic farm FMPE in Northern Alentejo had more quantity and richness of insect species than the conventional farm FBAS.

Again, this confirms our initial intuition – that more productivity, together with more organic matter in soils, yields livelier micro-habitats for wild soil fauna. Soil fauna, on the other hand, is also crucial for SOM increases and stability. As Bot and Benitez (2005) notice, soil fauna is crucial for creating channels for air and water, and also for promoting the binding of mineral and organic particles. While channels are particularly important for no-tilled soils, which are more compact, binding particles decreases erosion and increases water holding capacity.

Figure 33 – Insect biodiversity in natural and sown pastures.



N – Abundancy; S – Specific richness.
FMPE – Natural pastures, organic farming (Central Alentejo); ESPS - SBPPRL, organic farming (Northern Alentejo); FBAS – Natural pastures, conventional farming (Central Alentejo); ESPN – Natural pastures, conventional farming (Northern Alentejo).
Source: Henriques *et al.* (2008).

Even though these results do not point to a clear answer of which grassland system is better, they do show the importance of landscape planning as an integrated system where all elements matter. Most grassland systems can have positive effects on biodiversity, if they are complemented by other elements such as tree cover. We referred previously that it is very likely that the SBPPRL and NG systems always coexist in the same farm. Biodiversity benefits the most from this case, since there are more niches available, and also because there are areas richer in insects (bird feed), and others with lower stocking rates (bird getaway). In this respect, SBPPRL and NG are complementary and not competitors.

3.8 Synthesis of results and discussion

In this chapter, we quantified some environmental effects of pasture systems. First, we converted previously obtained SOM increase for SBPPRL into its equivalent carbon sequestration. We determined that on average rainfed SBPPRL sequester 5 t CO_2.ha^{-1}.yr^{-1}. This allowed us to estimate that existing SBPPRL areas in Portugal have sequestered more than 3.5 Mt CO_2 from 1996 to 2008. Afterwards, we determined the overall balance, by subtracting from the sequestration the emissions from animals, legumes and liming. The balance is still positive – 4.1 t CO_2.ha^{-1}.yr^{-1} sequestered.

This potential of SBPPRL to sequester carbon must be compared with that of other land uses and management activities. Soil carbon sequestration in no-till soils with crop residues was estimated as 7.7-8.5 t CO_2.ha^{-1}.yr^{-1}, and without crop residue cover is 1.9-

2.1 t CO_2.ha^{-1}.yr^{-1}. However, due to lack of data, it is unknown for how long these increases occur (when the maximum saturation is reached). Besides, we only considered the top 10 cm layer for SBPPRL, and in no-till crops 30 cm is the important depth for SOM determination (since roots go deeper). Regarding other land uses, we could not find significant evidence that there is any SOC dynamics.

We also showed that there is also an environmental advantage in sowing irrigated pastures, when we compare them to their alternative land use which is maize production. This effect comes up only when studying the whole production life cycle. This led us to calculate indirect environmental impacts also for rainfed SBPPRL, using LCA. We concluded that, for all impact themes, the impact of a scenario where farmers sown part of their farm with SBPPRL to replace commercial feeds is lower. The only significant impact of SBPPRL is the production of phosphate fertilizer.

Two environmental impacts, though, were left off the LCA method use: soil and biodiversity loss. So, we used the USLE to conclude that soils with SBPPRL are less eroded than soils with natural grasslands. We also reviewed results that show that some insect species are benefited by SBPPRL and, at least, there is no loss in some bird species in SBPPRL plots.

Therefore, in this Chapter we conclude that the whole system of SBPPRL (rainfed) or SIP (irrigated) is responsible for the sequestration of more carbon than their respective alternatives (NG or maize, respectively), and do so with positive environmental effects in all studied themes. We could not find any environmental impact theme where SBPPRL cause significantly more damage than NG. On the contrary, for most of them (carbon sequestration, soil loss) they provide environmental services.

4. Paying for the environmental services of SBPPRL

In Chapter 4 we show the several ways through which farmers can obtain revenue from pastures due to the fact that they provide environmental services. Namely, they can use direct agricultural support, they can sell products which consumers value due to its production method, and they can be paid for carbon they sequester.

We discuss consumer valuation of meat products from SBPPRL, before designing a scheme for payments for carbon sequestration in SBPPRL by the PCF. We conclude by determining the impact of project in terms of expected carbon sequestration.

4.1 Support of SBPPRL by rural development policies

The current Portuguese Rural Development Programme (MADRP, 2007) for the period 2007-2013 stipulate specific amounts to support the installation and maintenance of SBPPRL, but only if they are managed according to two specific agricultural norms: integrated production and organic farming. Base values for the maintenance of natural grasslands and SBPPRL are shown in Table 54.

Table 54 – Public support for the maintenance of natural grasslands and SBPPRL.

Land use	Integrated Production (€.ha^{-1})	Organic farming (€.ha^{-1})	Base area (ha)
Permanent (natural) grassland	106	172	30
SBPPRL	130	210	30

SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes.
Source: MADRP (2007).

These values should be read as follows: for a farmer with a grassland area inferior to the base area, the indicated values will apply. If a farmer has a superior area that that indicated, the values per hectare drop. The modulation factors are shown in Table 55.

Table 55 – % of total sum attributed to each area class.

Area class, in relation to Base Area (BA)	% of total sum
≤ BA	100
> BA and ≤ 2BA	80
> 2BA and ≤ 5BA	50
> 5BA	20

The major difference in the level of support to both types of grasslands lies in the income loss for the adoption of these systems, in relation to a conventional production method. Difference lies in the need for correct phosphate (P) fertilization, especially in SBPPRL. The value for the first area class under integrated production is consistent with the one we used for running costs (130 €.ha^{-1}.yr^{-1}), which means that public financing was designed to cover running costs.

Public financing was designed in collaboration with PNAC's objectives. This means that grasslands supported by the Rural Development Programme will be fulfilling PNAC's goals. All additional carbon sequestered in grasslands has to be contracted outside from this kind of public financing.

4.2 Consumer valuation

Farmers sometimes argue that payments for environmental services, just like all forms of public support, are not enough to guarantee security in their activity (Nuno Rodrigues, personal communication). Farmers seem to prefer what they see as the main result of their activity, which is the direct revenue of whatever products they produce. Therefore, it is important to determine if we can expect an increase in revenue due to the fact that meat is produced in SBPPRL. Consumers may perceive those meat products as more valuable than conventionally produced meat, as they do with organic meat, for example, for two reasons: higher product quality, and due to being sustainable. While the first reason is straightforward[48], the second reason has been increasingly study, as it is getting more valued by some types of consumers by the hour (Harris, 2007).

A first attempt to study this subject was done during Project Extensity, by directly enquiring consumers on their preferences and willingness to pay for meat products (Jorge *et al.*, 2006; Silva *et al.*, 2006, 2007, 2008a, 2008b). Results of these studies are summed up next.

It is important to explain that during Project Extensity a norm was developed for sustainable beef production (Domingos *et al*., 2005). This norm, named “Guaranteed Sustainability” (GSN), was based on the Forest Stewardship Council (FSC) norm for forest products. In that sense, it was guided by two main principles: the congregation of economic, environmental and social aspects, and the involvement of major stakeholders. It was subscribed by governmental entities, an association of farmers, a representative of consumers, and an environmentalist group. This norm is based on the integrated production method, but it goes beyond it by introducing some obligations in what regards to soil and biodiversity protection. It also highly promoted the use of no-tillage for annual forages and crops, and SBPPRL as the basis for animal feed. This Norm also sets some procedures for handling manure: facilities should take into account the daily production of effluents and that storage is necessary whilst manure application in soil is not advisable (3 to 4 months), assuring protection from rainwater. Also, manure piles must be kept at given distances from water streams, drains, fountains or wells. By following these measures, GHG emissions by leaching and runoffs are avoided, and only gaseous emissions are relevant.

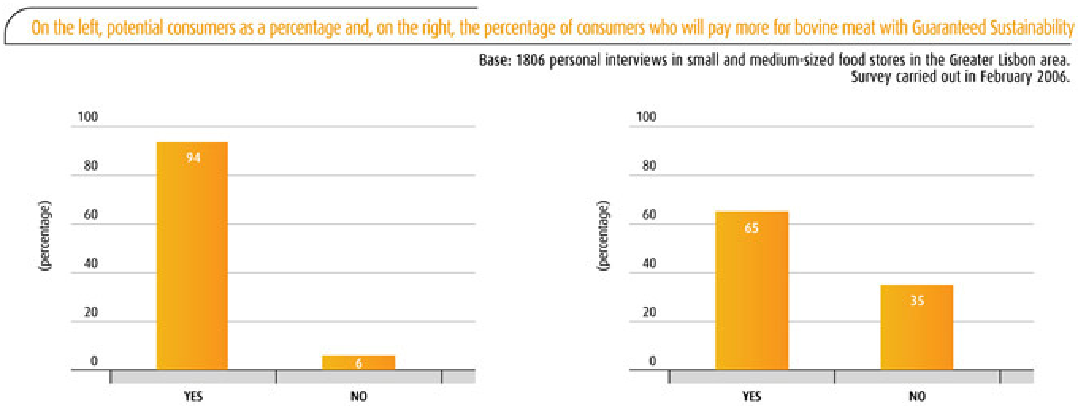
It was the specifications and obligations of the GSN that were evaluated by consumers in the surveys done during Project Extensity. Surveys were conducted by direct interviewing, either in person or by telephone. Surveys done in person took place in stores where meat was sold, in the Lisbon and Porto regions. Surveys done via telephone were done all over the country, and each had over 1 000 validated interviewees.

Results have shown that conventional bovine meat, with no specific label, is the most consumed type of meat. Meat produced by organic farming, together with meat from other certification systems, is no more than 30% of the total. The main reasons pointed by consumers to justify this consumption pattern are the high prices and low availability of high quality meat. However, consumers valued the principals that guide

[48] Note that, even though consumers may recognize that higher intrinsic quality is a reason to increase the price of the product, but it still does not mean that they are willing to pay more for it.

the GSN (on average, graded as a 4 in a scale of 1 to 5). 94% of surveyed consumers claimed to be potential consumers of meat produced following the GSN (Figure 34).

Figure 34 – On the left, potential consumers as a percentage and, on the right, the percentage of consumers who will pay more for bovine meat with GSN.



Survey carried out in the Lisbon region in February 2006. 1 806 interviews were carried out.

Another survey carried out 1 718 interviews in person, also in small and medium stores. When presented with the average conventional meat price, 79.3% of inquired consumers claimed to be willing to pay more than 10 €.kg^{-1} for GSN bovine meat. In this sample and on average, interviewed consumers claimed to be willing to pay about more 3.5 €.kg^{-1} for GSN meat than for conventional meat. However, due to budget restrictions of households, this would mean a decrease in meat consumption. A demand curve for GSN meat was built, showing that the monthly consumption of meat is reduced by 0.8 kg for each Euro paid in excess of the conventional meat price.

A telephone survey, in which 755 interviews were conducted in the beginning of 2007, showed that consumers highly value the fact that animals are allowed to graze in pastures instead of being stabled. In fact, 83% of inquired consumers claimed to be willing to pay more (about 1.5 €.kg^{-1} on average) for meat of an animal that isn't stabled after weaning. However, the main reason pointed is meat quality and not animal well-being. Afterwards, it was explained that animal feed based in green forages or pastures produces meat which is darker, while concentrated feeds are responsible for animals with clearer meat. When asked which one inquired consumers prefer, 89% claimed to prefer the first one.

In the context of this thesis, these results cannot be applied directly. Values were obtained for a whole bundle of characteristics which together make the GSN. They were never specifically asked whether they value SBPPRL over NG. However, these results show that, when explained, consumers prefer the characteristics that are accentuated by feeding animals in SBPPRL, namely the time spent grazing, and also the darker tone of the meat.

We have stated before that the darker tone of the meat comes from more grazing months for steers, and that has environmental advantages since it withdraws livestock from intensive production (Section 3.2.2.9). But we have also mentioned before that darker tones of meat seem to discourage retailers from buying animals from farmers

(Section 1.6). However, when asked wether they would prefer a dark meat over a red meat, if the dark meat meant that steers were kept in pastures, consumers answered yes.

These results hint that, in principle, consumers are willing to chose meat from SBPPRL over other types of meat, if it is presented as a package (like the GSN), and if it means that the meat has a higher quality and taste. The environmental advantages of products from SBPPRL have been shown throughout this thesis, and the quality and health standards of those products has also been studied during the course of Project Extensity. Results of food chain monitoring done in the context of Project Extensity (Ralha *et al.*, 2008) showed that meat from livestock fed in SBPPRL is of higher quality, and richer in saturated fatty acids.

4.3 Designing payments for carbon sequestration

So, farmers who install SBPPRL can resort to agricultural support and to increased market revenue due to higher quality (taste and health, for example) of their products. But is it also possible that farmers obtain direct payments for the environmental services they provide? Carbon sequestration seems to be the service to inspect, since carbon is the only environmental commodity with an established market. Next, we briefly describe how such contracts for environmental payments are established, why the Portuguese Carbon Fund (PCF) is interested in paying for carbon from SBPPRL, and finally how a Project was conceived and approved by the PCF, and what its expected results are[49].

4.3.1 Designing contracts for resource conservation

In standard economic theory, environmental co-effects of human activities are often considered externalities, in the sense that they are not reflected in commodity prices (Perman *et al.*, 1996). As this effect aggravates, we say that there is a market failure, since the current state of business is incapable of internalizing that impact. There are two main ways usually appointed to cope with this market failure: environmental taxes, and environmental contracts.

An environmental tax is imposed on polluters as a way of internalizing the environmental damage (Perman *et al.*, 1996). The tax will increase production costs, incentivating producers to switch to cleaner techniques, and the collected funds will be used to mitigate the damage effects. But it is a normative regime, and the state is required to regulate and control its operationalizitation. Alternatively, polluters may be given the option to compensate their impacts by participating in other markets created specifically for environmental assets. These markets involve a third party, the suppliers of environmental assets, with who are established contracts for the implementation of measures with positive effects.

The last case constitutes the universe of private resource conservation contracts, either voluntary (for example, biodiversity protection) or mandatory (for example, carbon emissions reduction). But even in this kind of contracts, the state plays a crucial role, since it is usually the source of most funds to such markets. Resource conservation is

[49] The next sections are partially based on a Msc. Thesis in Economics (Teixeira, 2008).

very dependant of state intervention, since in fact most environmental goods are public and non-exclusive.

It is not a surprise that most resource conservation revolves around agricultural practices and forestry activities. The agro-forestry sector has a very close link to direct environmental effects, and regulates many of the possible ecosystem services, like water cycle regulation, water quality, soil protection, carbon sequestration, air quality, and biodiversity. However, not all transactions involving environmental services from agriculture and forestry are efficient (Gulati and Vercammen, 2006).

The issue of resource conservation contracts has been fairly debated in the economic literature of the last decades. According to Antle *et al.* (2003), there are two types of costs regarding the implementation of contracts for the adoption of good environmental practices. The first type is farm opportunity costs, which includes the cost of the conversion to the required agricultural practice. The second type is contract costs, which include transaction, monitorization costs, and brokerage fees.

There are mainly two mechanisms for the payments to occur: on a “per hectare” base, and on a “per objective” base (Antle *et al.,* 2003). The per hectare scheme consists on a fixed payment for an area where a farmer adopts a land use or management practice leading to environmental benefits. This type of contract is mainly used when there is no specific monitorization method, or it is not possible to correctly assess the result of the adopted practice. The per objective scheme involves a payment based on the accomplishment of environmental objectives, measured in quantitative indicators. For example, for carbon sequestration, this would mean a “per ton” of CO_2 contract. Every time a farmer would switch its land use or farming practice, he would receive a payment for each unit of carbon incorporated in the soil. Models and field measurements are used to assess the respective sequestration (Antle and McCarl, 2002).

The difference between the two types of contract is that in the first one the farmer receives a payment regardless of the accomplishment of any environmental goal (for example, carbon sequestration). Therefore, the entity that finances the project assumes the risk, and there is only the monitorization cost of assuring that the farmer does indeed adopt the practice contracted. In the second type of contract, only the effective environmental gain is paid for, and it is the farmer who assumes the risk, which also means he has a real incentive to correctly apply management practices. There is, however, a higher cost of monitorization, due to the need of environmental studies (for example, soil analysis for SOM determination). It is thus unclear which of the two types of contract is more efficient, in the sense that more carbon is sequestered by the same total amount paid (Antle and McCarl, 2002).

4.3.2 The case of carbon sequestration

The United States of America did not sign the Kyoto Protocol (KP), but established in 2001 a voluntary initiative that, alongside other measures, included the possibility of carbon sequestration by forests and in grasslands and croplands (Mooney *et al.*, 2002). This initiative is private, consisting on contracts between industries that emit greenhouse gases (GHG) and farmers. But its contribution is not neglectable. Lal *et al.* (1998) indicate that carbon sequestration in agricultural soils could, alone, decrease U.S. emissions by 8%. The credits purchased in this way have prices competitive with those obtained by forest sequestration (Antle *et al.*, 2002).

So, in these resource conservation contracts relating to carbon sequestration, two questions are usually appointed: which economic incentives induce farmers to appoint management techniques that increase soil CO_2?, and would that form of sequestration be economically competitive with other forms of emission reduction (Antle and McCarl, 2002; Marland *et al.*, 2001a)?

Lewandrowski *et al.* (2004) studied which practices would be favoured by farmers for different ranges of incentives in the United States of America. They find that higher payments for each ton of carbon make afforestation the most adopted activity, while for lower payments changes in rotation and tillage practices occur. McCarl and Sands (2007) conclude the same: lower CO_2 prices make cropland and grassland management extremely attractive carbon sinks. They confirm that agricultural sequestration may be an economically effective carbon sink.

However, there are some critics of the use of land use, land use change and forestry (LULUCF, now named AFOLU, which stands for "agriculture, forestry and other land uses") practices for carbon sequestration. Their argument usually revolves around the issue of non-permanence.

4.3.3 The permanence issue

It is necessary to take into account that some ways to sequester carbon do not necessarily correspond to a permanent decrease. This is the case for carbon storage in masses of water (Herzog *et al.*, 2003), and also for soil carbon sequestration (Blanco and Forner, 2000; Ellis, 2001). The problem is that if there is a reversion of practices, such as the use of tillage for cropland management or deforestation for forest management, sequestered carbon will be emitted again (Antle and McCarl, 2002). Furthermore, soils may be net emitters in certain climatic adverse years. This is usually referred to as "leakage", and Murray *et al.* (2004) estimate that, for forestry activities, it may be as high as 90%.

One way to address this issue was proposed by Blanco and Forner (2000) and Chomitz (2000), who all use the concept of temporary or expiring credits. Temporary credits are basically a way to buy time while cleaner technologies are set in place, keeping all the environmental benefits of permanent sequestration. Many have followed to use this concept. Marland *et al.* (2001b) consider temporary sequestration to be completely different from permanent avoided emissions. They propose a market for temporary sequestration credits (which is equivalent to a rental market), separate from the carbon emissions credits market. More recently, Maréchal and Hecq (2006) picked up this idea, proposing the issuing of temporary credits from AFOLU activities.

But expiring credits create another market for carbon sequestration, parallel to the market of emissions reduction. Other authors have found other ways to cope with the issue of permanence. The other option is to use an equivalence factor (Kim *et al.*, 2007), which considers that a ton of carbon sequestered during a certain number of years is equivalent to a permanent reduction (Moura-Costa and Wilson, 2000). This is known as the ton-year accounting method, and has spawned several studies which yielded equivalence times from 42 to 150 years (Maréchal and Hecq, 2006). This means that there is a high uncertainty, and that for longer time spans sequestration projects become less interesting.

There are other alternatives, like the average storing capacity method (the main difference is that the variation in carbon stocks is used to generate credits) or liability mechanisms (in which each country would have to compensate AFOLU emissions by a reduction elsewhere), minimum duration or buffer credits (Maréchal and Hecq, 2006), but none of them are consensual or even desirable. Others determine equivalence factors. Keller *et al.* (2003) recognize that CO_2 sequestration is not a perfect substitute for the avoidance of CO_2 emissions, but still believe that they should be compared. In order to do so, they define an efficiency factor for CO_2 sequestration as the ratio between economically equivalent avoided and sequestered emissions. They find that afforestation is only about 60% efficient, while sequestration in water masses is about 90% efficient.

4.3.4 Problems with carbon contracts

The major problem with this type of contract is the fact that carbon sequestration may not be as easily measured as an emissions reduction in a point-source, like a factory, or above ground forest biomass (Mooney *et al.*, 2002). Direct measurement, either by flux measurement or soil analysis, is very expensive, and so many times the only possibly way to verify if a farmer does comply with the contract is to observe its practices. However, the only way to relate practices with sequestration is by using fixed sequestration factors, which usually underestimate the potential of soils, and do not consider how well the farmer manages his land. Therefore, the issue of efficiency and trade-off between monitorization costs and net CO_2 sequestration is not linear.

There is also a consequence related with contracts that only target one environmental objective, which is the fact that they ignore co-effects. Sometimes these co-effects are negative, but in the case of carbon sequestration in soils they are usually positive. Land uses and management practices that enhance the soil's carbon pool typically also reduce soil erosion, nutrient leaching and runoff, and increase the soil water retention potential (as shown in the previous chapter). The co-effects are not negligible, but they remain as positive externalities in carbon contracts (Feng *et al.*, 2007).

It is also important to notice that farmers who adopt techniques leading to carbon sequestration may face an important increase in productivity, due to the fact that an increased SOM content improves the soil fertility status (Lal *et al.*, 1998). This is usually a direct effect which, in this type of contracts, becomes a positive externality.

There are mainly two types of contracts available to farmers: private and public contracts. Private contracts occur via voluntary schemes, in which private firms finance carbon sequestration in soils (Antle and McCarl, 2002). The Portuguese state attributed maximum emissions levels for all polluting firms, but sequestration projects do not decrease their normative target. Therefore, private firms finance these projects mainly for image purposes, and in the process they help Portugal achieve their KP target for free. We present one such case in the next section, which is the contract established between EDP and Terraprima. As for public contracts, they refer to economic payments from the government, specifically with the objective of promoting carbon sequestration. One example of such is the Portuguese Carbon Fund (PCF), which we will address further down the chapter.

4.3.5 Laying the ground: The EDP-Terraprima Project

In 2006, a contract was established between EDP, the main electricity company in the country, and Terraprima, a small firm with agro-forestry activities in Quinta da França (QF), Portugal. It was the first private contract for carbon sequestration in all LULUCF practices in Portugal (forest, cropland and grassland management). It was also the first private contract in Portugal to finance no-tillage and SBPPRL as carbon sinks. In this contract, EDP will finance in the period 2006-2012 projects regarding forest management, cropland management and grassland management on a partial "per hectare" basis. Terraprima undertakes frequent monitorization of its carbon stocks, and is paid according to the effective fixation.

The yearly payment has two components: a fixed part (F_t) and a variable part (V_t). The fixed part is defined for 2006 and 2007 as $F_t = 45\,000\,€$, and for 2008 to 2012 as

$$F_t = R_t \times \min\left(c_t, 3000\ \mathrm{tCO_2}\right), \tag{4.1}$$

where R_t is 15 €.t^{-1} CO_2 in 2006, and is actualized in each year by the national consumer price index, and c_t is the amount of fixed carbon in year t.

The variable part is calculated for each year t, according to

$$V_t = y \times \min(x_t, X_t), \tag{4.2}$$

where y is 6 €.t^{-1} CO_2 when x_t and X_t are inferior to 1500 t CO_2, and 7 €.t^{-1} CO_2 otherwise, and x_t is defined as the difference between carbon sequestration in year t and 3000 tCO_2, and X_t is

$$X_{2006} = x_{2006}, \tag{4.3}$$

$$X_t = X_{t-1} - V_{t-1}/y + x_t,\ t > 2006. \tag{4.4}$$

By definition, if x_t or X_t is negative, then V_t is zero, which means that there is no payment. Prices obtained for the variable part are updated proportionally to the Powernext price variations.

This was the first experience in Portugal of a contract for carbon sequestration in grasslands, which are our case study. Prices and amounts involved in the contract were negotiated between the two firms, even though they reflect the carbon market price fluctuations. However, Terraprima was given a choice to sub-contract part of the carbon sequestration, if Quinta da França was not enough to fulfil the total quantity of contracted CO_2, which were 7 000 tons per year. Terraprima has chosen to do so. There are now six other sub-contracted farms.

In the case of public contracts, we wish to determine the optimum price and quantity of sequestered carbon based on market considerations that arise from the existence of multiple options and multiple farmers. This is the subject of the next section.

4.4 Why was the Portuguese Carbon Fund interested?

The Portuguese State wants to guarantee that the KP goals are achieved efficiently, in the sense of Antle and Mooney (2002): the government strives to maximize social benefits from carbon sequestration per unit of resource used.

Therefore, considering the state as a buyer of sequestration credits, Portugal will acquire them as long as the final price of each ton is lower than the reference price.

We define the reference price as the lower price between other available domestic projects and the lower possible price for international projects. We assume that, if the grassland carbon price is equal to the reference price, the state will prefer to finance grassland sequestration in Portugal, if the alternative is international carbon, due to its co-benefits and due to the fact that it is a national project.

But the Portuguese State needs to address the permanence issue. Even though the KP does not differentiate sequestration from emissions reduction, each State is not indifferent between both. If a non-permanent option is chosen, and all sequestered carbon is emitted after the KP period, then in a possible future period of a second agreement the country will be obliged to compensate for the carbon lost in some other way. This may be seen, in a negative way, as a double cost for the country. But it may also be seen, in a positive way, as buying time.

In the case of SBPPRL, even though it is highly implausible that all farmers who install them now would change land use after 2012, there is no guarantee otherwise, and so a risk of re-emission in the following periods after the KP has to be adressed. In order to determine the maximum price that the state would be interested in paying to farmers, we considered the worst case scenario, which is that after 5 years of payments, all farmers decide to switch practices in such a way that all sequestered carbon is lost[50]. This would mean that the PCF would only be financing temporary sequestration to borrow time. In a future period after the KP expires, Portugal will either continue to finance farmers to keep SBPPRL, or buy carbon credits, depending on which goals and trade mechanisms are defined for after 2012.

We obtain that price by using the concept of Net Present Value (NPV) (Perman *et al.*, 1996):

$$NPV = \sum_{t=1}^{T} A \cdot (1+r)^{-t} , \tag{4.5}$$

where A is the annuity value (€), r is the discount rate and T is the time horizon (years). The NPV of five years of sequestration, plus the updated value of buying an equivalent quantity of credits, must be the same for the state as obtaining the same quantity of credits in the present, instead of financing sequestration. In our case, the A term in Equation (4.5) is equal to the unit price of carbon (€.t^{-1} CO_2) times the number of tonnes sequestered (t CO_2).

We must now make one of two assumptions. Either the PCF would pay farmers only at the end of the period, or divide the payment throughout the period. We begin by formulating the first option, that farmers will only be paid at the end of the KP accounting period (6th year), as

$$p \cdot (1+r)^{-6} \sum_{t=1}^{5} C(t) + p_6 \cdot (1+r)^{-6} \sum_{t=1}^{5} C(t) = p_0 \cdot \sum_{t=1}^{5} C(t) , \tag{4.6}$$

[50] We will mention next that the actual PCF Project is only valid between 2009 and 2010. However, the application stage was launched in early 2008, and therefore the programme was being prepared for some time before the regulation was published. Therefore, we assumed that we should replicate in this study the same initial conditions that guided the decision of the establishment of the PCF, and consider the case of a 5-year payment.

where p is the price that the PCF is interested in paying farmers (€.t^{-1} CO_2), p_6 is the estimated carbon price in the sixth year (2013), p_0 is the reference price in 2008, $C(t)$ is the amount sequestered in year t (t CO_2.ha^{-1}), and r is the discount rate (for the PCF). The choice of discount rate is crucial, as we shall see. Therefore, we used four different values for r, namely 1%, 3%, 5% and 7%. Using Equation (4.6), we obtain p, since

$$p = \frac{p_0 - p_6 \cdot (1+r)^{-6}}{(1+r)^{-6}}. \tag{4.7}$$

Note that, under this assumption of payments to farmers in the sixth year, the price is independent of the quantity sequestered. Regarding the reference prices, the carbon prices in the present and future market has mostly been oscilating around 20 €.t^{-1} (according to the Bluenext website, http://www.bluenext.fr/). We used several combinations of three values for p_0 and p_6, namely 15, 20 and 25 €.

Results obtained are shown in Table 56. If both reference prices p_0 and p_6 are equal to 20 €, p varies from 1.23 to 10.01 €.t^{-1} CO_2. If both reference prices are lower (15 €), p decreases, and if reference prices are higher (25 €), then p increases. A change in both prices of one quarter (5 €) implies a change in p which is also close to a quarter. Therefore, the price that the PCF is willing to pay follows closely the reference prices. If the reference prices change independently, this result is even more expressive. If the estimated carbon price in the sixth year is higher than the price in the beginning of the project, than for some interest rates the PCF should not be interested in buying any carbon. But if the estimation is that the price is lower, than even for low discount rates the PCF would pay a high price for carbon sequestered.

Table 56 – Maximum carbon price (paid in the sixth year), depending on the discount rate.

	Scenario	r 1%	r 3%	r 5%	r 7%
p (€.t^{-1})	p_0 = 20 € p_6 = 20 €	1.23	3.88	6.80	10.01
	p_0 = 15 € p_6 = 15 €	0.92	2.91	5.10	7.51
	p_0 = 25 € p_6 = 25 €	1.54	4.85	8.50	12.52
	p_0 = 15 € p_6 = 20 €	-4.08	-2.09	0.10	2.51
	p_0 = 20 € p_6 = 15 €	6.23	8.88	11.80	15.01

r – discount rate (PCF); p – price paid by the Portuguese Carbon Fund for carbon sequestration in pastures; p_0 – carbon price in the present; p_6 – carbon price at the end of the project.

The second possible assumption is that the PCF would pay farmers in a yearly basis. This is translated by the equivalente to Equation (4.6), which is

$$\sum_{t=1}^{5} p \cdot (1+r)^{-(t+1)} \cdot C(t) + p_6 \cdot (1+r)^{-6} \sum_{t=1}^{5} C(t) = p_0 \cdot \sum_{t=1}^{5} C(t). \tag{4.8}$$

In this case, we obtain p using the expression

$$p = \frac{\left[p_0 - p_6 \cdot (1+r)^{-6} \right] \cdot \sum_{t=1}^{5} C(t)}{\sum_{t=1}^{5} (1+r)^{-(t+1)} \cdot C(t)} . \tag{4.9}$$

In this case, the price p depends of the total carbon sequestered. For any $r > 0$, $\sum_{t=1}^{5} C(t) > \sum_{t=1}^{5} (1+r)^{-(t+1)} \cdot C(t)$, which means that p increases with the total carbon sequestered. And price p also depends of the distribution of sequestration during the project. For any $r > 0$, the quantity $\sum_{t=1}^{5} (1+r)^{-(t+1)} \cdot C(t)$ is maximum when sequestration occurs soon (since the amount sequestered is subjected to a smaller discount). Therefore, if more carbon is sequestered early than late, the PCF will pay less for each ton of carbon.

Again doing the same analysis, and using for C sequestration the average sequestration potential of 5 t $CO_2.ha^{-1}.yr^{-1}$, we obtain results in Table 57. Overall results are quantitatively similar to those in Table 56, albeit relatively lower. This second approach is also approximately independent of the total carbon sequestered.

Table 57 – Maximum carbon price (paid yearly), depending on the discount rate.

	Scenario	r			
		1%	3%	5%	7%
p (€.t-1)	p_0 = 20 € p_6 = 20 €	1.21	3.66	6.15	8.71
	p_0 = 15 € p_6 = 15 €	0.90	2.74	4.62	6.53
	p_0 = 25 € p_6 = 25 €	1.51	4.57	7.69	10.88
	p_0 = 15 € p_6 = 20 €	-4.00	-1.97	0.09	2.18
	p_0 = 20 € p_6 = 15 €	6.11	8.36	10.68	13.05

r – discount rate (PCF); p – price paid by the Portuguese Carbon Fund for carbon sequestration in pastures; p_0 – carbon price in the present; p_6 – carbon price at the end of the project.

Therefore, since it simplifies calculations and graphical representations, and the values are very similar, we opted for the first assumption (payment in the sixth year). From here on, we also assume constant yearly prices of 20 € for both p_0 and p_6 (since they are intermediate estimates).

Note that considering that all carbon will be lost in 2012 is a very conservative approach. It is highly unlikely that all farmers will till their plots. This is important because if we assume that only a fraction α of the carbon which was accumulated during the project is lost after 2012, then the price p will increase, because in that case

$$p' = \frac{p_0 - \alpha \cdot p_6 \cdot (1+r)^{-6}}{(1+r)^{-6}} . \tag{4.10}$$

Since $\alpha > 0$, $p' < p$. By assuming that $\alpha = 1$ in the calculations above, we are considering the worst case scenario.

In this case, to the PCF, every ton of carbon paid at this price is equally worthwhile, since marginal costs are constant. Therefore, the demand curve[51] would be a horizontal line. Considering that there were no physical constraints, Portugal would be interested in financing sequestration in SBPPRL at the prices stated before until the KP deficit was compensated (3.73 Mt CO_2e). However, this is not the case. Carbon sequestration supported by the PCF has to be additional to carbon sequestration considered in PNAC, which is 0.5 Mt CO_2 for cropland and grassland management. The Portuguese Rural Development Programme predicts a supported area of SBPPRL of 70 000 ha, which we will assume to be the cut-off point, i.e., the point after which all sequestered carbon is additional[52].

However, at the time of this study, we considered that, for a matter of equality and justice, the PCF would predictably have to pay all farmers for the carbon they sequester, regardless of Rural Development support. Such is to say that whichever price is fixed, it will also be paid to the first 70 000 ha, even though their sequestration was already planned and will not be additional.

For simplicity purposes, we consider that those 70 000 ha will always be accounted using a fixed conservative factor of 5 t $CO_2.ha.^{-1}.yr^{-1}$. Therefore, we must adjust the equilibrium point to cope with this fact. Table 58 shows the quantity of carbon that is not additional to PNAC's objectives, depending on the use of a factor or a model.

Table 58 – Maximum price paid by the PCF for carbon sequestration additional to PNAC's objectives.

Average sequestration ($t\ CO_2.ha^{-1}.yr^{-1}$)	Area (ha)	Not additional carbon ($Mt\ CO_2.yr^{-1}$)	Real price to be paid ($€.Mt^{-1}\ CO_2$)
5.0	70 000	0.35	$p^* = p \cdot \frac{C - 0.35}{C}$

r – discount rate (PCF); p – price paid by the Portuguese Carbon Fund for carbon sequestration in pastures; p – price paid by the Portuguese Carbon Fund if all carbon sequestered is additional; p* - real price for additional carbon; C – total carbon sequestered in pastutes.

Since when the PCF pays price p there is a certain quantity of carbon that is not being truly being accounted for Kyoto purposes, a real price must be defined, p*, which is the maximum price (€) per ton of carbon that the PCF is willing to spend. This real price is a function of the model used of the total quantity of sequestered carbon. The expressions in Table 58 show that, as C increases, this effect decreases. Therefore, we may plot a demand curve, as shown in Figure 35. In our demand curve, the PCF will not be interested in paying for carbon sequestration in SBPPRL until the base sequestration is met. From then on, it will be progressively interested in paying more, since the effect of the first 70 000 ha is diluted in the larger area of interested farmers. The upper limit for the real price is the maximum price, indicated in Table 56. But when the total deficit is met, the PCF will stop financing any project.

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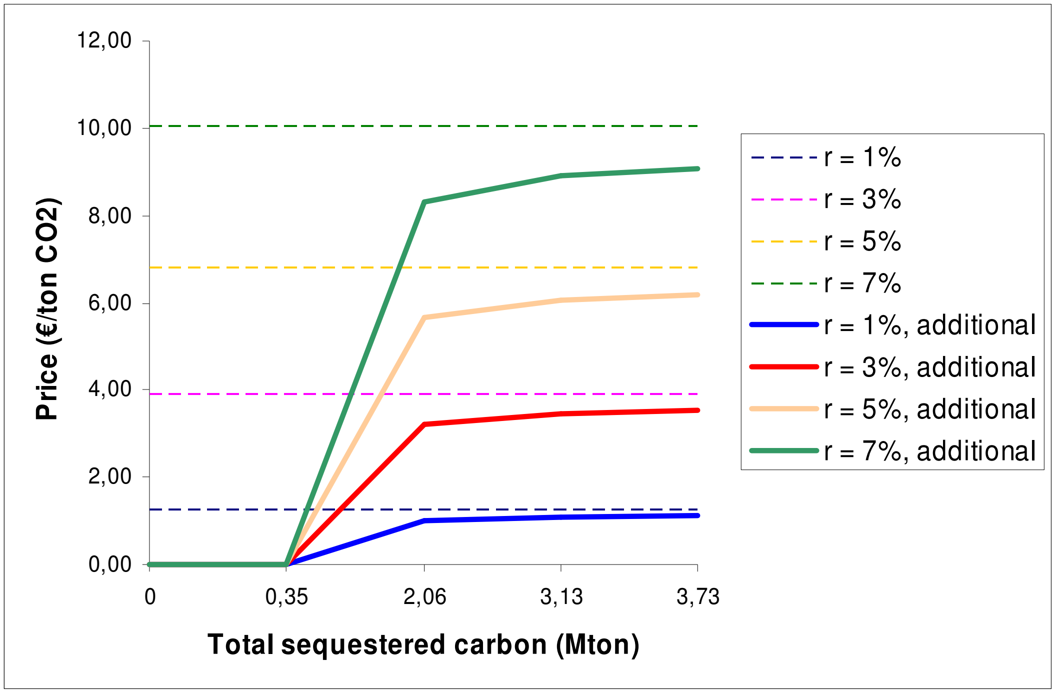
[51] We define supply curve for carbon sequestration in SBPPRL as a representation of the total carbon sequestered as a function of the payment per ton.

[52] Note that in practice it may not be the case, since the target is common for cropland and grassland management. It is the combination of both that must add up to 0.5 Mton, and so even it may not suffice to achieve the target for the implementation of SBPPRL. We will assume that cropland management achieves its part of the goal.

Figure 35 – Demand curve for carbon sequestration in SBPPRL, depending on the discount rate and total carbon sequestered.



r – discount rate (PCF).

4.5 Designing the Terraprima-PCF Project

The PCF launched the application stage in 2008. In October 2008, the firm Terraprima, taking advantage of its experience managing the EDP contract mentioned before, did make a proposal to the PCF for payments for carbon sequestration in SBPPRL, for a Project that would start in 2009.

The proposal, however, was quite different from what was expected at the time this work was carried out. Our work was based on the principle that no farmer with SBPPRL could be excluded, for fairness reasons. However, the PCF included in the regulation a mandatory clause that obliges all projects to start only after the proposal is delievered. Such is to say that only new pastures could be included in the proposal (pastures installed from 2009 on).

The next section shows the calculations which underlined the PCF project that was approved.

4.5.1 Proving adittionality

There are three additionality criteria set by the PCF. The first one is project additionality, which requires that the project does not happen without support from the PCF. The second one is regulation additionality, which implies that the project is not mandatory due to some regulation or legal obligation. The third one is financial additionality. This criterium requires the proponent to show that the level of the payment is enough to significantly influence its realization, but not so much that it completely depends on it to be maintained after the PCF support finishes. We turn our focus to this last criterium.

We said before than one explanation why the installation of new SBPPRL has decreased is that for some farmers (namely those with lower stocking rates and which do not recur to public support) installing pastures never has a positive economic balance. This may be further illustrated by depicting average costs and revenues from SBPPRL. We chose to make this balance for the scenario when the steer is sold after weaning, because it is the most common scenario in Extensity farms, and for consistency with the rest of this thesis. The balance comprises three instances: pasture installation, pasture maintenance, and costs related to livestock. We chose a time span for calculations of 10 years, since it is the approximate average time that well-managed SBPPRL can be sustained without new installation. Costs and revenues will be determined for a functional unit of 1 hectare.

Note that in these calculations we do not consider the future revenue from increases in production, as a consequence of SOM increases and better soil structure and fertility in general. This effect, though being very important, may not be visible to farmers during the first 10 years after pasture installation.

4.5.1.1. Pasture installation

The installation of SBPPRL requires the following field operations:

- First-year tillage, necessary to prepare the field for sowing and fertilizing;
- Liming, when pH(H_2O) is lower than 5.3, in order to correct soil acidity levels;
- Cover fertilization, using most commonly phosphorus, borax[53] and zinc sulphate[54];
- Sowing of the biodiverse seed mix;
- Rolling, after all other operations, to compact the soil.

The cost per hectare of these operations is shown in Table 59. Table 59 was built using field data from the farm Quinta da França for prices and Carneiro *et al.* (2005) for input quantities (Nuno Rodrigues, personal communication). Costs include inputs, transportation, machinery used and labour.

[53] Legumes are highly sensitive to borax concentration in soils. Therefore, borax should be applied in soils with concentrations lower than 0.40 ppm (INIAP-LQARS, 2006).

[54] Grasses are sensitive to zinc concentration in soils. Therefore, zinc should be applied in soils with concentrations lower than 1.4 ppm (INIAP-LQARS, 2006).

Table 59 – Average costs of operations required for the installation of SBPPRL.

Operation	Nr.	Labour cost (€)	Machinery cost (€)			Input costs				Total cost per operation (€)
			Variable cost	Total fixed cost	Ammortization	Input	Quantity	Unit cost (€)	Total cost (€)	
Lime transportation	1	8.20	8.88	7.37	6.80					31.25
Liming	1	1.08	0.90	0.86	0.80	Dolomitic lime (t)	2.0	65.52	131.04	134.67
Tillage	2	28.69	13.22	10.01	6.80					58.72
Rolling	1	14.34	13.22	10.01	6.80					44.37
Fertilizer transportation	1	4.78	2.34	1.95	1.79					10.86
Cover fertilization	1	9.56	7.33	6.16	4.42	Superphosphate 18 % (kg)	200.0	0.41	82.40	109.87
						Borax (kg)	10.0	0.35	3.52	3.52
						Zinc sulphate (kg)	7.0	1.82	12.72	12.72
Sowing	1	15.78	8.08	10.45	7.75	AC 700 Seed mix by Fertiprado[c] (25 kg)	1.0	107.83	107.83	149.89
									Total	555.87

This value must be added to other costs, which according to the GPP crop sheets are the interest of circulating capital (equal to 2.94 €, using an interest rate of capital of 1.5%) and general costs, which amount to 5% of the total costs with inputs (in this case, general costs are 16.88 €). Therefore, the total gross cost of installing a SBPPRL is approximately 575.68 €. We then converted this value into an annualization (10 years, discount rate 5%), which resulted in an average cost of 74.55 €.yr^{-1}.

However, PRODER supports 25% of the installation costs. Considering this support, the total net cost is 431.76 €, or 55.92 €.yr^{-1} (10 years, discount rate 5%).

4.5.1.2. Pasture maintenance

A well-managed pasture will only require one yearly event of fertilization with phosphorus. The costs respecting to this operation are shown in Table 60. Note that to assume that the same quantity of phosphorus is applied during 10 years is an overestimation. Towards the end of the period, if the pasture is well-managed, a cycle within the pasture will be established that recycles phosphorus flowing between the soil, plants and animals, and returning to the soil.

Table 60 – Average costs of operations required during maintenance of SBPPRL.

Operation	Nr.yr^{-1}	Labour cost (€)	Machinery costs (€)			Input costs (€)				Total costs per operation (€)
			Variable cost	Total fixed cost	Ammortizations	Input	Quantity	Unit cost (€)	Total cost (€)	
Fertilizer transportation	1	4.78	2.34	1.95	1.79					10.86
Cover fertilization	1	9.56	7.33	6.16	4.42	Superphosphate (kg)	200.0	0.41	82.40	109.88
									Total	112.74

The cost of phosphorus is particularly important. The reference unit cost in Table 60 for phosphorus is 0.41 €.kg^{-1}. Maintenance costs are 112.74 €.yr^{-1}. But if the price doubles, then maintenance costs become 195.14 €.yr^{-1}. Therefore, doubling the cost of phosphorus will increase 73% the maintenance costs, or around 50% the total costs related to pastures (maintenance and installation – installation also becomes more expensive).

This justifies why the price of an input such as phosphorus may be a crucial factor in the decision of a farmer to install SBPPRL. And it also explains why, in years such as 2008, farmers prices skyrocketed, and so many farmers already with SBPPRL skipped fertilization (Pedro Silveira, personnal communication). In the first years of settlement, this may be a serious threat to the permanence of the pasture.

There are other costs which are not directly for maintenance, but are also yearly costs. These costs are the rent (we assumed a rent of 39.90 €.ha^{-1}.yr^{-1}), circulating capital interest and general costs. Adding all contributions, the yearly costs with pastures are 165.45 €.ha^{-1}.yr^{-1}.

4.5.1.3. Costs related to livestock

We considered that each cow has a steer per year. The steer is weaned and sold at 6 or 7 months old. Several costs are associated with the existence of livestock in the pasture, namely silos for forage and feed storage (to complement cow feeding), establishment of plots (e.g fences), and labour costs with the farm manager and cowherd. The following costs were considered:

- While they are in the pasture, steers are fed only with milk. Cows, however, are fed by the pasture itself during its most productive monts (7 months) and by feed during the rest (5 months). As a simplification, we assumed that the supplementation is entirely provided by silage maize, and that each cow consumes approximately 2.6 t.yr^{-1}. Assuming an indicative price of 35 €.t^{-1} for silage maize, the total cost with feeds is 92 €.yr^{-1}.
- Three silos, which store silage maize for 62 cows, cost 10 000 € and are paid in 12 years.
- Setting a cow plot of about 65 ha, costs about 11 900 €, which are paid in 20 years. The fences for the plot cost about 18 200 €, paid in 10 years.
- A full-time cowherd can manage a plot of around 150 cows. The average salary of a full-time cowherd is 10 545 €, including social expenses.
- The farm manager spends around 10% of its working time managing a farm with 62 cows. The estimated average cost of that ammount of time is 1 400 €.
- Sanitary costs are around 9.3 €.cow^{-1}.yr^{-1}.

All values were annualized using the respective time frame and a discount rate of 5%. Even assuming the average figures shown before, the cost for each farm is still very dependent on the number of cows and total farm size, which is the same to say that it is very dependent on the stocking rate (LU.ha^{-1}). Therefore, we spawned cost scenarios for six possible stocking rates: 0.15; 0.3; 0.5; 0.7; 1.0 and 1.5 LU.ha^{-1}.

4.5.1.4. Revenue from SBPPRL

There are two sources of revenue for farmers. The first one is a direct payment for each breeding cow of 230 €.cow^{-1}.yr^{-1}. The second one comes from selling a steer each year. The price is highly volatile and variable, and therefore we studied three scenarios: 250 €, 375 € and 500 €. Therefore, the revenue depends on the cow stocking rate (more cows equals more steers per hectare) and on the price itself for which each steer is sold.

Note that we did not consider as source of revenue the specific payment for SBPPRL from PRODER. Only organic or integrated production farmers are eligible for this support, and those are not only the minority, but also have different costs and revenues from those shown here. We also did not consider activity maintenance payments[55] since they are equal for SBPPRL and other alternative land uses (natural pastures or cropland).

4.5.1.5. Support from the PCF

Farmers who install SBPPRL in 2009 will receive 50 €.ha^{-1} during the first 3 years after installation. We annualized this value for 10 years with a 5% discount rate, in order to make it comparable with the other costs and revenue. It is then equal to 17.59 €.ha^{-1}.yr^{-1}. As for farmers who install SBPPRL in 2010, they receive 75 €.ha^{-1} during the first 3 years after installation. Again annualizing this value, we obtain 12.01 €.ha^{-1}.yr^{-1}.

4.5.1.6. Final balance

Table 61 reviews and synthesises all the previous information for costs and revenue.

Table 61 – Synthesis of costs and revenue of producing steers in SBPPRL.

	Item	€.ha^{-1}	Item	€.ha^{-1}
Revenue	Payment from the PCF – Sowing in 2009	17.59 €	Steer sale	250 € - 500 €
	Payment from the PCF – Sowing in 2010	12.01 €	Support for breeding cows	230.0 €
Cost	Maintenance costs	165.5 €	Sanitary costs	18.5 €
	Installation costs (without support)	74.6 €	Cow feed	91.4 €
	Installation costs (with support)	55.9 €	Silos for maize silage	18.2 €
	Fences and other plot costs	51.0 €	Cowherd labour	70.3 €
			Manager labour	22.6 €

Table 62 shows why we needed to generate several scenarios throughout the previous exposition of the costs and revenues of SBPPRL. For example, for each class of farm size and number of cows there is a different balance (balances for all scenarios are shown in Appendix V – Economic balances for steer production in SBPPRL). We

[55] "Regime de Pagamento Único", in Portuguese.

now must determine the average scenario for Portuguese farmers with pastures. In order to do so, we make the following assumptions:

- Within each stocking rate class, 80% of farmers receive the installation support from PRODER and 20% do not;
- 25% of farms support a stocking rate of 0.5 $LU.ha^{-1}$, 60% of farms support 0.7 $LU.ha^{-1}$, and 15% of farms support 1.0 $LU.ha^{-1}$;
- 25% of steers are sold for 250 €, 50% for 375 €, and 25% for 500 €.

Under these assumptions, we obtain the global balance in Table 62. On average, installing SBPPRL has a negative economic balance, on a yearly basis and for a 10 year time frame. However, even though support from the PCF is only received during the first 2 or 3 years, it is enough to make the balance positive, both for farmers sowing in 2009 and in 2010.

Table 62 – Final balance between costs and revenue for SBPPRL.

Scenario	Balance ($€.ha^{-1}.yr^{-1}$)
Without support from the PCF	-9.2
With support from the PCF (sowing in 2009)	8.4
With support from the PCF (sowing in 2010)	2.8

It should be noticed that these calculations were made under the assumption that prices will be constant for 10 years. The expected value for each of the costs and revenues considered here has a very high variance, but since the farmer must decide at present whether to install pastures or not, and he has no information available as to how prices will evolve, it seems plausible to use present values and assume them constant throughout the time frame considered.

4.6 How much carbon will be sequestered with the PCF project?

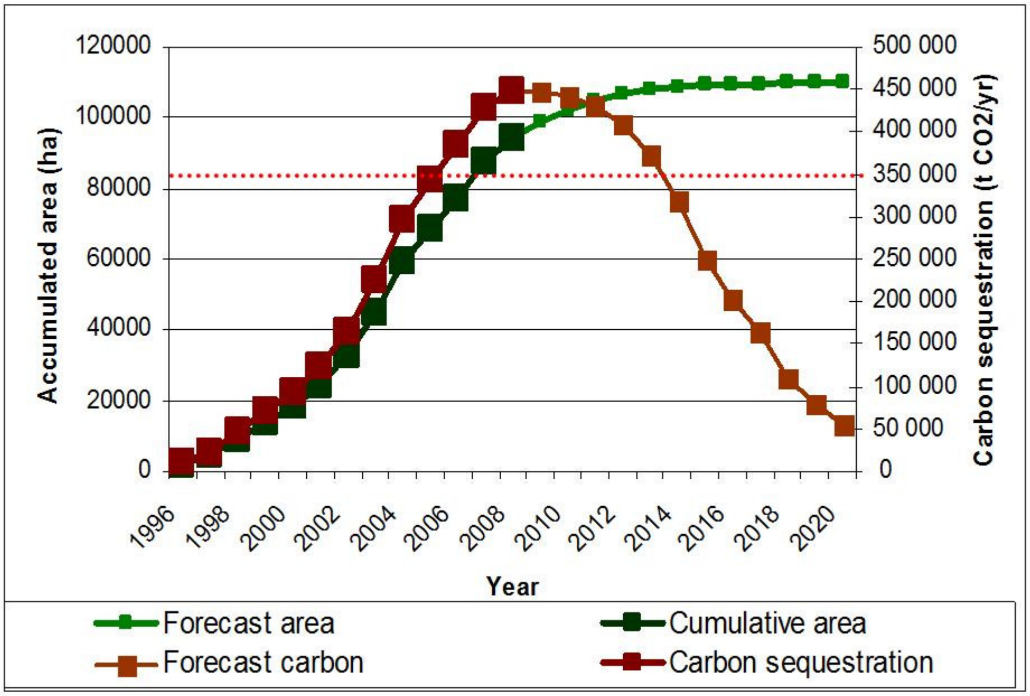
The project estimates the installation of 42.000 hectares, half of which in 2009 and the other half in 2010. 2011 was left off because farmers would only benefit of the support for one year. The area estimate was obtained from inquiring farmers from the Extensity Project and associations of farmers.

Considering the payments refered above (50 and 75 $€.ha^{-1}$, for installations in 2009 and 2010, respectively), farmers will be paid around 5-6 € per ton of CO_2 sequestered, and the project will be responsible for the sequestration of about 0.91 Mt CO_2[56].

To show how this project changes the reference situation, we can use the logistic model established in Section 1.4.4 for the area increase of SBPPRL in the absence of specific payments. This model allows us to forecast how much area would be installed in the future. Multiplying the area of SBPPRL by the specific sequestration factors determined in 3, and summing the yearly contribution of pastures from all ages, we obtain Figure 36. The PNAC target is shown in the dotted line. We can see that, throughout the whole period, it would be complied, but from 2012 on the area would stabilize and carbon sequestration would decrease steeply as the pastures age.

[56] Results for SOM increases in SBPPRL from the model in Teixeira *et al.* (2008), and respective carbon sequestration equivalent determined in Teixeira (2008), were used in these calculations.

Figure 36 – Baseline area and carbon sequestration of SBPPRL.



The PCF project will cause a structure break in this series, as Figure 37 shows. The area installed per year has increased until 2005, and then started decreasing, following a trend that quickly approaches zero. Due to the PCF project, more are will be installed than in any other year in the records. Figure 38 shows the accumulated area of SBPPRL obtained from Figure 37.

Figure 37 – Area of SBPPRL installed yearly, observed, modelled and due to the PCF project.

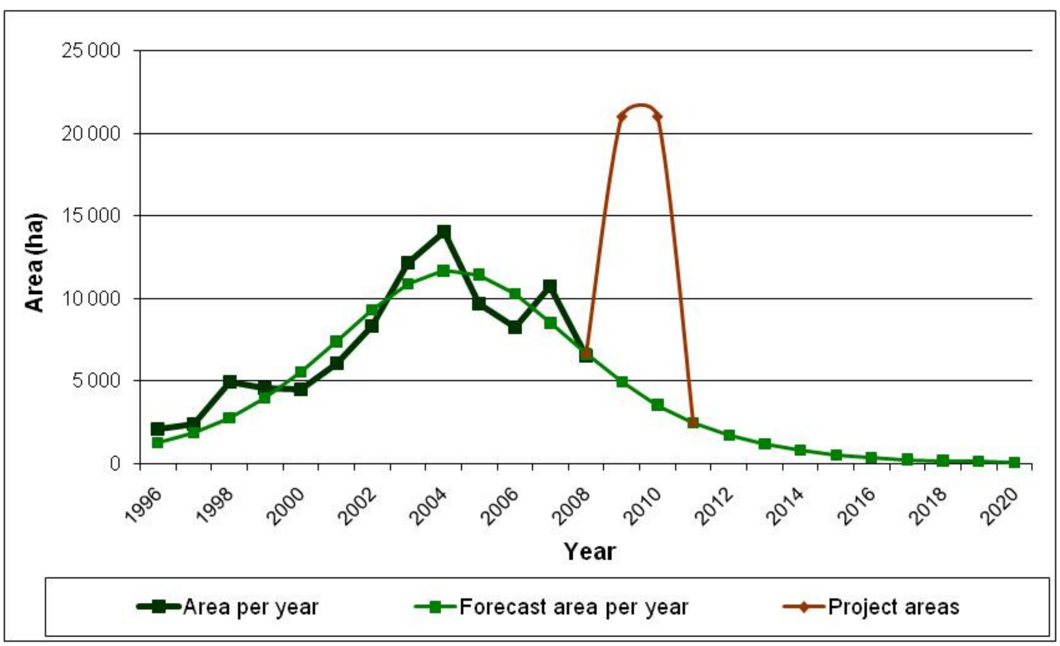
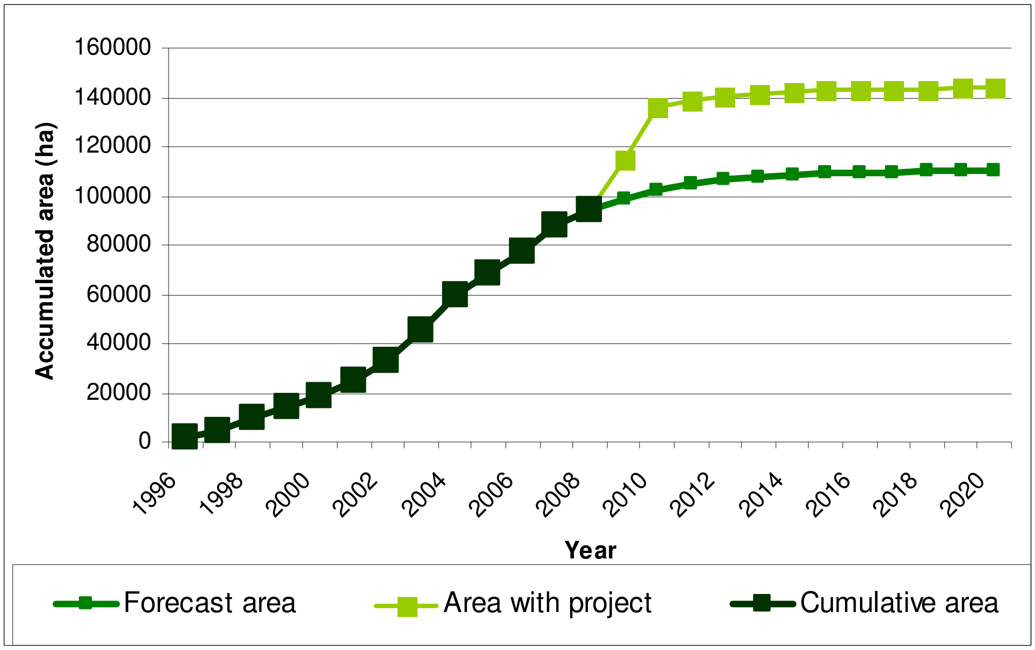


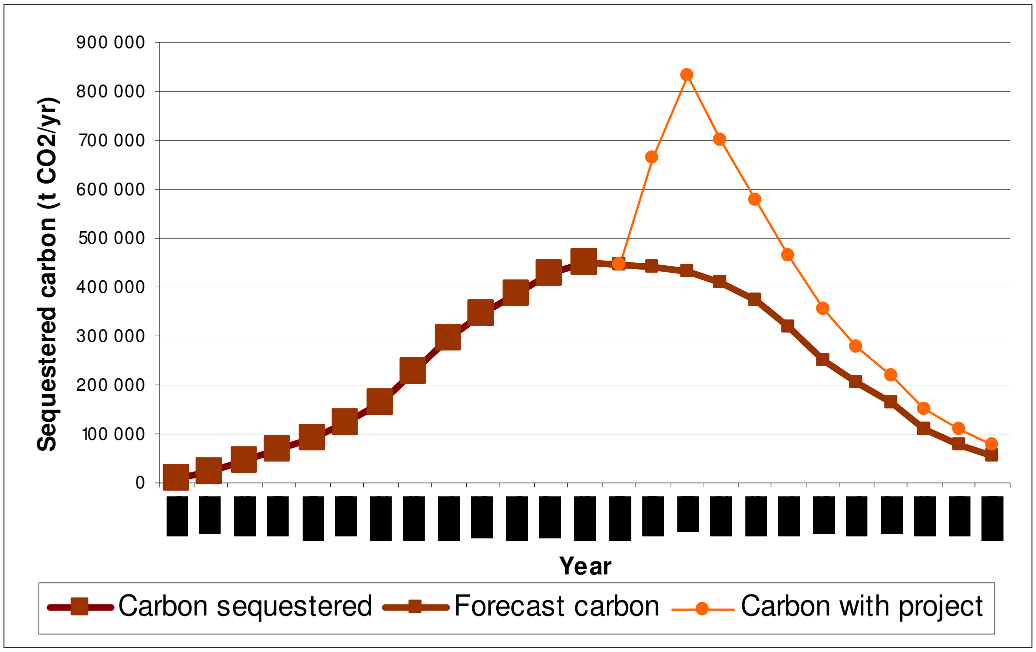
Figure 38 – Accumulated area of SBPPRL, observed, modelled and due to the PCF project.



The result in terms of total carbon sequestered per year is a very high increase from the expected sequestration that would occur in the absence of the project, as shown in Figure 39. This additional amount of carbon sequestered continues after 2012. If we assumed that all farmers kept their pastures, than an extra 0.67 Mt CO_2 would be sequestered between 2013 and 2020. This means that each ton sequestered has a lower price to the Portuguese State than it seems, since there will be free additional carbon remaining after the end of the Kyoto period.

Note that we considered for calculations in Section 4.4 that all area would be lost after 2012. At the time, we argued that it was a very conservative approach, since farmers will hardly change land use after such an investment (note that we studied viability for 10 years). Farmers in the PCF project are techinally supported to better manage their pastures. This technical support guarantees that the output of the pasture is maximized, and transferes knowledge to farmers which they may use for years to come. This fact, together with the financial support, almost nulls the risk of complete or almost complete loss of area.

Figure 39 – Total carbon sequestered per year in the area of SBPPRL, observed, modelled and due to the PCF project.



A final word goes to notice that the PCF will support SBPPRL just due to carbon sequestration, but Portugal will benefit from the other environmental and socio-economic positivie effects for years to come. Other ecosystem services will remain as externalities until new markets are created and new work is done to create options for the valuation of those services.

4.7 Synthesis of results and discussion

In this chapter we began by making a simple economic comparison between SBPPRL and NG. We monetized the environmental impacts obtained in the previous Chapter, and then internalized them in the economic balance. We found that SBPPRL represent an overall lower cost than NG, because they have lower economic costs and also lower environmental costs. It was curious to notice that environmental costs are much lower than the direct economic costs. We were then sure that SBPPRL were a win-win solution.

We then turned to improving the benefits of SBPPRL. We briefly mentioned the valuation that consumers make of meat products produced in SBPPRL, only to switch then our focus to payments for public goods. We studied the issue of resource conservation contracts, namely contracts for carbon sequestration in SBPPRL. We referred the case of the private contract established between EDP and Terraprima, and introduced the possibility of the PCF to incentive farmers to adopt this system.

In all calculations, we used the concept of NPV. Fist, we determined the demand curve, which is a horizontal line that translated the fact that the Portuguese state will finance every ton of sequestered carbon in pastures for the same price, as long as it is lower than the alternative. This result is independent of the contract scenario, but is extremely dependent of the discount rate used, due to the fact that carbon sequestration is only temporary.

This study, however, assumed that all farmers had to be paid. The regulation of PCF imposed that only new SBPPRL could apply. Therefore, the actual project was different from this study. Under this project, 0.91 t CO_2 more will be sequestered in Portugal until 2012.

5. Conclusions

The last Chapter is mainly dedicated to conclusions, but also intends to make a bridge to future studies by stating the main limitations and unanswered research questions. We start by summarizing the main results obtained, in order to draw conclusions from their integration. We state the main conclusions and policy recommendations, before acknowledging the shortcomings of our work, and proposing ways to answer questions that we left unanswered. In the end, we will wrap our work up by listing the contributions of this thesis.

5.1 Summary of results and conclusions

At the present moment, there is an ongoing debate on which are the best strategies to mitigate climate change. Some of those strategies do not take into consideration a full scale sustainability assessment considering all environmental aspects and the economic trade-offs involved in the project. Some projects for emissions reduction or carbon sequestration have negative effects in other environmental themes, which doubtfully compensate the benefit in terms of carbon emissions. The best example of this is biofuel production, a subject we refer on Section 3.3 for the case of bioethanol from maize, and which is now widely regarded as an unsustainable option for climate change mitigation.

Therefore, optimum options for climate change mitigation should have other environmental co-benefits. For emissions reduction, this means that projects have to be economically competitive and not have significant risks (carbon capture and storage seems to be an example of a potentially hazardous and expensive project, while N_2O scrubbers in fertilizer industries a very positive project, which is also being paid by the Portuguese Carbon Fund). In the case of carbon sequestration in soils, the general principle of “green carbon” implies that all projects must provide widespread ecosystem services. Ecosystem services in Portugal are crucially dependent on agricultural sustainability. Sustainable land uses must guarantee quality food and other goods, thus increasing farmer revenues, and also co-produce environmental amenities.

Therefore, in this thesis, we study the ecosystem services provided by grasslands, focusing on carbon sequestration. This choice occurred for two main reasons. First, to study carbon sequestration one must first describe soil organic matter (SOM) accumulation in grasslands, and SOM is the key parameter to most environmental services. Second, SOM improves soil structure and restores fertility, which guarantees sustainable increases in production for farmers. Therefore, systems which accelerate SOM increases will have both environmental and economic positive effects. Since SOM increase is also the mechanism through which carbon is stored in soils, we obtain a bundle of co-effects that occur at the same time as carbon sequestration. If those co-effects are positive, than we have found “green carbon”.

It may still be asked: if there is a bundle of ecosystem services to be studied, why focus on carbon? Most of all, because it is the only environmental amenity with an established economic market. Therefore, chosing carbon sequestration in grasslands as the main object of study allowed us to finish the present thesis with a proposal for a connection with a policy measure, namely payments for carbon sequestered in domestic projects for emissions reduction by the Portuguese Carbon Fund (PCF).

Increasing economic revenue is one of the key aspects to obtain agricultural sustainability, since it breaks recent tendencies in land use change in Portugal, which has been moving towards abandonment and more extensive natural (unmanaged) grassland systems. Those have many problems pointed out in Chapter 0 and confirmed for some particular aspects in the next Chapters.

We studied three types of pastures: Sown Biodiverse Permanent Pastures Rich in Legumes (SBPPRL), fertilized natural grasslands (FNG) and natural grasslands (NG). SBPPRL are a Portuguese innovation and have been installed in more than 90 000 ha in the country. Because SBPPRL are more productive, they support a higher stocking rate. We show that the increase in SOM is higher in SBPPRL than in the other grassland systems studied. SBPPRL soils are usually tilled before the pasture is installed (even though no-tillage is possible is some situations), and therefore lose SOM in the beginning. But they quickly recover, due to the increased grass root production, which is a result of the complementary effects of plant biodiversity and functional roles of legumes and grasses.

On average, SBPPRL increase SOM by 0.20 percentage points per year during 10 years. Since SOM increase in the mechanism through which athmospheric carbon is stored in soils, we could convert this increase to equivalent carbon, and found that SBPPRL sequester around 5 t CO_2 per hectare and per year. This is a very high potential. We studied carbon accumulation in other land uses, and found the only management practice that topps this potential is the use of no-tillage and mulching in cropland areas.

However, SBPPRL, when compared with NG, may have increased emissions from livestock, legumes and liming for pH increase. SBPPRL allow an increase in sustainable stocking rate. This may mean that more animals can be fed in SBPPRL plots, thus effectively increasing emissions. But it may also mean that farmers will use less concentrated feeds and keep animals in pasture plots longer, thus decreasing life cycle emissions from feed production. Therefore, emissions from livestock are crucially dependent on the scenario. Even the worst case, however, the sum of all increased emissions in SBPPRL is never high enough to null carbon sequestration. The level of sequestration is enough to guarantee that the SBPPRL, as a whole, is a carbon sink.

But what makes SBPPRL a good environmental and socio-economic policy to mitigate climate change is the fact that there is much more than carbon to it.

Due to SOM increases and to the fact that, unlike in NG, no tillage is required during maintenance for shrub control, less soil is lost in SBPPRL, and soil fauna is more abundant. Decreased erosion and improvements in soil structure mean that there is an improved capacity for holding water, which increases water availability for roots and decreases superficial runoff. This fact yields another positive effect. SBPPRL provide not only mitigation of greenhouse gas emissions, but also adaptation to climate change.

Climate change will alledgedly increase atmospheric CO_2 concentration, average temperature and precipitation (Schils *et al.*, 2008). Some authors argue that these effects will promote vegetable growth, hence increasing productivity and soil quality (Niklaus *et al.*, 2001). However, other authors believe that there are other limiting effects (such as the availability of nitrogen), and since temperature and precipitation distribution and variability will become more irregular, more adaptability to extreme circumnstances will be required from plant species. Existing models are insufficient to determine how soil.plant dynamics will evolve under stress or rapid change (Chapin III *et al.*, 2009).

But that is precisely one of the advantages of SBPPRL – their resilience. Biodiversity and grassland soils with a well established and diverse bank of seeds allows production of each species to adapt to climate circumnstances, and take full advantage of positive environmental factors. Soils with increased fertility and capacity to hold water will be more adaptable to a future with larger temperature and precipitation extremes. Furthermore, nitrogen fixation from legumes diminuishes the probability of grass productivity limitation.

Results for some other environmental themes also show a clear trend. Biodiversity clearly benefits from high SOM, even though results for birds are inconclusive. The territory for birds is larger, and the spatial scope in our study is the plot level, and that makes it harder to find a clear preference for any type of grassland. However, the least that may be concluded is that birds do not seem to prefer NG to SBPPRL. Results for life cycle effects are always favorable to the scenario in which SBPPRL are installed to replace commercial feeds, even though they require more inputs than NG. One very important input, however, is phosphate fertilizer, which has important impacts due to its production and transport.

Other effects, which were unaccounted here, may also favour SBPPRL. Due to livestock pressure, herbaceous competition and high SOM, shrubs seldom grow in SBPPRL. This could seem as one less source of biomass entering the system (since shrubs also use photosynthetic carbon to grow, when compared to the reference situation of NG, easily invaded by shrubs. However, in practice, this is not the case, due to fire and SOM mineralization in NG. While shrub growth inhibition in SBPPRL decreases fire risk, NG are highly susceptible to conduct forest fires. For that reason, NG must be intervened to control shrubs. This usually occurs using tillage. Tillage will mineralize most of the SOM pool, which had been improved using shrub biomass. And, even in that case, there are periodic stages (before each tillage) where fuel is maximum and risk is high.

We are now able to provide justification for the seven statements made in Chapter 1:

1. SBPPRL are more productive than natural pastures, both above and belowground. Higher belowground productivity is translated by SOM, which we showed to increase more in SBPPRL in Chapter 1.
2. Increased SOM and soil cover by pasture plants implies an improved soil structure due to SBPPRL, leading to decreased erosion, better capacity to hold water and consequent flood regulation. Less superficial runoff leads to decreased erosion as well, particularly at extreme rainfall events. This soil loss effect was modeled in Chapter 3, using the Universal Soil Loss Equation.
3. Since SBPPRL are more productive, they feed more animals, and so are typically exploited with high sustainable stocking rates. High stocking rates mean that animals control shrubs either by stomping or by using them in their feed, since shrubs are rich in fiber, to complement for excess protein from the consumption of legumes. Besides, higher productivity of grasses and legumes leaves fewer resources available for shrubs. we discuss these effects through the present thesis, and present data that supports them in Chapter 3.

4. Increased stocking rates in SBPPRL mean that livestock in pastures emits more methane, which is a GHG. There are also emissions due to biological fixation of nitrogen by legumes, and emissions from liming, a process required to increase the pH of soils (if they are too acid). However, the global carbon balance is positive, as we show in Chapter 3.
5. Since SBPPRL are more productive enough, there is less need to resort to concentrated feeds. These feeds are composed mainly of cereals and oilseeds, which required fertilizers to be produced, and which are then avoided. The LCA study of this effects was done in Chapter 3.
6. Higher SOM has a positive effect on soil fauna, but the effects for birds are so far inconclusive. We can at least say that SBPPRL are not worst than natural pastures, as we refer in Chapter 3.
7. High SOM increases mean that carbon sequestration occurs, and we account it in Chapter 3. We studied other land uses in the agricultural sector, and from those only the combination of no-tillage and mulching has a higher potential to sequester carbon than SBPPRL.

Therefore, as a whole system, the scenarios that we studied for SBPPRL (rainfed) or SIP (irrigated) increased soils with more SOM, thus sequestering carbon, more than their respective alternatives (NG or maize, respectively). They do so with positive environmental effects in all studied themes. We could not find any environmental impact theme where SBPPRL cause significantly more damage than NG. On the contrary, for most of them (carbon sequestration, soil loss) they provide environmental services. Therefore, as a whole, SBPPRL are a good all-around sustainable land use. Carbon sequestration is not traded-off with other environmental amenities, but rather a win-win effect. In SBPPRL, we obtain many environmental services in the same bundle. This is why we may coin carbon sequestered in SBPPRL as “green carbon”.

So, after showing that SBPPRL may be considered a sustainable land use, we turned to the possibility of obtaining economic incentives for farmers who install SBPPRL. This may be done in two ways: valuing products which along its production cycle were related to SBPPRL, and payments for the environmental services provided by SBPPRL.

Regarding the private goods produced in SBPPRL, namely meat, consumers seem to value the fact that these pastures are used to produce cattle, as they claim to be willing to pay more for it. However, commercial networks still have to be created to value meat products from livestock produced in SBPPRL. On the public goods provided by SBPPRL, carbon sequestration is the most directly valuable. After previous experience with a smaller scale project, an application of new SBPPRL areas was made to the PCF. We showed that, even facing the risk of non-permanence after the end of the project (in 2012), this project is worthwhile for the PCF. This happens because even if in the highly implausible case that all carbon stored in SBPPRL soil is lost in 2013, the economic benefit alone of buying time to acquire permanent reduction credits is already enough to justify paying for that carbon.

Turning to farmers, we find that, in a 10 year time frame, the revenue obtained from SBPPRL is higher or lower than the costs, depending on three crucial factors, which are the stocking rate in the farm, the total area of pastures in the farm, and the price paid for steers. Larger farmers with higher stocking rates will always have higher revenue from SBPPRL, but for smaller farms when lower prices are paid for steers SBPPRL are not economically positive. This fact justifies two observations. First, that the installation of SBPPRL has decreased its pace in the last years. This is because this system has already been installed in places where it is more prone to be profitable. Second, this justifies the need for payments from the PCF. The payment will effectively turn loss into profit by some scales of farmers, significantly improving the probability of adoption of this system.

The PCF will pay for carbon sequestered in 42.000 ha of new SBPPRL installed in 2009 and 2010. This may be a significant push to increase the area of SBPPRL in the future. In this project, farmers will be supported by technicians who will guide them in using the best management for pasture installation and maintenance. This means that, with this project, one last important issue in improving ecosystem serices will be achieved with SBPPRL, which is transfer of knowledge to farmers.

The overall conclusion we obtain is that SBPPRL are, in general, a more sustainable extensive system of meat production than natural pastures. They maximize the bundle of services provided by grasslands, and are an answer to the decline in ecosystem services in Portugal. Results show that SBPPRL verify most responses pointed to negative conditions and trends identified by the ptMA, namely:

- Changes to sustainable land uses – Our work shows that SBPPRL are a more sustainable land use than the alternative (NG). An increase in their area (due to the increase in revenue and policy changes) will yield widespread envrionmental benefits, one of which is carbon sequestration.
- Increased revenue for private goods – Consumers seem to value the principles of SBPPRL, even though no surveys specifically about this subject were done yet.
- Increased revenue for public goods – Projects in the voluntary carbon market are now available to farmers who correctly manage SBPPRL, and the PCF will pay for carbon sequestered in new areas. Besides, there is a differentiation in the public support to installation and maintenance of SBPPRL instead of natural pastures, which is also mainly due to their environmental benefits.
- Technical support and information management – Farmers in the PCF project receive technical support in the field for the installation and maintenance of SBPPRL.

5.2 Future research plan

In the previous section, and also throughout the present thesis, conclusions were presented identifying several caveats. Those were issues outside the scope of this thesis, or issues in which this thesis was inconclusive. We now turn to the proposal of research questions, suggested as future work, which directly tackle some of the most important lacunas in the knowledge on environmental services of grasslands.

5.2.1 Should we sow pastures all over the country?

Throughout this thesis, the spatial scope was always plot level. This was due to the fact that available data was collected at specific plots, and due to it we disregarded two important effects: interactions with other land uses in the zone, and specificity of the location of the plot.

Regarding interactions, landscape planning is needed. When discussing the application of SBPPRL in the country (for example, via the PCF support), one question arises: do SBPPRL remain a sustainable land use when sown over large continuous areas? Different areas could have different sustainable uses. A mosaic between natural and sown pastures may be better, in overall sustainability terms, than an entire zone of either one of them.

As for local specificities, in some very degraded NG zones, sowing pastures may not even be the best technical option. First, soils shoud be recovered (for example, by substituting tillage for shrub controls), and only then SBPPRL can be installed. Besides, there may be differences between opportunity costs. Some pasture areas have to be analysed relating to the alternative of conversion to forest, while for others the most plausible alternative use is conversion to cropland. For each of these areas, the cost-benefit environmental and economic balance will be different.

Therefore, the effects of large continuous areas of SBPPRL, as well as the regionalization of effects, are still to be determined. Field data would have to be gathered: on soil loss, biodiversity, and large scale effects on SOM and productivity from large areas of each grassland system. Those should be compared with specifically determined alternative land uses.

5.2.2 Is it plant diversity or functional group diversity?

As we have mentioned before in section 5.1, according to the literature, composition (number of functional groups) seems to be more important than functional group richness (number of species in each group). It is the composition that mainly determines the stability (resistance to disturbance) of ecosystem properties and ecosystem resilience (Wardle *et al.*, 2000).

However, in principle, the total number of species in SBPPRL should be important, since in heterogeneous Mediterranean soils each species will find better conditions to grow than others. It is always important to have more than one species with a given ecosystem function but with a different response to ecosystem changes (Hooper *et al.*, 2005). This is known as the “portfolio” effect. Besides, we would not even know which grasses or legumes should be sown, if we were to sow less species in SBPPRL.

In order to test this effect, an experimental setting should be devised using a SBPPRL plot and several plots with pasture composed only of one legume and one grass. The plots should then be compared in terms of aboveground productivity and carbon sequestration.

5.2.3 Other scenarios for sustainable stocking rate increase

We mentioned before, right at the beginning of this work, that SBPPRL are more productive than natural grasslands, and it is possible to increase the sustainable stocking rate. This led to the design of several scenarios to analyse, one of which was effective increase in number of animals. If there truly are increases in stocking rate in SBPPRL, there are several co-effects that must be studied. First, if we consider that the Portuguese self-provisioning level of meat does not change, then the higher stocking rates in SBPPRL could mean that we can feed the same number of animals in a smaller area. Therefore, there is some land sparing. The environmental and economic effects of these marginal lands which are abandoned for meat production must also be accounted.

There is another possibility, which is that Portugal substitutes meat imports with meat from the increased stocking rate in SBPPRL. In this case, there will be more emissions from livestock inputed to Portugal, under the current Kyoto framework. But there is an important economic effect that can balance out this environmental burden.

Therefore, an integrated, sectoral economic model is required to study what is most likely to happen after natural pastures are substituted by SBPPRL:

- Do stocking rates increase, or do farmers maintain the same stoking rate but decrease the concentrated feed supplementation (which depends on feed prices and meat prices)?
- If stocking rates increase, will the resulting meat substitute national or international meat (which depends on meat prices and logistic chains)?
- If stocking rates increase and national meat is substituted, what happens to the land which is no longer used for animal production (which depends on the opportunity cost of each field used for grazing)?
- If stocking rates increase and international meat is substituted, what is the overall economic balance and the excess in emissions (which depends on meat prices and provenience of imports)?

5.2.4 Eat meat or go vegan?

The study of extensive production has a curious side effect. Grazing animals are consuming a resource (grass) which does not directly compete with other sources of human feed (like cereals and oilseeds which compose aggregated animal feeds normaly are). Therefore, the commom energy efficiency argument for direct consumption of vegetable commodities is not applicable. It would only be applicable for intensive livestock production. Even though feeds are still required in extensive animal production, and the inputs for pasture installation and maintenance also use energy, so does the production of crop cultures.

It would be interesting, then, to perform the same kind of sustainability analysis, but applying it to compare vegetarian meals with meat produced in SBPPRL. An LCA approach would be particularly important, but direct modeling of the effects in farms would have to be particularly accurate. There are also differences between marginal effects or widespread effects (for example, if 5% of the population become vegetarian, or if the whole population becomes vegetarian). Land that would marginally change from meat to crop production would have different environmental effects of widespread ending of meat production (end of pastures). An economic assessment of trade effects would also show dramatic differences in the agricultural sector balance.

5.2.5 Trees or grasses?

We have refered in section 3.3 carbon sequestration in grasslands and croplands. One land use, however, was left off – forest land.

The comparison seems, at first, unfair, since forests and grasslands are managed with radically different objectives. The main product of the first is wood or woody products, and the second one meat or meat products. But the comparison is not farfetched because both uses are sometimes combined, like in the case of Portuguese *montado*. In this framework, pastures and grasslands are not different or separated, but rather extreme land uses in a possible continuum of combinations of both. Trees in pastures creates shaded areas, increasing available biota, and biodiversity within forests may also have positive effects (Huston and Marland, 2003).

Therefore, the following question could be asked: what is the optimum rate of tree cover in grassland systems? Or, to put the question the other way around, what's the maximum tree cover where the installation of pastures still has positive effects? This is the fair comparison, especially when speaking of carbon balance of land uses.

5.2.6 Reduced forest fire risk

During a forest fire, large scale emission of greenhouse gases occurs. Using SBPPRL has positive effects on fire hazard for two reasons:

- The grazed agro-forestry mosaic landscape interrupts large areas of forest monocultures prone to fire;
- Unlike in natural pastures, increased grazing pressure in improved grasslands stops invasion by shrubs, which are also good fuel for forest fires.

The use of improved pastures to control fires is already foreseen in several agro-forestry management plans. However, accounting for this effect, as well as monitoring it, is extremely complicated. Only inferences on greenhouse gases' emissions based on the direct comparison with similar burnt areas is possible.

A geostatistical model is necessary to determine the exact effect that SBPPRL plots have in zonal fire risk. The decrease in fire risk can only be accounted by employing statistical models, and taking into consideration the spatial location of the pastures.

There is also another possibility worth testing regarding animal control of shrub invasion. It has been suggested (Carlos Aguiar, personal communication) that livestock grazes shrubs that eventually come up in SBPPRL to compensate for excessive protein (due to the high percentage of legumes in their composition). This effect has to be discerned in test fields, accompanying grazing animals and registering their feeding habits.

5.2.7 Natural or artificial regeneration of *montado*?

We've mentioned that SBPPRL are less prone to be invaded by shrubs. This is due to animal stomp and also due to increased grass production. Plant diversity in itself decreases invasion, as noted by Fargione and Tillman (2005). They also show that some grassland C_4 plants strongly limitate nitrogen for invader's biomass growth.

However, this may also be a limitation for natural regeneration of forest areas. In agro-forestry areas, such as the *montado* system in Portugal, "invasive" plants are primary stages of natural succession that leads to new oak trees. By suppressing their development, there is no need for tillage operations to destroy some of their woody biomass, and there is no fire risk (since they are high combustible). Nevertheless, there is no natural regeneration either. Regeneration requires a natural grassland system, with the disadvantages we described throughout this thesis.

Therefore, it would make sense to devise an experimental setting that would study in two plots whether natural or artificial regeneration of *montado* is more efficient. The first one would be a natural pasture plot in a *montado* ecosystem, were a tillage operation would be performed to control shrubs. However, some shrubs would be protected, so that new trees could grow. The second plot would be a SBPPRL, and so there would be limited to none shrub growth. In this second plot, however, there would be artificial introduction of young trees (in the same number of the "natural" trees protected in the first plot), protected from livestock. The test should be maintained for several years, and in the end the number of trees that survived should be determined, and conclusions taken on which one was more successful.

5.2.8 Optimization of phosphate fertilizer use

Fertilization operations were crucial in the development of agriculture. Fertilizer inputs were essential during the green revolution, propelling food production ahead of population growth. But the environmental costs of pollution are also significant. Those impacts include the degradation of downstream water quality, eutrophication of superficial waters, winter and summer smog and also emission of N_2O, a greenhouse gas (Vitousek *et al.*, 2009).

Most of these problems are due to nitrogen fertilizers. In our work, as we have noticed in section 3.5.4, we found that phosphate fertilizers also have high life cycle impacts. Since they are crucial for the management of SBPPRL, in virtue of the dependence of legumes, their optimisation is crucial. This optimization should include a detailed study of transportation distances, types of fertilizer (single or triple superphosphate, for example), other micronutrients provided by each type of fertilizer, yield output and also economic cost-benefit analysis.

5.2.9 CO_2e emissions reduction due to reduced fertilizer use

As referred before, in section 1.4, the use of an extensive animal production system, using SBPPRL, may decrease the need for concentrated feed supplementation for animals. Therefore, the need for industrial processing of concentrate feed, as well as the need for ingredients, such as cereal grains and oilseeds, decreases. Since such crops are very demanding in terms of fertilization (particularly nitrogen), less concentrate feed demand also means less fertilizer use, and less emissions due to its production and application. We did not account this effect, since it depends on feeds used and actual reduction.

Beside this life cycle effect, the reduction in fertilizer use occurs directly, since improved grasslands are rich in legumes, and these capture nitrogen. Therefore, nitrogen fertilization is not required. Cassman *et al.* (2003) note that mineral nitrogen fertilizers are responsible for direct field emissions (N_2O) and indirect emissions due to fossil fuel consumption during their production and application. Improved grasslands may require the use of phosphorus, potassium and, eventually, other nutrient applications, but the reduction in life cycle use for feed ingredients (which also require phosphate fertilizers) approximately compensates the extra use.

5.2.10 There is so much more than carbon

Our study in this thesis was somewhat asymmetrical, since it privileged the carbon sequestration service over other services provided by SBPPRL. There was, however, a reason to this, which is the connection to the PCF done in Chapter 4. There is now a market for carbon, and so it was the best choice to develop this integrated work, from the sustainability analysis to policy options and economic incentive schemes.

But it is possible to establish the same connection for other ecosystem services of SBPPRL. It is possible and desirable to devise schemes for environmental payments for biodiversity, soil protection or water cycle regulation. To such purpose, the analysis done in this thesis for each of those other services would have to be extended. Since there is no established market for them (there is no "biodiversity fund", for example), that would require an innovative framework.

5.2.11 Closing the cycle – expanding the borders of the analysis

We mentioned before, in section 1.8, that the limits of our analysis (at the gate of the animal farm) account for the vast majority of environmental impacts. However, such may not be the case in economic terms. In fact, it must be recognized that meat produced in SBPPRL and natural pastures have different quality and value. As we mentioned in section 4.2, some surveys have already been done. But they did not report consumer choices on the pasture type itself.

More than inquiring consumers on their preferences, it would be important to study the dynamics of the consumption of both types of meat (from natural pastures and from SBPPRL). It would also be important to determine their impact (if any) on the diet of consumers. If one type of meat is richer in any nutrient or parameter (fibre, protein, digestible energy, etc.), then the equivalent quantity of other source must be considered "spared", and the corresponding impacts removed from the overall balance.

5.2.12 Estimation of errors and uncertainties

Finally, we should notice that we did calculate errors and uncertainties for LCA results. Furthermore, we did some sensitivity analysis when we changed our assumptions and checked if results would be maintained. Those processes altogether assure us of our conclusions.

However, the overall uncertainty of the sustainability analysis was not determined. In fact, when combining several methods of environmental analysis, it is unclear how uncertainty in results should be assessed. Therefore, we suggest as a verification of our results the development of analythical calculations of errors and uncertainties in this work.

5.3 Contributions of the thesis

The present thesis had research contributions and policy contributions. All the work presented in the thesis is original and was prepared as part of the PhD. (even though part of it was published in other instances), except when specifically noted.

The most impactful results had to do with the finding of the sequestration factors, namely for SBPPRL. Those factors are used today for Portugal to account the carbon sequestration is the exhisting areas of this system of grasslands. This work has been developed together with the Working Group to account carbon sequestration in Portugal from agriculture, forestry and other land uses, in the framework of the Kyoto Protocol. The Group is part of the National System of Inventory of Emissions by Sources and Removal by Sinks of Atmospheric Pollutants, and is coordinated by the Portuguese Environmental Agency. The sequestration factors obtained here are also used in the private voluntary carbon market projects, and were also used in the PCF project for new SBPPRL areas. The PCF project itself was also an offspring of the present thesis.

That choice of SBPPRL was important for the establishment of a specific agricultural support for SBPPRL in the Portuguese Rural Development Programme (PRODER) 2007-2013. Part of the PhD. work that led to the present thesis was the Strategic Environmental Assessment of PRODER.

Work done in this thesis was also published in several instances. The main publications are:

- 3 chapters in a book with peer review:
    - Rosas, C., **Teixeira, R.**, Mendes, A.C., Valada, T., Sequeira, E., Teixeira, C., Domingos, T. (2009). *Chap. 7: Agricultura*, in Pereira, H., Domingos, T., Vicente, L., Proença, V. (eds.), *Ecossistemas e Bem-Estar Humano: A Avaliação para Portugal do Millenium Ecosystem Assessment*, Escolar Editora, Lisboa, pp. 213-249.
    - Domingos, T., Valada, T., **Teixeira, R.**, Rodrigues, O., Rodrigues, N., Aguiar, C., Belo, C.C. (2009). *Cap. 19: Quinta da França*, in Pereira, H., Domingos, T. (eds.), *Ecossistemas e Bem-Estar Humano: A Avaliação para Portugal do Millenium Ecosystem Assessment*, Escolar Editora, Lisboa, pp. 661-684.
    - Pereira, H., Domingos, T., Marta, C., Proença, V., Rodrigues, P., Ferreira, M., **Teixeira, R.**, Mota, R., Nogal, A. (2009). *Cap. 20: Uma Avaliação dos Serviços dos Ecossistemas em Portugal*, in Pereira, H., Domingos, T. (eds.) (2009), *Ecossistemas e Bem-Estar Humano: A Avaliação para Portugal do Millenium Ecosystem Assessment*, Escolar Editora, Lisboa, pp. 687-716.
- 4 papers published in international journals with peer review:
    - **Teixeira, R.**, Domingos, T., Costa, A.P.S.V., Oliveira, R., Farropas, L., Calouro, F., Barradas, A.M., Carneiro, J.P.B.G. (2008). The dynamics of soil organic matter accumulation in Portuguese grasslands soils. *Options méditerranéennes – Sustainable Mediterranean Grasslands and Their Multi-Functions*, A-79: 41-44.
    - **Teixeira, R.**, Domingos, T., Canaveira, P., Avelar, T., Basch, G., Belo, C.C., Calouro, F., Crespo, D., Ferreira, V.G., Martins, C. (2008). Carbon sequestration in biodiverse sown grasslands. *Options méditerranéennes – Sustainable Mediterranean Grasslands and Their Multi-Functions*, A-79: 123-126.

- Valada, T., **Teixeira, R.**, Domingos, T. (2008). Environmental and energetic assessment of sown irrigated pastures vs. maize. *Options méditerranéennes – Sustainable Mediterranean Grasslands and Their Multi-Functions*, A-79: 131-134.
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- 3 papers published in Portuguese journals with peer review:
  - **Teixeira, R.**, Domingos, T., Costa, A.P.S.V., Oliveira, R., Farropas, L., Calouro, F., Barradas, A.M., Carneiro, J.P.B.G. (2010). Dinâmica de Acumulação de Matéria Orgânica em Solos de Pastagens. *Revista da Sociedade Portuguesa de Pastagens e Forragens* (in press).
  - **Teixeira, R.**, Domingos, T., Canaveira, P., Avelar, T., Basch, G., Belo, C.C., Calouro, F., Crespo, D., Ferreira, V.G., Martins, C. (2010). Balanço de Carbono em Pastagens Semeadas Biodiversas. *Revista da Sociedade Portuguesa de Pastagens e Forragens* (in press).
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- 3 papers published in Portuguese journals without peer review:
  - Domingos, T., Marta-Pedroso, C., **Teixeira, R.** (2007). Projecto Extensity – O desafio da sustentabilidade. Revista Mais Ambiente, Nº4, pp.18-21.
  - Domingos, T., **Teixeira, R.** (2008). O Papel da Biodiversidade no Sequestro de Carbono em Pastagens. Revista Im))pactus, Edição n.º 11 - Os desafios da Biodiversidade e dos potenciais serviços ecológicos para as empresas e sector financeiro, pp. 17.
  - Domingos, T., **Teixeira, R.** (2008). The Role of Biodiversity in Pastures' Carbon Sequestration. Im))pactus Magazine, Issue nº 11: The Challenges of Biodiversity and the potential ecological services for companies and financial sector, pp. 17.
- 6 papers published in conference proceedings with peer review:
  - Fiúza, C., T. Domingos, **R. Teixeira** (2006). Assessing the direct and environmental costs of an activity: price painting with DALY. In Proceedings of the II Conferência da AERNA – Associação Hispano-Portuguesa de Economia dos Recursos Naturais e Ambiente, 2-3 June, Lisbon.
  - **Teixeira, R.**, T. Domingos (2006). Computable general equilibrium models and the environment: Framework and application to agricultural policies. In Proceedings of the II Conferência da AERNA –

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- **Teixeira, R**., T. Domingos, A. Simões, O. Rodrigues (2007). Local vs. global grain maize production: where should you get your maize from? In Proceedings of the 7th International Conference of the European Society for Ecological Economics, 5-8 June, Leipzig.
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- **Teixeira, R.**, Domingos, T., Fernandes, S.C., Paes, P., Carvalho, A.C. (2010). Promoting innovative solutions for soil carbon sequestration: The case of sown biodiverse pastures in Portugal. I Proceedings of the Gira 2010 Conference – Corporate Governance, Innovation, Social and Environmental Responsibility, 9-10 September, Lisbon.

Furthermore, the study of feeds and feed ingredients has been used in a thesis in Husbandry at the Évora University. The author of the thesis was Maria Maurícia Caeiro Rosado, and the dissertation was entitled "Contributo para a Integração da Componente Ambiental na Avaliação Económica de Sistemas de Produção Agro-Pecuários" ("Contibution Towards the Integration of an Environmental Component in the Economic Evaluation of Animal Husbandry Systems", in Portuguese).

Work done in this thesis was recognized three times:

- By the Organizing Committee of the XXIX Spring Meeting of the Portuguese Society of Pastures and Forages, which granted the "Progresso dos Pastos" ("Progress of Pastures") award, for a paster consisting on the analysis of SOM dynamics, similar to an earlier version of Chapter 1.
- By the Portuguese Order of the Engineers – Southern Region, which granted me an honorable mention, attributed in the contest for the Young Engineer Innovation Award 2007, for the work entitled "The Contribution of Sown Biodiverse Pastures in the Fight Against Climate Change", consisting of an earlier version of Chapters 1 and 3.
- By the Portuguese Association of Environmental Engineering, which awarded the PCF Project with the Gold Climate Network Award 2010.

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References have been formatted according to guidelines by the scientific publication *Agriculture, Ecosystems and Environment* [57]. This publication is widely referred in the present thesis.

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Afterword

So was Mr. Crespo right in his 2006 statement that he had been responsible for the sequestration of enough carbon to compensate the emissions of his family for generations to come?

According to the United Nations Statistics Division[58], average per capita emissions in Portugal from 1990 to 2006 were 5.55 t $CO_2e.yr^{-1}$. Mr. Crespo would only need to maintain slightly more than one hectare of SBPPRL per year to compensate the emissions of each member of his family. Just in his farm (Herdade dos Esquerdos, Monforte, Portalegre), he has 360 ha of them, some of which are more than 30 years old. More than enough area for generations to come.

But of course, as the person who developed the system of SBPPRL, we may give him credit for more than the area of SBPPRL that he owns.

Let us very roughly assume that these are the emissions of a Portuguese citizen during any given year (past, present or future), and are constant throughout his lifetime. Let us also assume an average life expectancy of 75 years. We conclude that the average lifetime emissions of a Portuguese citizen are 412.5 t CO_2e.

In 2006 alone, and considering the estimates in PNAC of 70 000 ha and the average factor of 5 t $CO_2e.ha^{-1}.yr^{-1}$, SBPPRL sequestered 350 000 t CO_2e.

Therefore, and if we give credit to Mr. Crespo for the existence of this system, only due to the pastures sown in 2006 in Portugal, enough carbon was sequestered to compensate the lifetime emissions of nearly one thousand Portuguese citizens.

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[58] Available at the official website for the United Nations Millenium Development Goals in: http://mdgs.un.org/unsd/mdg/SeriesDetail.aspx?srid=751&crid=.

Appendix I – Alternative estimations of the SOM model

Finding different approaches

As we have mentioned before in Chapter 1, we used a simple statistical model to try to identify the SOM dynamics. This model was calibrated using available data. The model states that the mass percent balance of SOM is the difference between input and mineralization:

$$\frac{dSOM_{i,t}}{dt} = K_i - \alpha_i SOM_{i,t-1}, \tag{1}$$

where $SOM_{i,t}$ is the SOM concentration (%) in grassland type $i = \{SBPPRL, FNG, NG\}$ at time t, K_i is the SOM input in each parcel and period, and α_i is the organic matter mineralization rate.

There are many ways to estimate the parameters in Equation (1). We have shown several in Section 2.3.3. In the present Appendix, we show several more, namely:

- Linearization of the function vs. obtaining the analytical solution;
- Making the SOM input term depend on site-specific conditions, using precipitation and texture as a proxy (instead of the initial SOM concentration);
- Using results of soil samples of a different project for 2007 and 2008.

Linearized model

Base data used for model calibration were collected between 2001 and 2005. This period comprises the first five years after the beginning of the trials. Therefore, even if the dynamic pattern is a saturating exponential, our results may be indistinguishable from a pure linear trend. Therefore, we tried estimating a model consisting of constant increases. Such is to say that we considered:

$$\frac{d^2SOM_{i,t}}{dt^2} = 0, \tag{2}$$

which is equivalent to:

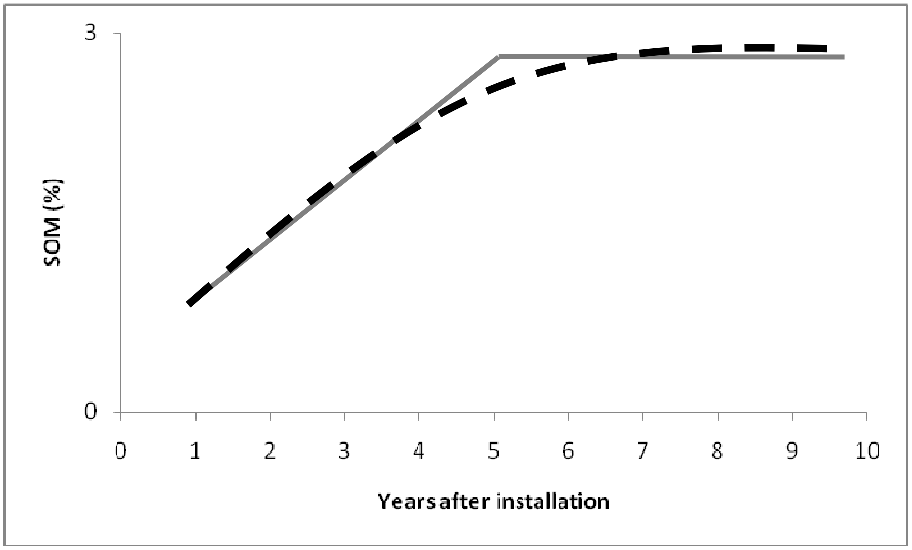
$$\frac{d}{dt}\left(K - \alpha SOM_t\right) = -\alpha \frac{dSOM_t}{dt} = 0. \tag{3}$$

The equality in Equation (3) is satisfied if:

$$\alpha = 0 \wedge \frac{dSOM_t}{dt} = 0. \tag{4}$$

This depicts a linear approximation of the saturating exponential. In the beginning, the mineralization rate is zero, and therefore the change in SOM over time is equal to *K*. From a certain point on, the second condition is true, and we reach the maximum. This approximation is shown in Figure 40.

Figure 40 – Difference between the analytical model and the linear approximation.



In the first year, the best approximation is obtained by considering $\alpha = 0$. From then on, we reach saturation, and the best fit is obtained from $\frac{dSOM_t}{dt} = 0$.

Therefore, for the first years, the model we estimate is

$$\frac{\Delta SOM_{i,t}}{\Delta t} = K_i \ . \qquad (5)$$

Since $\Delta t = 1$ (yearly data), we obtain the linear model

$$SOM_{i,t} - SOM_{i,t-1} = K_i \ . \qquad (6)$$

The specific input is equal to the average increase per year. Note that the results corresponding to this model are obtained directly by averaging the soil analyses data. Results obtained in this way only respect to the first years, and so cannot be used to extrapolate future gains in SOM.

Then, after the first years, when saturation is obtained, the linearization of the function implies that:

$$\frac{dSOM_t}{dt} = 0 \Rightarrow SOM_{i,t-1} = \frac{K_i}{\alpha_i} . \qquad (7)$$

This is the (constant) value of *SOM* for all years when saturation is reached.

Specification of SOM input using other variables

In Section 2.3.3 we used the initial SOM concentration as a proxy for soil natural conditions. Other possibility is that the best proxy for site-specific natural conditions is not the initial SOM concentration, but meteorological and soil data. The first variable we tested is accumulated yearly precipitation (P_t), and the second term is the percentage of sand (S_t), as an indication of soil texture.

The basic Equations are now Equation (5) in the linear model, and Equation (6) in the analytic model:

$$SOM_{i,t} = K_i^{'} + \left(1-\alpha_i\right) SOM_{i,t-1} + pP_t + sS_t\,, \tag{5}$$

$$SOM_{i,t} = \frac{K_i^{'}}{\alpha_i}\left(1-e^{-\alpha_i}\right) + e^{-\alpha_i} SOM_{i,t-1} + \frac{p}{\alpha_i}\left(1-e^{-\alpha_i}\right) P_t + \frac{s}{\alpha_i}\left(1-e^{-\alpha_i}\right) S_t\,. \tag{6}$$

Enlarging the data pool

Project Biopast collected samples in the plots of Project Agro 87 in the years of 2007 ans 2008. Even though some plots were lost and no data exists for the year 2006, the results from Project Biopast may also be included in our analysis.

Procedure for calculations

Besides the missing values for 2002, we now also faced missing values in 2006. In the main text of the present thesis we had filled-in missing data using geometric averages. In this appendix, we used an alternative method, namely logarithmic regression.

In this second approach, we adjusted a logarithmic curve to each location and treatment in MS Excel, since we know, due to the analytical solution of the model, that SOM dynamics follows a saturating path. We attributed to the missing values the result of the logarithmic estimation for the respective year.

Since our regression models compare pairs of points (SOM_t, SOM_{t-1}), by using these methods instead of just using measured values we end up doubling the number of observations in the regression.

Then, using both filled-in and unfilled data tables, we followed the same procedure for calculations as shown before in Section 2.3.3, only this time for a different data set (two more years of sampling) and for the new approaches to the model.

Results of the calibration of the new SOM models

Alternative filling-in of missing values

Results for the filling in of missing values using a logarithmic regression are shown in Table 63. The R^2 of the adjustment is generally high (above 0.70). The main exceptions are two cases in which the fit for NG in Farms #3 and #5. In those cases, since increases are very low (SOM is stable), the logarithmic curve is not well adjusted.

Table 63 – SOM concentration in each type of pasture for experimental sites (0-10 cm) – missing data filled in using a logarithmic regression.

Farm No.	Grassland system	SOM (%)								Logarithmic regression	R^2
		2001	2002	2003	2004	2005	2006	2007	2008		
1	SBPPRL	1.55	2.51	3.05	3.60	3.80	3.32	3.40	2.57	y = 0.7387ln(x) + 1.9953	0.4495
1	FNG	1.30	2.15	2.60	3.40	3.00	3.16	2.90	3.36	y = 0.9213ln(x) + 1.5132	0.8029
2	SBPPRL	1.75	2.54	2.65	2.70	5.40	3.58	3.40	3.10	y = 0.9486ln(x) + 1.883	0.3391
2	FNG	1.95	2.82	3.00	4.50	3.50	-	-	-	y = 1.251ln(x) + 1.957	0.7082
2	NG	1.95	2.75	2.70	4.00	4.00	4.12	5.10	3.70	y = 1.2456ln(x) + 1.8893	0.7217
3	SBPPRL	0.33	0.73	1.20	1.63	1.60	1.70	1.60	1.98	y = 0.769ln(x) + 0.3269	0.8663
3	FNG	1.08	1.25	1.10	1.40	2.00	1.51	1.50	1.34	y = 0.2388ln(x) + 1.0801	0.0834
3	NG	0.28	0.74	1.10	1.20	1.15	1.47	1.60	1.74	y = 0.6648ln(x) + 0.2784	0.8387
4	SBPPRL	3.40	3.08	5.10	4.60	5.60	5.20	4.60	5.80	y = 1.1272ln(x) + 3.1786	0.6352
4	FNG	3.80	4.51	4.70	5.40	5.60	-	-	-	y = 1.114ln(x) + 3.7347	0.9542
4	NG	3.80	4.54	4.70	5.60	5.42	5.59	6.40	5.10	y = 0.9604ln(x) + 3.8696	0.6726
5	SBPPRL	0.65	0.94	1.00	1.28	1.50	-	-	-	y = 0.4914ln(x) + 0.6033	0.9157
5	FNG	0.55	0.87	1.10	1.15	1.25	-	-	-	y = 0.4378ln(x) + 0.5644	0.9865
5	NG	0.55	0.60	0.62	0.75	0.55	-	-	-	y = 0.051ln(x) + 0.5657	0.1483
6	SBPPRL	1.82	2.11	2.40	2.18	2.70	2.37	2.20	2.29	y = 0.2324ln(x) + 1.9491	0.3643
6	FNG	1.75	2.30	2.90	2.70	2.70	-	-	-	y = 0.6462ln(x) + 1.8511	0.7957
6	NG	1.75	2.33	3.10	2.40	2.98	3.11	3.20	3.35	y = 0.7086ln(x) + 1.8374	0.7676
7	SBPPRL	0.55	0.83	1.14	1.60	-	-	-	-	y = 0.7124ln(x) + 0.464	0.9059
7	NG	1.10	1.20	1.20	1.33	-	-	-	-	y = 0.1442ln(x) + 1.0929	0.8456
8	SBPPRL	0.80	1.40	1.54	2.08	-	-	-	-	y = 0.8501ln(x) + 0.7796	0.9441
8	NG	0.84	1.06	1.10	1.45	-	-	-	-	y = 0.3853ln(x) + 0.8064	0.8424

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; in the logarithmic regression, y is SOM (%) and x is year.

Table 64 shows results for the model calibration with the original data set (2001-2005). The unfilled data and geometric average filling-in columns are equal to those already shown in Section 2.3.3, and are here just for comparison with the new columns of logarithmic filling-in. We can see that results of both fillin-in methods do not differ, both in absolute terms and relatively to the results using unfilled data.

Table 64 – Results of the estimation of models (logarithmic filling-in).

Model	Grassland system	Using SOM_0?	Unfilled data						Filled data											
									Logarithmic filling-in						Filling-in using geometric averages					
			R2	K			Alpha	a	R2	K			Alpha	a	R2	K			Alpha	a
				SBPPRL	FNG	NG				SBPPRL	FNG	NG				SBPPRL	FNG	NG		
Pooled	All	No	0.952	0.422	0.171	0.136	-0.017		0.963	0.374	0.252	0.176	-0.027		0.966	0.370	0.248	0.165	-0.034	
		Yes	0.956	0.500	0.303	0.128	0.403	0.630	0.969	0.401	0.276	0.087	0.353	0.556	0.969	0.415	0.289	0.109	0.267	0.430
Specific	SBPPRL	No	0.760	0.413			-0.020		0.781	0.353			-0.036		0.794	0.353			-0.042	
		Yes	0.731	0.531			0.237	0.348	0.784	0.364			0.205	0.358	0.794	0.379			0.151	0.276
	FNG	No	0.810		0.428		0.071		0.870		0.454		0.051		0.886		0.442		0.043	
		Yes	0.841		1.083		1.105	1.434	0.907		0.500		0.538	0.707	0.912		0.508		0.443	0.566
	NG	No	0.899			-0.034	-0.105		0.924			0.032	-0.099		0.935			0.011	-0.113	
		Yes	0.920			-0.282	0.512	1.073	0.939			-0.067	0.328	0.625	0.943			-0.048	0.190	0.432

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

Using the linear model to forecast average SOM increases

According to the linear approximation and the analytical model we chose, we obtain the set of parameters presented in Table 65. The underlying assumption of the linear approximation model is that the SOM increase in each year is equal to K in the first years (unknown, depending on the dynamic pattern for each grassland system), and it is zero after that (saturation).

Table 65 – Statistics for analytical and linear approximation models' parameters for each grassland system.

Time	Grassland system	Parameters in analytical and linear models					
		Analytical solution			Linear approximation		
		K	a	α	K	a	α
First years	SBPPRL	0.415	0.430	0.267	0.28	0	0
	FNG	0.289			0.16		
	NG	0.109			0.03		
Later years	SBPPRL	0.415			$\alpha \cdot SOM_t$		$\frac{K_i}{SOM_t}$
	FNG	0.289					
	NG	0.109					

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

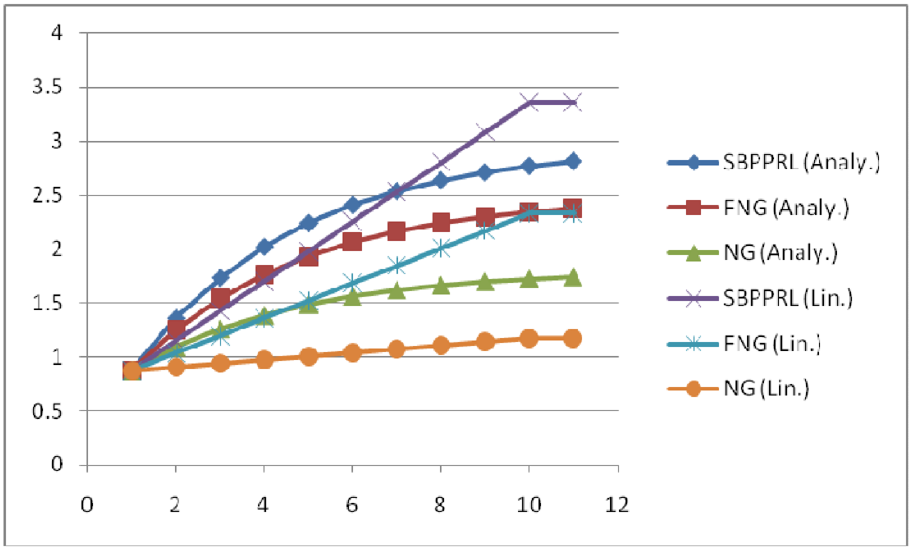
Using dynamic parameters in Table 65, we determined the average SOM increase in 10 years from each grassland type. We also assumed a starting hypothetical SOM concentration of 0.87%. Note that for the case of the analytic model we chose one set from those already obtained in Section 2.3.3, merely for comparison purposes. Results are shown in Table 66 and Figure 43, which depicts the increases in the three grassland studies in the first 10 years. In the case of the linear approximation, we assumed that pasture soils increase their SOM concentration for 10 years, before saturating.

Table 66 – Estimated SOM concentration per year in each model, starting from 0.87% SOM.

Year	Estimated SOM concentration (%)					
	Analytical model			Linear approximation		
	SBPPRL	FNG	NG	SBPPRL	FNG	NG
1	0.87	0.87	0.87	0.87	0.87	0.87
2	1.36	1.25	1.09	1.15	1.03	0.90
3	1.73	1.54	1.26	1.42	1.20	0.94
4	2.02	1.76	1.39	1.70	1.36	0.97
5	2.24	1.93	1.48	1.97	1.52	1.00
6	2.41	2.06	1.56	2.25	1.68	1.04
7	2.54	2.16	1.62	2.53	1.85	1.07
8	2.63	2.24	1.66	2.80	2.01	1.11
9	2.71	2.29	1.70	3.08	2.17	1.14
10	2.77	2.34	1.72	3.35	2.33	1.17
11	2.81	2.37	1.74	3.35	2.33	1.17
Average increase (percent points)	**0.19**	**0.15**	**0.09**	**0.25**	**0.15**	**0.03**

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter; in the linear approximation, we assumed that pasture soils increase their SOM concentration for 10 years, before saturating.

Figure 41 – Simulated SOM concentration in each year, as estimated by a pooled data model, using data filled-in using geometric averages, starting from 0.87%.



NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

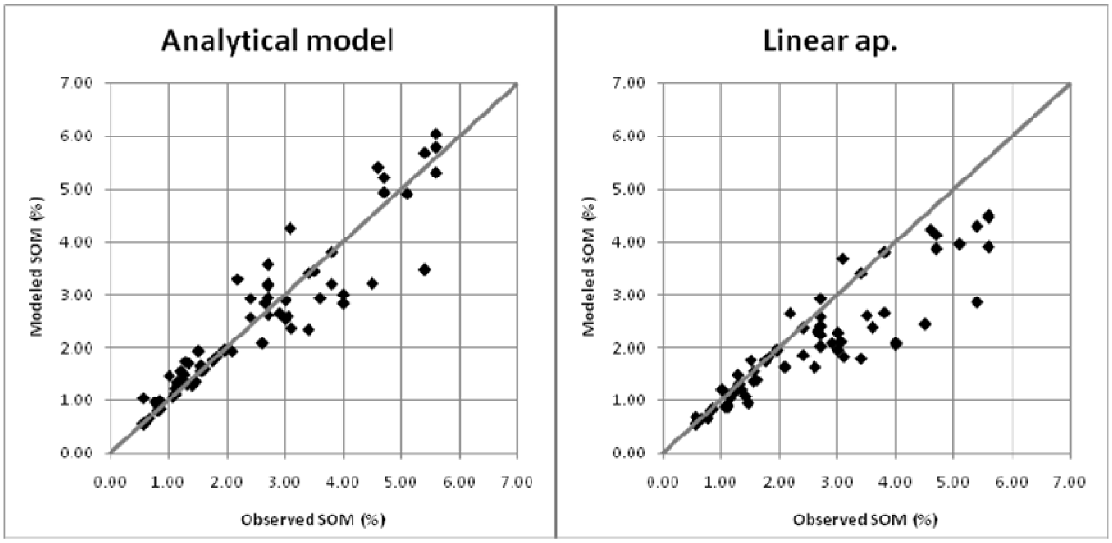
From Table 66 and Figure 43, we can see that the use of a simple average would imply a higher SOM concentration for SBPPRL, and a lower SOM concentration for both natural grasslands. For example, SBPPRL increase their SOM concentration, on average, by 0.19 percent points per year using the model and 0.25 percent points using the linear approximation.

Ultimately, a model is useful if it allows us to accurately depict data. In order to verify the adjustment provided by the model, we applied it to each farm. We considered the models that use the average parameters indicated in Table 65.

Results are shown in Figure 42, which plots all observed and simulated results. The closer that the points are to the 45º line, the better the fit. We can see that, for lower SOM concentrations, the linear model is a good approximation, but it systematically underestimates higher SOM concentration. The analytical model estimation is a good fit for all SOM concentrations (notwithstanding a slight bias to overestimate results for low SOM concentrations). However, for higher SOM concentrations, the differences seem unbiased, but variance is higher.

Our conclusion is, then, that the linear model does not provide a good fit to the data.

Figure 42 – Observed and simulated SOM concentration for all farms and grassland systems, using an analytical model (on the left) and the linear approximation (on the right).



SOM – Soil Organic Matter.

Testing for precipitation and percentage of sand (2001-2008)

In this section, we tested precipitation and percentage of sand as explanatory variables of SOM increases. In this instance, we tried to use the linear model again, this time with the complete data set of all sampling years (from 2001 to 2008). Results are shown in Table 67 for unfilled data, Table 68 for data filled with a logarithmic regression, and Table 69 for data filled-in using geometric averages.

Results show that the statistical fit of the models to base data, measured with the adjusted-R^2, is very high for the estimation of the analytical expressions, but very low for linearized first-differences. This is further evidence that the linear model is not a valid fit to the data.

We can also see that the introduction of the initial SOM concentration as an independent variable in the model increases the level of the parameters K and α. It also increases the R^2 of the estimation. However, the use of precipitation and percentage of sand as independent variables is never statistically significant.

Table 67 – Results of the estimation of models using unfilled data.

Model	Grassland system	Variables	Unfilled data															
			Analytical solution								Linearized differences							
			R^2	*K*			α	*a*	*p*	*s*	R^2	*K*			α	*a*	*p*	*s*
				SBPPRL	FNG	NG						SBPPRL	FNG	NG				
Pooled	All	Only ct. and auto-regress	0.951	0.588	0.431	0.170	0.077				0.118	0.567	0.415	0.164	0.075			
		Ct., auto-regress and SOM(0)	0.960	0.670	0.719	0.183	0.697	0.955			0.286	0.483	0.518	0.132	0.502	0.688		
		Ct., auto-regress and prec+sand	0.962	1.510	1.405	1.114	0.116		0.000	-0.011	0.096	1.425	1.325	1.050	0.110		0.000	-0.010
Specific	SBPPRL	Only ct. and auto-regress	0.755	0.379			-0.006				-0.042	0.382			0.005			
		Ct., auto-regress and SOM(0)	0.747	0.501			0.444	0.649			0.023	0.406			0.359	0.524		
		Ct., auto-regress and prec+sand	0.827	0.699			0.030		0.000	-0.005	-0.213	0.681			0.030		0.000	-0.005
	FNG	Only ct. and auto-regress	0.822		0.339		0.045				-0.072		0.331		0.044			
		Ct., auto-regress and SOM(0)	0.826		1.073		0.942	1.199			0.175		0.695		0.610	0.776		
		Ct., auto-regress and prec+sand	0.933		0.659		-0.020		-0.001	-0.002	-0.116		0.666		0.020		-0.001	-0.002
	NG	Only ct. and auto-regress	0.851			0.399	0.174				0.127			0.366	0.160			
		Ct., auto-regress and SOM(0)	0.893			0.090	1.010	1.496			0.432			0.057	0.636	0.942		
		Ct., auto-regress and prec+sand	0.893			3.409	0.162		-0.001	-0.036	0.160			3.148	0.149		-0.001	-0.033

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

Table 68 – Results of the estimation of models using data filled with a logarithmic regression.

Model	Grassland system	Variables	Logarithmic filling-in															
			Analytical solution								Linearized differences							
			R^2	*K*			α	*a*	*p*	*s*	R^2	*K*			α	*a*	*p*	*s*
				SBPPRL	FNG	NG						SBPPRL	FNG	NG				
Pooled	All	Only ct. and auto-regress	0.963	0.451	0.470	0.417	0.089				0.130	0.433	0.450	0.399	0.085			
		Ct., auto-regress and SOM(0)	0.972	0.533	0.477	0.449	0.529	0.683			0.334	0.415	0.370	0.349	0.411	0.530		
		Ct., auto-regress and prec+sand	0.960	0.882	0.849	0.776	0.107		0.000	-0.006	0.118	0.834	0.803	0.734	0.101		0.000	-0.006
Specific	SBPPRL	Only ct. and auto-regress	0.754	0.648			0.167				0.074	0.599			0.154			
		Ct., auto-regress and SOM(0)	0.809	0.737			0.680	0.805			0.278	0.536			0.494	0.583		
		Ct., auto-regress and prec+sand	0.730	1.469			0.255		0.000	-0.010	0.057	1.295			0.226		0.000	-0.009
	FNG	Only ct. and auto-regress	0.858		0.327		0.033				-0.029		0.322		0.032			
		Ct., auto-regress and SOM(0)	0.893		0.336		0.418	0.596			0.225		0.275		0.342	0.487		
		Ct., auto-regress and prec+sand	0.810		0.820		0.083		0.000	-0.006	-0.121		0.787		0.079		0.000	-0.006
	NG	Only ct. and auto-regress	0.906			0.331	0.057				0.005			0.321	0.055			
		Ct., auto-regress and SOM(0)	0.925			0.363	0.431	0.579			0.205			0.294	0.350	0.471		
		Ct., auto-regress and prec+sand	0.916			-0.031	0.049		0.000	0.001	-0.061			-0.034	0.047		0.000	0.001

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

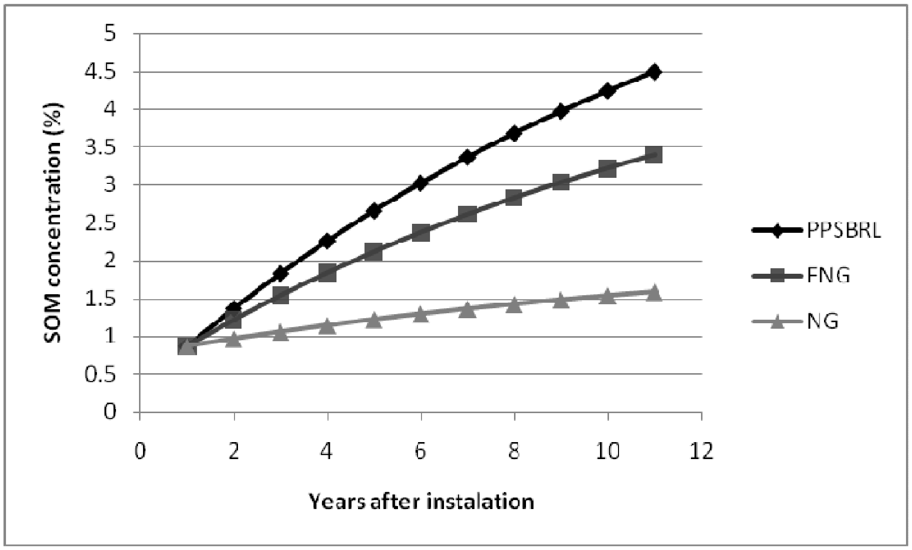
Table 69 – Results of the estimation of models using data filled with geometric averages.

Model	Grassland system	Variables	Geometric averages filling-in															
			Analytical solution								Linearized differences							
			R^2	*K*			α	*a*	*p*	*s*	R^2	*K*			α	*a*	*p*	*s*
				SBPPRL	FNG	NG						SBPPRL	FNG	NG				
Pooled	All	Only ct. and auto-regress	0.967	0.406	0.437	0.357	0.073				0.132	0.393	0.422	0.345	0.070			
		Ct., auto-regress and SOM(0)	0.974	0.407	0.418	0.258	0.440	0.608			0.322	0.330	0.339	0.209	0.356	0.492		
		Ct., auto-regress and prec+sand	0.964	0.909	0.897	0.834	0.084		0.000	-0.007	0.126	0.872	0.861	0.799	0.081		0.000	-0.007
Specific	SBPPRL	Only ct. and auto-regress	0.780	1.533			0.146				0.063	0.555			0.137			
		Ct., auto-regress and SOM(0)	0.821	0.592			0.556	0.680			0.238	0.456			0.426	0.521		
		Ct., auto-regress and prec+sand	0.762	1.350			0.232		0.000	-0.010	0.053	1.204			0.208		0.000	-0.009
	FNG	Only ct. and auto-regress	0.875		0.328		0.028				-0.029		0.324		0.028			
		Ct., auto-regress and SOM(0)	0.910		0.434		0.426	0.575			0.264		0.354		0.348	0.468		
		Ct., auto-regress and prec+sand	0.834		1.055		0.088		0.000	-0.008	-0.098		1.012		0.085		0.000	-0.007
	NG	Only ct. and auto-regress	0.924			0.262	0.037				-0.011			0.257	0.036			
		Ct., auto-regress and SOM(0)	0.938			0.121	0.355	0.557			0.181			0.102	0.299	0.469		
		Ct., auto-regress and prec+sand	0.928			-0.107	0.002		0.000	0.001	-0.111			-0.109	0.002		0.000	0.001

NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

To understand what kind of dynamic plot these values correspond to, we use the first case shown (unfilled data, only constant and auto-regressive terms, pooled data). Beginning from an arbitrary 0.87% SOM concentration plot, Figure 43 depicts the increases in the three grassland studies.

Figure 43 – Simulated SOM concentration in each year, as estimated by a model using unfilled data, for an arbitrary situation starting from 0.87%.



NG – Natural Grasslands; FNG – Fertilized Natural Grasslands; SBPPRL – Sown Biodiverse Permanent Pastures Rich in Legumes; SOM – Soil Organic Matter.

We decided not to use the complete data set with results for 2007 and 2008 in the main text of the thesis. The results were obtained in a different project, and therefore they do not guarantee the same standards of sampling. However, the previous analysis shows that results using the complete data set provide much higher increases of SOM. If they are true, then we are being conservative in the potential for carbon sequestration that we obtained in the Conclusions Chapter.

Appendix II – Estimation of CO_2e emissions from livestock

In the base scenario, which for Kyoto will be that of 1990, we assume that bovines are distributed as follows:

- Breeding cows (1 LU = 1 head·ha^{-1}) are fed in natural pastures, with a stocking rate of about 0.5 LU;
- All steers (1 LU = 0.6 head·ha^{-1}) are fed in an intensive system in stables.

We propose a scenario in which natural pastures are converted into sown pastures, and stocking rate is increased by 0.5 LU. We assume new animals will be:

- Steers transferred from stables;
- Cows in such quantity that they exist in the same number as steers. 0.5 LU refers to cows that were already in the pastures, and remaining cows are new.

Therefore, steers are now fed and finished also in an extensive system. Therefore, we must:

- Calculate emissons of cows installed in pastures and steers transferred;
- Subtract to that value the emissions avoided of steers in stables.

Therefore, we find the number of animals of each type considering that we intend to have the same number of each (x) in the end, and a total stocking rate of 1 CU. This translates into equation $x.1LU + x.0.6LU = 1LU$, and so $x = \frac{1}{1+0.6} = 0.625$.

Therefore, in the end, there are 0.625 steers·ha^{-1}, and in the beginning there were none. Initially there are 0.5 cows·ha^{-1}, and in the end 0.625 cows·ha^{-1}; variation is 0.625 - 0.5 = 0.125 cows·ha^{-1}.

Emissions from breeding cows in pastures:

We calculated the CH_4 and N_2O emissions from enteric fermentation and faeces of 0.125 cows·ha^{-1}, as shown in Table 70.

Table 70 – Emissions from breeding cows in pastures.

Gas	Emission factors		Stocking rate	Emissions	
	Enteric fermentation	Faeces		Enteric fermentation	Faeces
	kg·head^{-1}·year^{-1}	kg·head^{-1}·year^{-1}	head·ha^{-1}	kg·ha^{-1}·year^{-1}	kg·ha^{-1}·year^{-1}
CH_4	73[59]	2.156[60]	0.125	9.125	0.270
N_2O	0[61]	1.927[62]	0.125	0	0.241
CO_2e				191.625	80.331

CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

[59] Enteric fermentation CH_4 emission factor for breeding cows in grasslands, in 2004.

[60] Faeces CH_4 emission factor for breeding cows in grasslands, in 2004.

[61] IPCC (who establishes the Kyoto accounting method) does not consider N_2O emissions from enteric fermentation.

[62] Faeces N_2O emission factor for breeding cows in grasslands, in 2004.

Emissions from steers in pastures

We calculated the CH_4 and N_2O emissions from enteric fermentation and faeces of 0.625 steers·ha^{-1}, as shown in Table 71. Note that each steer only emits 0.6 of an adult.

Table 71 – Emissions from steers in pastures.

Gas	Emission factors		Stocking rate	Emissions	
	Enteric fermentation	Faeces		Enteric fermentation	Faeces
	kg·head^{-1}·year^{-1}	kg·head^{-1}·year^{-1}	head·ha^{-1}	kg·ha^{-1}·year^{-1}	kg·ha^{-1}·year^{-1}
CH_4	50.2[63]	0.679[64]	0.625	31.375	0.424
N_2O	0	0.659[65]	0.625	0	0.412
CO_2eq				658.875	136.593

CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

Emissions from steers in stables

We calculated the CH_4 and N_2O emissions from enteric fermentation and manure of 0.625 steers·ha^{-1}, as shown in Table 72. Again, note that each steer only emits 0.6 of an adult.

Table 72 - Emissions from steers in stables.

Gas	Emission factors		Stocking rate	Emissions	
	Enteric fermentation	Manure		Enteric fermentation	Manure
	kg·head^{-1}·year^{-1}	kg·head^{-1}·year^{-1}	head·ha^{-1}	kg·ha^{-1}·year^{-1}	kg·ha^{-1}·year^{-1}
CH_4	50.2	1.156[66]	0.625	31.375	0.723
N_2O	0	1.122[67]	0.625	0	0.701
CO_2eq				658.875	232.560

CO_2 – Carbon dioxide; CH_4 – methane; N_2O – nitrous oxide.

Balance of emissions

Results are summarized in Table 29. Emissions' balance is obtained by subtracting avoided stable emissions from total pasture emissions. Therefore, there is an increase in emissions of $191.625+80.331+658.875+136.593-(658.875+232.560)=175.989$ kg CO_2-eq. This means that the avoided emissions from transferring steers from an

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[63] The emission factor ranges from 60 for female steers (more than 2 years old) to 87.5 for male steers (more than 2 years old), in 2004. Emissions from animals between 1 and 2 years old range from 55.1 to 70.5. Therefore, the weighed average is 50.2.

[64] Emisson factor for steers from 1 to 2 years old and finishing beef calves (>2) in grasslands, in 2004.

[65] Emisson factor for steers in grasslands, in 2004

[66] Emisson factor for steers from 1 to 2 years old and finishing beef calves (>2) in stables, in 2004.

[67] Average steer emission factor from several manure management systems (from stables), in 2004.

intensive to an extensive system almost compensate the increased animal stocking rate that the system implies.

We then conclude that carbon balance is almost not affected by increasing animal stocking rate, as long as some animals are withdrawn from intensive breeding.

Appendix III – Environmental analysis of maize production

Introduction

International trade of agricultural commodities is common. Market globalization allowed trade rates between countries to increase, such that production may be transferred to the economically most appropriate sites in the World. Transportation thus becomes a relevant factor.

A question arises from this global fragmentation of agricultural systems: is it better, in economic and ecological terms, to transfer production to the places where it is most adequate, and then transport it to where it shall be consumed? Or, instead, is it better to reduce transportation to a minimum by producing nearby where it is mostly consumed? The answer to this question has deep implications in each country's economic structures, and can be posed in many forms and regarding all sorts of products.

To answer this question, a global life cycle approach is required, in order to comprehend not only the production and processing steps, but also the influence of both transportation steps, and finally determine the relative impacts of production and transportation. This method was strongly stimulated by the Communication on Integrated Product Policy (COM(2003) 302 final) of the European Commission. This policy intends to reduce the environmental impacts of products and services during their life cycle. The determination of the product groups with the greatest impacts was done by Project EIPRO – Environmental Impact of PROducts (Tukker *et al.*, 2006). They conclude that those groups are food and drink, housing and private transport. Minimizing the impacts of grain crops has an effect on food.

In the European Union (EU), grain crops cover over 40% of the cultivated area and are present in every member state (European Commission, http://europa.eu.int/comm/agriculture, 2005). From the beginning of the Common Agricultural Policy (CAP), their production was greatly promoted, in order to achieve self-sufficiency in Europe, as well as competitive food prices, through the intensification and specialization of agriculture (Avillez *et al.,* 2004). These policies were implemented with increasing mechanization, large irrigation and drainage systems, mass conversion of grassland to cropland, production in large monocultures, and intensive use of fertilizers and pesticides (EEA, 2003). Commodity prices were also kept high, as a guarantee of farmers' revenue.

In time, the environmental burden of both productivity and production increases became evident. Environmental impacts were considerable. Soil quality decreased, fossil fuel consumption grew, emissions to air and water caused relevant problems of acidification and eutrophication, large biodiversity loss was registered and many ecosystem values were degraded (Ventura-Lucas *et al.,* 2002). But there were also unwanted economic impacts. European prices increased in relation to world prices, and this was a large restriction to trade. Therefore, the EU, in the 1992 CAP reform, tried to end the unnecessary production surplus, thus bringing European crop prices close to those current in the rest of the World (Avillez *et al.,* 2004). Nowadays the choice for crop buying industries ranges from using local raw materials to importing from remote places where production is cheaper.

As a result, incentives to production for several crops began to decrease. One example of such was maize (*Zea mays* L.), which is an especially important food crop. In the EU, about 75% of all grain maize is used for animal feed, depending on price fluctuations, which in turn depends on the demand for human consumption. Virtually all commercial feeds for all types of animals contain maize or industrially processed maize by-products (INE, http://www.ine.pt, 2007). Maize is poor in proteins, but is extremely rich in starch. (EC, 2000). Therefore, it has high energetic content, and some farmers also feed their animals exclusively on grain maize that may be produced in the farm or bought elsewhere.

Maize is targeted by several agri-environmental measures in Portugal, namely integrated production and protection, and soil improvement and erosion prevention (direct sowing and minimum mobilization), and by the Nitrate Directive. This happens because the main environmental impacts of maize production are nitrate leaching and soil loss, especially when the lack of cover crops leaves the soil unprotected during winter. Rain may then damage the top layers of soil, and carry nutrients to lower layers of earth, and from there contaminate underground water resources and ultimately rivers. If irrigation is excessive, nitrate leaching also happens during the maize growing phase as a side effect. Fertilizers, as well as pesticides, are also responsible for emissions leading to acidification and for changes in the biological structure of the soil (EC, 2000).

Today's energy policy has turned its attention to maize as a possibility for biofuel production. This started whole new market possibilities for maize trade. Even though we will consider the optimization from the viewpoint of an animal farmer, we could as well consider it from the standpoint of an investor in a bioethanol factory.

In this paper we study the impact of grain maize production using a Life Cycle Assessment (LCA) tool, SimaPro 6.0. We characterize and select from several options regarding maize production, in terms of production zone, techniques and inputs. To assess the effect of transportation in the global impact of consumption, we consider a case study located in Beira Interior, which is a farm called Quinta da França. We chose this particular location since there is plenty of information available on how maize is produced there.

In the next section, we describe the method and data used. We also refer the specific changes to LCA considered. We then present the results obtained and analyse their uncertainty.

Method

LCA Tool

SimaPro 6.0 was developed by the National Reuse of Waste Research Programme and Pré Consultants of the Netherlands, and is widely used in assessing environmental performance. It consists of a data base of inputs and outputs from several processes and production of materials.

Therefore, the assessment of environmental impacts consists in the sum of impacts from each step of its life cycle. Impacts are then added by environmental themes. The total impact in each theme is then normalized and aggregated into a single impact indicator, usually using one of two methods: "Ecoindicator 95" and "Ecoindicator 99". Both methods aggregate impacts into a subjective and abstract unit called "Ecoindicator

Point", or Pt. Even though conclusions drawn are often similar (Luo 2001), it is important to use both, as they present different themes and a different conception.

"Ecoindicator 95" classifies, characterizes and normalizes the environmental impacts based on their contribution to several themes (Luo, 2001). The environmental aspects related to a given product are first aggregated into a number of effects caused, and those are then characterized according to the degree of damage inflicted on the environment; finally, these results are normalized into a single score, based on subjective evaluation (Goedkoop, 1998).

"Ecoindicator 99" is an update and extension of Ecoindicator 95, which emphasises its damage-oriented methodology by considering three areas of environmental damage: human health (measured in DALY – Disability Adjusted Life Years), ecosystem quality (expressed as PAF – Potentially Affected Fraction and PDF - Potentially Disappeared Fraction) and resource depletion (expressed as $MJ.kg^{-1}$) (Luo, 2001; Goedkoop and Spriensma, 2000).

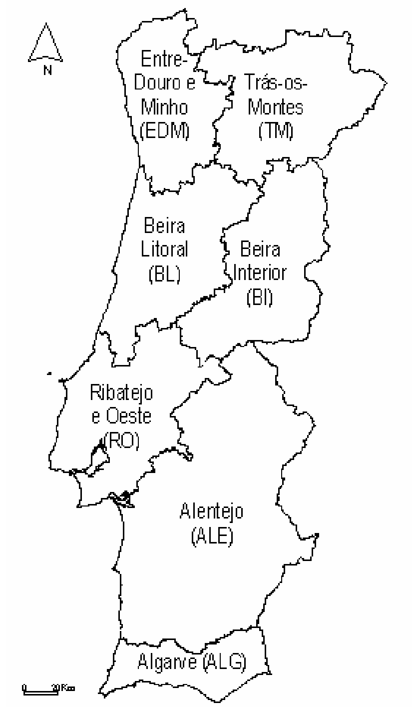
The main advantages of using LCA software such as SimaPro are the speed of assessment and the fact that its data base is very wide-ranging, since all existing inputs for any given activity are considered. However, many processes are country-specific, and the fact that it uses a foreign database is a strong limitation. Therefore, we decided to use SimaPro as the basis, but we incorporated national information whenever it was available or the impact resulting from the process was significant. Those methodological changes are described in the next sub-sections.

Analyzed zones and case study description

Maize production in Portugal was about 5.1×10^5 t in 2005, of which 2.3×10^4 t were exported. Portugal imports 1.2×10^6 t of grain maize, of which more than a third is from Argentina (FAOSTAT, http://faostat.fao.org, 2007). According to this data, the country's grain maize self-sufficiency level is about 30%.

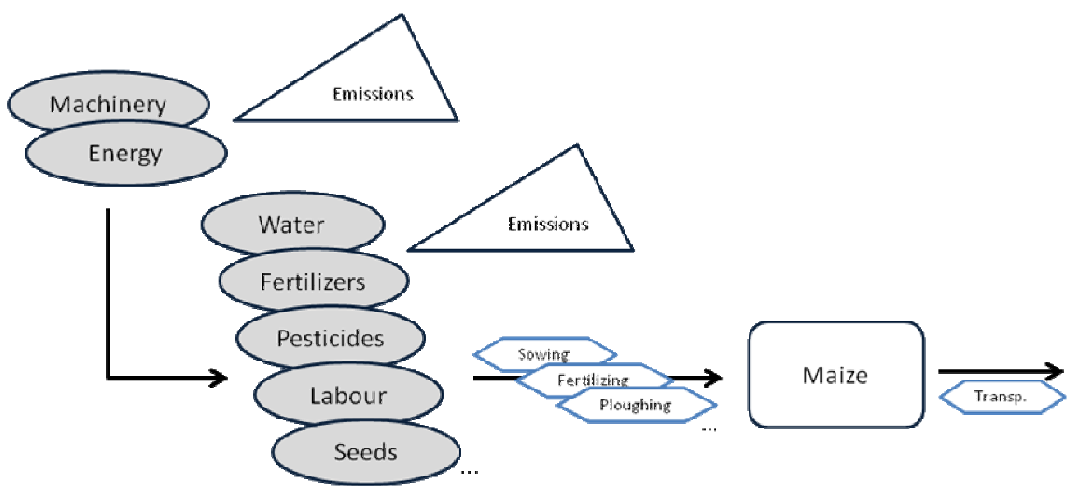
In Portugal, Entre-Douro e Minho (EDM), Beira Litoral (BL) and Ribatejo e Oeste (RO) (locations shown in Figure 44) are the zones with the largest area of cultivated grain maize. The zones with the highest productivity, in $t.ha^{-1}$, and/or total production, are Beira Interior (BI), BL and RO (INE, http://www.ine.pt, 2007).

Figure 44 – Agricultural regions in Portugal



We chose the most productive zones and characterized the production systems using the technical coefficients from crop fact sheets in GPP (2001). Then, every product entry (fertilizers, chemicals, …) and every action (sowing, ploughing, …) considered in the fact sheets were simulated using SimaPro 6.0. The system's frontiers are shown in Figure 45.

Figure 45 – System studied for maize production



We included a case study – Quinta da França (QF), in BI. We studied whether it would be an environmentally better option for QF to import maize (i.e. buy outside the farm) or to produce it. We also considered that Quinta da França could choose to produce maize with conventional tillage or using no-tillage techniques. Information regarding maize production in Quinta da França was collected at the farm.

Maize produced in Portugal is transported by road to QF. From BI itself, we assumed a distance of 50 km; from BL, we estimated the distance as being 100 km, and from RO 250 km. We also studied the impact of transportation by train. We assumed that transportation distance by railway is the same as by road. But there is additional transportation by road from the railway station (located in Covilhã) to QF. The distance between the Covilhã station and QF is 20 km. Since the other locations are generic regions, we assumed as a simplification that the farms are close to the station.

Since most imported maize comes from Argentina, we determined the average impacts of Argentinean production. We used information from Argentina, and maize production in the Pampas region. Information was obtained in an Argentinean article collection page, at http://www.elsitioagricola.com/maiz/actual/maiz.asp. We assumed that maize was transported from this region to the closest sea port by road. We considered that all ports in the area are within an average range of 200 km (Bahía Blanca, La Plata and Buenos Aires). We also considered sea transportation of 9500 km and road transportation of 200 km from the port in Portugal to QF.

The Argentinean information already corresponds to an optimized production with no-tillage. Not only are Argentinian soils and climate better to produce maize, but we are also comparing national options with maize produced using an environmental friendly technique. Therefore, we chose a good international option, in order to guarantee a fair comparison.

It should be noticed that all distances indicated in this section are only rough approximations, since we are dealing with generic regions and not specific locations, except the final consumption place. This choice is justified because we wish to draw generic conclusions regarding the impacts of transportation.

Base information

Relevant information for the cases studied is shown in Table 73. Comparing the productivity considered by each fact sheet (second row) with the average for the corresponding zone (third row), only the result for RO is coincident. The sheets are, still, a good approximation if we consider that they are overestimating productivity in the regions that, as results will show, are already the ones with a greater environmental impact (BI and BL). Table 73 also shows that the most expensive place to produce maize is BI (at the farm gate). Maize cannot be produced anywhere in Portugal with a positive revenue (without subsidies).

Since no sheet was available for no-tillage in Quinta da França, we built it by removing tillage operations from the conventional tillage sheet. This is an approximation, but since specific machinery for conventional and no-tillage is not present in SimaPro's database. We also assumed that productivity is the same for both methods (which is an empirical observation in QF).

We considered that no-tillage has an additional effect on soil carbon sequestration, by increasing soil organic matter. This effect is not permanent, but it is still a positive environmental impact. The potential for carbon sequestration by no-tillage is estimated as 0-3.0 $tonCO_2 \cdot ha^{-1} \cdot year^{-1}$ (ECCP, 2003). Calculations based on Carvalho and Basch (1995) for Portugal point to a sequestration potential of 1.8 to 2.5 t $CO_2eq.ha^{-1}.year^{-1}$. Therefore, we used the lower bound of the interval.

Table 73 – Maize production technical coefficients, for each production site and method studied

Sheet	mil8	mil14	mil15	QF	QF-NT	ARG
Production zone	Beira Interior [1]	Ribatejo e Oeste	Beira Litoral	Beira Interior	Beira Interior	Argentina
Fact sheet productivity ($t.ha^{-1}$)	6.5	10.0	10.0	6.5	6.5	8.5
Region average productivity in 2004 ($t.ha^{-1}$)	3.2	10.2	4.3	3.2	-	-
Space occupation ($ha.t^{-1}$)	0.154	0.100	0.100	0.154	0.154	0.117
Market value ($€.kg^{-1}$)	0.14	0.14	0.14	0.13	0.13	0.04
N fertilizer ($kg.t^{-1}$)	10.6	21.9	16.5	37.8	37.8	8.1
P_2O_5 fertilizer ($kg.t^{-1}$)	8.6	10.5	7	26.6	26.6	1.1
K_2O fertilizer ($kg.t^{-1}$)	8.6	10.5	7	40.3	40.3	2.7
Irrigation method [2]	Sprinkling	Furrows, gravity	Sprinkling	Furrows, gravity	Sprinkling	Gravity
Water consumption ($m^3.ha^{-1}$) [3]	7 200	6 480	8 850	9 000	7 200	2 500
Water consumption ($m^3.t^{-1}$)	1 108	648	885	1 385	1 108	294
Number of activity months [4]	8	4	4	8	8	-
Total cost ($€.t^{-1}$)	224.8	157.8	150.5	167.3	-	98.0
Revenue [5] ($€.t^{-1}$)	140.7	140.7	140.7	130	-	42.5
Gross margin ($€.t^{-1}$)	-3.1	54.5	79.8	20.2	-	26.6
Net margin [6] ($€.t^{-1}$)	-84.2	-17.2	-9.9	-37.4	-	-

[1] Production in Beira Interior requires an additional 20 $t.ha^{-1}$ of manure
[2] For more on irrigation methods, and their environmental and energetic evaluation, see, for example, Esteves et al. (1995)
[3] Water needs for national locations were considered accoding to IDRHa (http://www.idrha.min-agricultura.pt/hidrologia/necessidades/inecini.htm, 2007). Irrigation water was obtained by multiplying water needs by a factor of 1.5 for gravity irrigation, or 1.2 for sprinkling
[4] The number of activity months is the number of months during which production operations occur. This period is longer than the plant's growth cycle
[5] Revenue does not account for subsidies
[6] Net margin refers to gross margin minus the fixed costs of machinery and land. For the case of Argentina, no information could be gathered

The work was repeated iteratively. The first runs with SimaPro were used to pinpoint the most striking facts. We found that the impact in the eutrophication theme in Quinta da França is surprisingly high, and in Argentina is lower than expected. The impact was traced back to fertilizers used, and therefore the issue was addressed with additional care.

Fertilization

The values recommended by INIAP (2006) for integrated production practices in a soil with average fertility are shown in Table 74.

Table 74 – Fertilization for integrated production practices in a soil with average fertility. Values were used as the corrected fertilization for Argentina and QF.

Plant	Product	($kg.t^{-1}$)	N	P_2O_5	K_2O
Grain maize	Grain	Fertilizer applied	22	10	11

N – Nitrogen; P_2O_5 – Phosphorus oxide; K_2O – potassium oxide.

Comparing fertilization levels shown in Table 73 for QF and Argentina with recommended fertilization, it is clear that in QF an excess of fertilizers is used, and in Argentina an extremely small part of the required nutrients are being replaced. This indicates that Argentinean soils are more productive, but they are also being drained of their nutrients by unsustainable production rates, and therefore maize production has an extremely high burden on soil production capacity.

Observing Table 73 and Table 74 it may also be noticed that RO uses slightly more fertilizers than needed, while BL uses slightly less. In BI less N-fertilizer is used due to the application of manure. Therefore, since fertilization levels in QF and Argentina are extremely different from those needed, the impact was corrected taking into account a generic fertilizer applied to Argentina in order to return to the soil the nutrients that the production withdraws. It was also considered a reduction in fertilization in QF. In our scenario, only the strictly necessary quantity of each nutrient is replaced with an inorganic fertilizer.

Since, in our case study, the optimization is done from the viewpoint of the farmer in QF, it is plausible that he collects soil samples and obtains an analysis of required fertilization. The same cannot be said about Argentina, since an importer in Portugal has no control on production methods. Still, using these values provides a fairer comparison.

Emissions

Fertilization is responsible for emissions leading to eutrophication (process through which groundwater becomes too rich in nutrients, leading to the overgrowth in algae populations, hence oxygen depletion), acidification (process through which rain's pH decreases) and climate change, as shown for the Netherlands by Pluimers *et al.* (2000). Therefore, it is one of the most damaging steps of production, and so we corrected values used by SimaPro. Nutrient leaching, run-off and emissions were considered to be as shown in Table 75, based on van der Werf *et al.* (2005).

Table 75 – Emission values and uncertainty intervals used

Process	Unit	Value used	Uncertainty interval	Probability distribution used [1]
NO^-_3 leaching	kg NO_3 .ha^{-1}	40	15 - 70	Triangle
NH_3 emitted from ammonium nitrate fertilizer	NH_3 .kg^{-1} N applied	0.02	-	-
N_2O emission due to N fertilizer use and biological N-fixation	kg N_2O.kg^{-1} N applied	0.0125	0.0025 – 0.0225	Triangle
N_2O emission due to atmospheric deposition of NH_3	kg N_2O.kg^{-1} N applied	0.01	0.002 – 0.02	Triangle
N_2O emission due to leaching and run-off of NO_3	kg N_2O.kg^{-1} NO_3	0.025	0.002 – 0.12	Triangle
PO_4 runoff to surface water	kg PO_4.kg^{-1} P applied	0.01	-	-

[1] Refers to this paper. In a triangle distribution the probability of the given value is the highest, and decreases linearly in the uncertainty interval until it reaches zero in the extreme values. For more on probability distributions used by SimaPro, see Grant (2005)

NO^-_3 leaching values vary with crop, production method, soil characteristics and content in nutrients and precipitation, and therefore a local analysis should clearly study these parameters and their relation to leaching. We did not find any data directly referring to the zones studied. Therefore, in order to assess whether the value indicated above is adequate for this paper's conditions, we used two strategies. First, we consulted several Spanish studies. Spain is especially relevant, given its comparability with Portugal in terms of soil and climate characteristics. Then, we calculated the overall nitrogen mass balance.

Regarding Spanish studies, Moreno *et al.* (1996) studied nitrate leaching under irrigation in Spain and reached values of 150 and 43 kg $NO_3^-.ha^{-1}$ leached, corresponding respectively to 500 and 170 kg $N.ha^{-1}$ applied. Villar-Mir *et al.* (2002), for an N application of 250 to 340 kg $N.ha^{-1}$, measured 60 kg $NO_3^-.ha^{-1}$ leached. Diez *et al.* (1997) studied the effect of various fertilization and irrigation choices, emphasising that with convenient irrigation it is possible to diminish leaching in more than 90%[68]. These results confirm that a value of 40 kg $NO_3^-.ha^{-1}$ is plausible as a first approach.

The nitrogen mass balance provided further justification for the use of van der Werf *et al.*'s (2005) values. In steady state, fertilization (Table 74) should be equal to the plant nutrient export plus losses in Table 75. This means that there is neither nutrient mining nor over-fertilization. For example, in QF 6.5 tons of maize are produced. We considered that the maize plant exports from the soil 15 kg of N for each ton produced (Feil *et al.*, 2005). Subtracting this value, as well as quantities in Table 75, from nitrogen fertilization in Table 74, we obtain a value of -1.6 kg $N.ha^{-1}$, about 1% of all nitrogen applied. This means that the law of conservation of mass is verified for nitrogen. For other locations, the balance is also correspondent to less than 10% of nitrogen applied, and so we conclude that values are consistent.

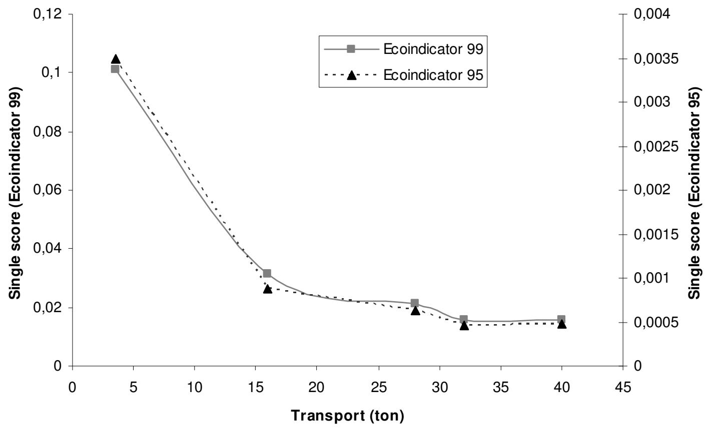
These analyses justify the use in this paper of the generic value indicated above. Still, we carried out an uncertainty analysis, in order to verify whether results would change significantly from the variation of input parameters.

The impact of transportation

This paper considered inland transportation to occur by road, the predominant form in Portugal. Still there are different options regarding the capacity of the truck. The impact of each t.km in Ecoindicators 95 and 99 varies as illustrated by Figure 46. We also considered that transportation occurs in a 16 t truck, since smaller capacities are not used, and larger ones have almost no change in impact.

[68] The study was made for several possible fertilizers, and the reduction in leaching occurs for all of them.

Figure 46 – Impact of transportation. The functional unit is 1 t.km of transport (Ecoindicators 95 and 99).



We also considered railway transport as an alternative to road vehicles. According to SimaPro's database, this would decrease the transportation impact of each t.km in 20%.

Transportation from Argentina occurs by sea. The impact of overseas transportation in a transoceanic freight ship is 4.1E-05 $Pt.t^{-1}.km^{-1}$ in Ecoindicator 95 and 1.3E-03 $Pt.t^{-1}.km^{-1}$ in Ecoindicator 99.

Results

The impact of maize production

The main results provided by SimaPro's LCA using Ecoindicators 95 and 99 are shown in Table 76. Only the most significant categories are depicted. Values for QF and Argentina are shown with corrected fertilization, since these are the more realistic. We also included a column for no-tillage production in QF (QF-NT).

Regarding only average national regions, maize grown in RO has the lowest environmental impact. It is also the best in every theme except acidification (where the best is BI). However, maize produced with no-tillage techniques in QF has a lower environmental impact than that produced in all other national regions. No-tillage has an impact about 20% lower than conventional tillage for QF. In all cases, the themes with higher single score impacts are: heavy metals, acidification and eutrophication..

Table 76 also shows that maize produced in Argentina has an impact similar to no-tillage maize in QF for Ecoindicator 95. In Ecoindicator 99 the impact is lower for Argentina. This is coherent with the method of production, since we used an Argentinean no-tillage fact sheet. A higher productivity is not the reason for this result, since 8.5 $t.ha^{-1}$ (in Argentina) is lower than in BI and RO.

More land is used in BI and QF to produce maize, which confirms the results obtained by observing direct occupation of space. RO is the zone where the least area is needed for production (high productivity). However, Argentina has the smallest land use, but not the lowest direct occupation of space, which shows that the latter may not always be a correct indicator of area needed for production.

As for water consumption, in Argentina maize requires less water than anywhere in Portugal. This is mainly due to the climatic difference between both countries. As for national production zones, the most productive ones (BL and RO) have lower consumption per unit produced. However, optimized production in QF-NT, by changing from gravity irrigation to sprinkling, may correspond to a 20% decrease in water use.

All other themes share the same relative impact order with single score results. The only significant difference is that eutrophication in QF is smaller than in RO, which is a consequence of correct fertilization.

Table 76 – Contributions in the most important environmental categories (at the farm gate).

Assessment method	Impact category	Unit	BI	BL	RO	QF	QF - NT	ARG
Ecoindicator 95	Greenhouse	kg CO_2	353	457	343	421	106	423
		% Pt	4.8	6.1	5.4	6.2	1.9	7.2
	Acidification	kg SO_2	4.01	5.26	4.16	2.99	2.69	5.13
		% Pt	25.4	32.4	30.3	20.4	22.5	40.4
	Eutrophication	kg PO_4	1.43	1.38	1.24	1.09	1.04	1.47
		% Pt	13.4	12.6	13.4	10.9	12.8	17.1
	Heavy metals	kg Pb	0.00737	0.00644	0.0056	0.0075	0.00626	0.00377
		% Pt	48.4	41.2	42.2	53.1	54.3	30.7
	Energy resources	MJ LHV	3810	4130	3400	5100	4570	2400
		% Pt	-	-	-	-	-	-
	Total	**Pt**	**1.4**	**1.4**	**1.2**	**1.3**	**1.1**	**1.1**
Ecoindicator 99	Resp. organics	DALY	0.00038	0.00042	0.0004	0.0003	0.00028	0.00034
		% Pt	30.7	31.3	32.5	23.2	22.0	35.2
	Climate change	DALY	8.3E-05	0.00011	8E-05	1E-04	3.2E-05	0.0001
		% Pt	6.7	8.3	7.0	6.6	2.5	10.8
	Acidification/ Eutrophication	PDF.m^2.yr	28.8	38.9	30.3	20.5	18.5	39.2
		% Pt	7.0	8.8	8.0	4.1	4.4	12.3
	Land use	PDF.m^2.yr	38.8	27.2	23	39.1	38	21.8
		% Pt	9.4	6.1	6.0	7.9	9.0	6.9
	Fossil fuels	MJ surplus	394	456	405	571	505	289
		% Pt	29.1	31.6	32.4	35.1	36.5	27.7
	Total	**Pt**	**32.2**	**34.5**	**29.8**	**38.8**	**32.9**	**24.8**
	Water consumption	$m^3.t^{-1}$	1 108	648	885	1 385	1 108	294
	Direct occupation of space	$ha.t^{-1}$	0.154	0.1	0.1	0.154	0.154	0.117
	Revenue	$€.t^{-1}$	-8.42	-0.99	-1.72	-	-	-

BI – Beira Interior; BL – Beira Litoral; RO – Ribatejo e Oeste; QF – Quinta da França; NT – No-Tillage; ARG – Argentina. Functional unit: 1 t of maize (Ecoindicators 95 and 99).

Operations' impact

As for the impact of specific steps in production, taking RO as an example, and for the impact of specific production operations, fertilization is by far the critical component, accounting for 71% of the total impact.

Fertilization contributes to eutrophication in almost 90% of the total theme's contributions. The impact is mainly due to nitrogen leaching caused by application of fertilizers. 80% of the impact in the theme is the fertilizer itself (application and subsequent loss), and only 20% the fertilizing operation (in terms of machinery use).

In Ecoindicator 99 the most relevant parameter is fossil fuels. The operations responsible for such impact are transportation, machinery use and fertilizer production (56% for QF).

Engström *et al.* (2007) indicate that the most important environmental themes for Swedish agriculture are eutrophication, global warming and resource use. These are also the themes generally referred in the literature. Our analysis confirms that these impacts are important, but indicates some others of interest, like acidification. The importance of heavy metals is striking. SimaPro considers this parameter to be of extreme relevance when it weights its categories to produce a single score. It allocates the impact of building the machinery used to the production in which it intervenes.

Heavy metals are not usually considered an important theme when analysing agricultural life cycles. However, agriculture is an overcapitalized industry. Unlike other types of machinery (industrial, private transportation vehicles), agricultural machinery is used for a relatively small time frame, and only in a very specific time of year. Therefore, costs and inputs of machinery building and use must always be considered, since its impact is comparable to that of maintenance and fuel consumption. Furthermore, in Portugal, recycling or reuse is not necessarily the final destination of materials, and emissions may be aggravated by lack of adequate final destination. For example, in the case of irrigation, machinery needed stands for 46% of its heavy metal emissions, while electricity consumption stands for 38%.

The impact of heavy metals may also be explained by fertilizer use. Fertilizers are currently the main sources of cadmium emissions to the soil, which is an important problem in The Netherlands, where the method was developed. The average European value is 3.8-6.8 $g.ha^{-1}$ in crop land, whereas in The Netherlands values are as high as 7.5-8.5 $g.ha^{-1}$ (Ferrão, 1998). For example, in BI (where the impact on heavy metals is the highest), we found that over 35% of the impact comes from fertilizing. Irrigation also has a very significant part (over 25%).

One simplification we used in this paper was to assume all agricultural machinery similar. In SimaPro's database there is only one choice for each type of machinery. However, depending on weight and power of the machinery, its composition, heavy metal content and energy consumption will differ. In order to confirm if this simplification, as well as all others, could have an influence on final results, we performed an uncertainty analysis.

Uncertainty analysis

Table 77 shows the results of the uncertainty analysis for single score results in Ecoindicator 95. SimaPro uses a Monte Carlo analysis, where random numbers are generated to determine the parameters from the uncertainty domain of each input used in the analysis.

Table 77 – Uncertainty analysis' results, indicated as a percentage of the number of times impact of A > impact of B (Ecoindicator 95).

A \ B	BI	BL	RO	QF	QF - NT	ARG
BI	-	55%	1%	11%	0%	0%
BL	45%	-	0%	17%	0%	0%
RO	99%	100%	-	70%	0%	20%
QF	89%	83%	30%	-	0%	10%
QF – NT	100%	100%	100%	100%	-	82%
ARG	100%	100%	80%	90%	18%	-

BI – Beira Interior; BL – Beira Litoral; RO – Ribatejo e Oeste; QF – Quinta da França; NT – No-Tillage; ARG – Argentina.

It is thus shown that QF-NT is the best option, regardless of parameter variation, except in direct comparison with Argentina 18% of times. These results provide some confidence in the conclusions taken, since they show small variation with changes in parameters. The only exception is BI and BL, since both have very similar results, and it is not clear which one is better.

The uncertainty of the results for each studied location is considerable, as shown for the case of QF in Figure 47. Furthermore, the variation is not symmetric. In fact, the average value obtained with the uncertainty analysis is not the value obtained in calculations with fixed parameters. This is a direct consequence of the fact that the final result is a non-linear function of the uncertain parameters. SimaPro uses error distribution of the lognormal form in almost every entry used, but those values are combined between them and with those introduced. The distribution used for those was triangular. The result is that the final distribution of probability is not itself linear, and therefore the average of the distribution of impacts is not the average impact.

Figure 47 – Overall results for QF, with the error bar indicated (95% confidence interval, standard deviation of 0.395 and standard error of mean of 0.0229).

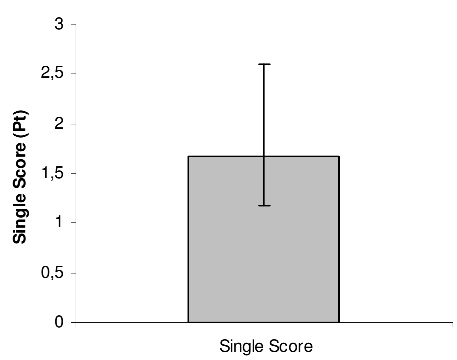
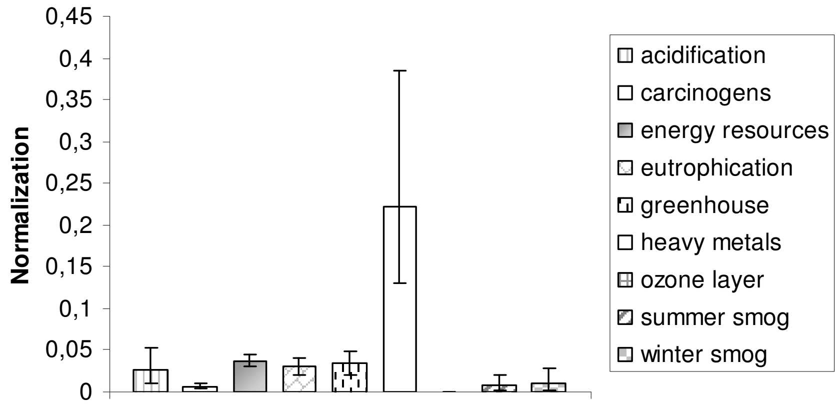


Figure 48 represents single score values and their uncertainty in each theme in Ecoindicator 95 for QF. The most significant themes are those for which we presented the results, and it may be noticed that they are also the most uncertain ones. This analysis was also the criteria for results' precision. Since uncertainty is proportionally high for all themes, and particularly for the aggregated single score as shown in Figure 47, only two significant digits were used.

Figure 48 – Single Score values with correspondent uncertainty in each theme of Ecoindicator 95, for QF maize.



It should be noticed that crop fact sheets do not have any uncertainty data. However, the variation in productivity is of crucial importance in overall impact. It depends on inputs like fertilizers and irrigation, as noted in Megyes *et al.* (2003), Kiniry *et al.* (2004) and Rátonyi *et al.* (2003). But its values may be very different even with the same amount of inputs, depending on regions and meteorological conditions of each year. These determine not only nutrient and water availability for plant uptake, but also its physiological conditions (Bert *et al.*, 2006; Park *et al.*, 2004). For example, Novák and Vidovic (2003) determined a linear relationship between maize nutrient uptake (and therefore plant growth) and the rate of transpiration.

We incorporated the variability in productivity in the results above by observing time series of average productivity values for each region (INE, http://www.ine.pt, 2007). Since the values in the fact sheets are not necessarily the average values, the intervals of variation could not be precisely defined. Still, values of 10% above the normal for a good year and 30% below for a bad one seem plausible from the data observed (maintaining all inputs constant). These were introduced in the analysis with a triangle distribution of probability.

Case study – importation options for Quinta da França

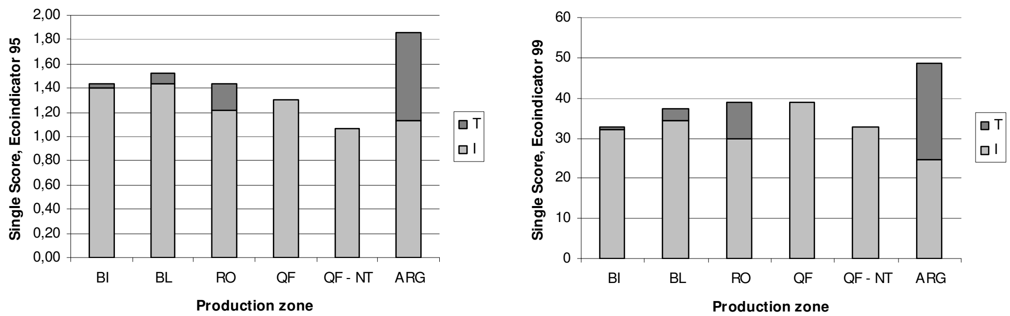
The farmer in QF may choose to produce maize or to import it from elsewhere. We want to study which is his best environmental option. Figure 49 shows that he should produce it himself.

In Ecoindicator 95, QF has the lowest environmental impact even with conventional tillage, but in Ecoindicator 99 production in QF is only the best option if no-tillage is used. Considering the uncertainty level, all national options (using conventional tillage) have very similar results. In both Ecoindicators' analyses maize from Argentina becomes the worst environmental option. Transportation from such a distance makes its net impact clearly the highest.

One explanation for the difference between this evaluation and the one provided by Ecoindicator 95 is that in the latter the impact difference to the other regions at the farm gate was relatively smaller. In Ecoindicator 99, since QF was by far the worst place to produce maize, transportation is not enough to compensate for the difference.

Therefore, the farmer's ideal choice would be to produce maize in QF, where he can control the production conditions, use the best practices (like correct fertilization and no-tillage) and careful irrigation. But even if he does not, the impact of transportation may be enough to justify production in QF.

Figure 49 – Impact of maize production in each region with corrected fertilization and considering transportation to QF by road.



Right: Ecoindicator 95. Left: Ecoindicator 99. Functional unit: 1 t of maize. T – transportation impact; I – ingredient production impact

If transportation by road is substituted by railway, then transportation impacts would decrease significantly. QF with no-tillage would remain the optimal choice, but all other zones would be a better option then conventional tillage in QF, except Argentina. The global impact of importation from Argentina would decrease by 15%, but that is still not enough to compensate the impact of sea transportation.

Conclusions

In this work we studied the impacts of maize production and transportation. Maize is a very important tradable good, since it is highly used for animal and human food, and increasingly used as a biofuel.

We used an LCA tool, SimaPro, to determine the environmental impact of grain maize production in some regions of Portugal and in Argentina. We also studied alternative production methods and the corresponding impacts. We used Quinta da França (QF) as a case study, in order to determine how transportation impacts influence the net result of a management decision to produce maize locally or to import it from elsewhere.

Results obtained show that the distance of transportation is a relevant parameter in the Life Cycle Assessment method, since it may change the results enough to alter the conclusions. For example, the best national zone for production of maize is RO, but since that zone is further distanced from QF, it is environmentally better to produce maize in QF even if production is not optimized. Railway is a valid alternative to road transportation, but the corresponding impact decrease is not enough to change global results.

Importing from a foreign country, Argentina in the case, proved at first to be a valid option, since that is a suitable place for production, even correcting the fertilization practices. But after transportation, Argentina becomes the worst environmental option. Argentina's only advantage is the fact that it requires less water consumption. Water consumption by itself may be an impact whenever it is a scarce resource. It is also a very important theme in agriculture that is not always considered in Life Cycle Assessments, even though irrigation itself may be contemplated via energy consumption.

But QF is not always the best option regardless of its management. Maize production impact is only lower if best environmental techniques are used, such as correct fertilization, correct irrigation and no-tillage. It may be argued that such is also the case in all other places. But, for a given farmer, it is easier to ensure that production is correct if crops are nearby, since the quality of imported crops can seldom be guaranteed. For example, the owner of QF may very easily collect soil samples and find advice on correct fertilization, but it would be impossible for him to individually oblige Argentinean producers to correct their own.

The most important impacts from maize production are greenhouse gas emissions, acidification, eutrophication, energy use and heavy metals. The contribution of heavy metals in Ecoindicator 95 is an unexpected result, since it is not one of the usual relevant themes in agricultural production.

We suggest as further work an economic analysis of alternatives that would widen the scope of the analysis. When optimizing technical solutions and options, like deciding between producing locally or importing from elsewhere, not only direct impacts should be considered but also alternative occupations of space. Because of price regulation, if farmers decide to intensify in one environmentally damaging zone maize production, then former import sources would have to produce less. If the occupation that would substitute maize production there brought a global positive environmental impact, then that would be the better solution. This means opportunity costs must be addressed, as noted by Jungk *et al.* (2002) and Manbiot (2004). However, the LCA method used in this paper assume that choices do not change the whole agricultural sector enough for such effect to be relevant. LCA results have a very low spatial and temporal resolution, and do not regard social and economical aspects, as noted by Owens (1997) and Udo de Haes (2004).

Still, Life Cycle Assessment software such as SimaPro is an important first step in evaluating environmental impacts. We used in this paper a method which included iteratively refined data for the most important aspects of the evaluation. We consider that this method makes SimaPro an easy tool that provides quick but reliable estimations for multiple different indicators, and is an important evaluation option for policy-making.

Appendix IV – Environmental analysis of concentrated feeds

Introduction

Nowadays, a question arises from the global fragmentation of agricultural systems: is it better, in economic and ecological terms, to transfer production to the places where it is most adequate, and then transport it to where it shall be consumed, or, instead, reduce transportation to a minimum by producing nearby where it is mostly consumed? The answer to this question is dominant in any conception of the distribution of space dedicated to agriculture, as well as relevant in the urban/rural segregation, and can be posed in many forms and in respect to all sorts of products. A good example of this is animal feed, since in its composition may be found crop products with different production systems, depending on the place of cultivation, and all those ingredients have to be processed industrially before being given to the animals. Therefore, the optimization of their life cycle has to comprehend not only the production and processing steps, but also the influence of both transportation steps, in order to determine the optimal composition and transportation.

Since the 1950's, consumption of meat products has increased steadily. It is considered that 1 kg of beef requires 7 kg of high-protein feedstuffs (Brown *et al.,* 1999), 1 kg of pork requires 4 kg of grain (CIWF, 1999) and 1 kg of poultry requires 2 kg of feed (CIWF, 1999). Therefore, a higher meat production corresponds to a higher ingredient demand. Today, 95% of the world's soybean production and a third of commercial fish catches are used for animal rather than human feed (Millstone and Lang, 2003). The area needed to produce feeds for the increasing number of animals is, then, high; 75% of all agricultural land in the United States is used to produce ingredients in livestock feed (Millstone and Lang, 2003). This contributes significantly to the impacts of crop production, aggravated by the impact of the animals themselves, namely regarding soil loss and desertification – 85% of all topsoil loss in the United States is attributable to livestock ranching (Millstone and Lang, 2003). Therefore, an increasing number of studies have tried to assess sustainability in agriculture (Lewandowski *et al.*, 1999), particularly researching the environmental impact of feeds (van der Werf *et al.,* 2005; Cederberg and Mattsson, 2000).

In Portugal, the intensification trend as also followed. The animal feed industry is the third most important in the agricultural sector, representing 10.5% of total business volume in 2002 (IACA, 2004). The total production of feeds in the same year has been estimated in almost 3.5 million tons (IACA, 2004). Its relevance derives from the fact that in both intensive (always) and extensive (during the least productive seasons) production systems of meat it is necessary to provide feeding to the animals elsewhere than the pasture. Therefore, it is of great relevance to determine the best composition possible that the feed should have. This work focuses mainly on commercial processed feeds.

Examples of studies on agricultural and meat production are Pereira *et al.* (2004a), Serra *et al.* (1996), Serrano *et al.* (2003) and Simões *et al.* (2003). The determination of the best feed composition, based on a least cost nutritional need, and how it influences the physical quality of the pellet, has already been published (Thomas *et al.,* 2000). Studies have also been made in what respects to feed utilization and its relation to seasonal distribution, as well as improvements in animal production

(Zemmelink *et al.,* 2002; Coleman and Moore, 2003). So far, the consideration of environmental aspects has only been done seldom and as a complement (Castrodeza *et al.,* 2004), and even more rarely using Life Cycle Analysis; to our knowledge, the only examples are pig feed in Bretagne (van der Werf *et al.,* 2005) and dairy cattle feed for milk production (Cederberg and Mattsson, 2000).

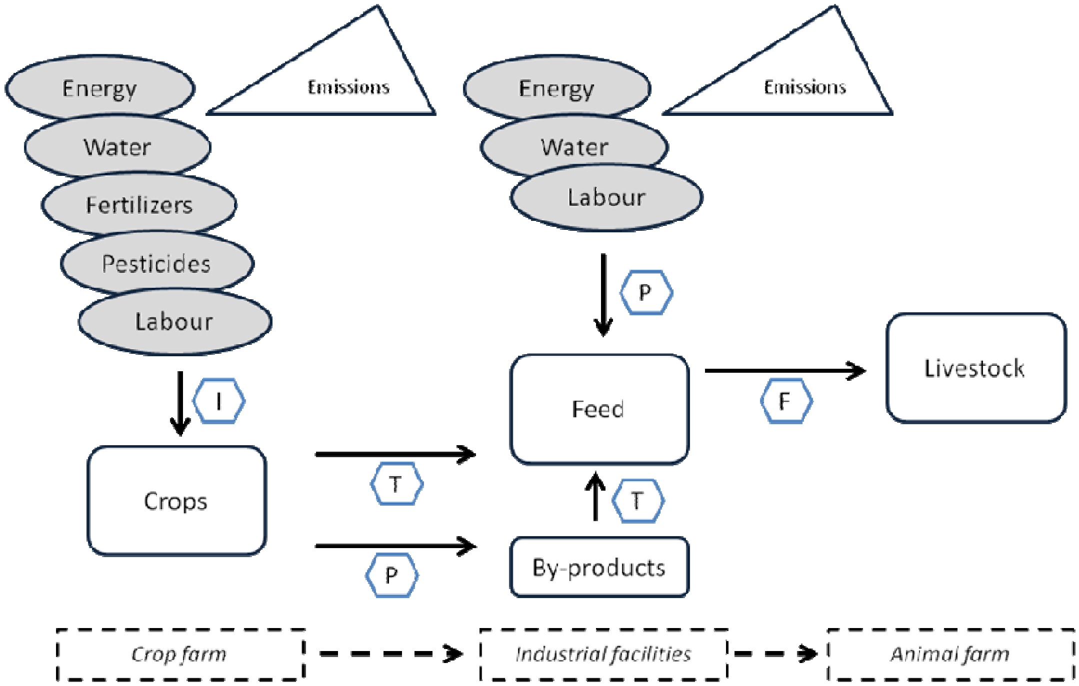
The main goal of this paper is to obtain an optimized feed in the case of beef calves fed in Portugal, regarding its environmental impact. That feed must have the best combination of ingredients (those with the least environmental impact), and the ingredients themselves must be produced in the most appropriate zones through the most appropriate production processes, when possible. That feed also has to fulfil nutritional requisites for healthy animal growth.

Method

Life cycle studied and system boundaries

The typical commercial feed is composed basically of two major groups of ingredients: cereals and oilseeds (protein sources) (IACA, 2004). In both cases, Portugal is a major importer, and so the cost of the raw materials is almost 70% of the total cost of production (IACA, 2004). The life cycle of feed, shown in Figure 25, starts with the production of its constituents, is continued by the transport (national or international) to the factory, where the feed is processed, and from where it is transported again, this time to the farm where it is given to the animals. So, it is important to characterize the very ingredients that compose the feed, because it is during that phase that much of the environmental impact of the final product occurs. That closes the feed’s life cycle, since we consider that impacts after the moment the feed is given to the animal are no longer allocated to the animal feed sector, but to the animal production sector.

Figure 50 – Feed's life cycle scheme.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Impact calculation

Therefore, the environmental impact ($\boldsymbol{I_{feed}}$) of the feed consists of the sum of the impact of each of the $\boldsymbol{n}$ ingredients ($\boldsymbol{I_i}$), the impact of transportation of those to the factory ($\boldsymbol{T_i}$), the impact of the production of the feed in the factory ($\boldsymbol{P}$), and, finally, the impact of transportation of the feed to the farm ($\boldsymbol{F}$):

$$I_{feed} = \sum_{1}^{n} I_i + \sum_{1}^{n} T_i + P + F \qquad (1)$$

The impact I_i is not only that of the ingredient production at the farm, but also, in cases where it exists, of its industrial processing prior to that of the feed. This means that there is transportation from crop farms to industrial units considered in impact I_i.

Data gathering

In this paper the ingredients used in the considered feeds were characterized, in terms of its place of origin and ways in which it is produced and quantities used. Then, we determined how transportation involved in the life cycle occurs, and also how the feed is processed. From all the data, different scenarios were considered, in order to determine the optimal configuration of the feed, using the indicators eutrophication, acidification, greenhouse gas emissions, heavy metals, energy and land use.

Alentejo is the region where more reproductive cows are fed, as shown in Table 78 (INE, http://www.ine.pt, 2005). However, only half of the calves are fed in Alentejo (considering each cow has one offspring per year), since they are usually transported elsewhere to be fed for some months before slaughtering. In this paper, we assume an additional optimization that calves are finished in Alentejo.

Table 78 – Number of animals in each region in Portugal

	Beef Calves				Cows	
	Age < 1 year		Age from 1 to 2 years		Others	
Agricultural region	**Heads x 10^4**	**%**	**Heads x 10^4**	**%**	**Heads x 10^4**	**%**
Entre-Douro e Minho (EDM)	9.4	28.4	2.2	31.5	4.1	11.9
Trás-os-Montes (TM)	2.3	7.1	0.1	1.6	2.4	7.0
Beira Litoral (BL)	4.3	12.8	1.0	14.2	1.1	3.4
Beira Interior (BI)	1.3	3.9	0.3	4.3	1.4	4.1
Ribatejo e Oeste (RO)	4.7	14.0	1.7	25.2	2.3	6.7
Alentejo (ALE)	11.0	32.9	1.5	21.7	20.0	65.8
Algarve (ALG)	0.3	0.9	0.1	1.4	0.4	1.2
Total:	**33.3**	**100**	**6.9**	**100**	**31.7**	**100**

As for particular aspects, indicator-based methods used to assess environmental impact at farm level have already been compared (van der Werf and Petit, 2001; Payraudeau and van der Werf, 2004), in order to determine their differences and scope. Virtual water flows (the water needed to produce a commodity which is then sold) in respect to crop trades has also already been analysed (Hoekstra *et al.,* 2004), as well as the best indicators related to land use to provide a good picture of long-term soil fertility and biodiversity (Mattsson *et al.,* 1999). This paper intends to use all of the above in order to produce the most correct evaluation of the life cycle of both feed and feed ingredients.

Feed composition

The European Union regulates which ingredients are safe to use in animal feeds, and that list is eventually incorporated in the Portuguese legislation. A full list of national and Communitarian applicable legislation to the animal feed sector may be found at IACA (2004). Generally, the ingredients can be grouped in the following categories:

- Plant origin products and by-products, as well as transformation products and by-products (cereal grains, oilseeds, legumes, tubers, roots…);
- Animal origin products and by-products (milk, fish, eggs…);
- Mineral origin products;
- Conservants.

The national availability of the most relevant of these ingredients for animal commercial feeds (excluding those given directly to animals) is presented in Table 79 (IACA, 2004).

Table 79 – National availability of feed ingredients

Product	Internal Production (10^3 t)	Gross import (10^3 t)	Gross Availability (10^3 t)
Plant origin products and by-products	934	2 126	3 060
Cereals	845	1 766	2 611
Broken rice	2	-	2
Dry leguminous	1	5	6
Potato	3	2	5
Plant fats and oils	-	20	20
Dehydrated green fodder	-	55	55
Manioc	-	175	175
Other plant origin products and by-products	83	103	186
Transformation products and by-products	925	784	1 709
Grinding and hulling by-products	264	12	276
Malt and beer industry by-products	93	73	166
Distillation products and by-products	32	10	42
Starch extraction industry products and by-products	-	487	487
Of which: Corn gluten feed	-	482	482
Sugar industry products and by-products	63	30	93
Of which: molasses	21	31	52
Oilseeds	401	172	573
Other animal origin products and by-products	72	-	72
Animal origin products and by-products	22	14	36

The table shows that, as a whole, cereals, corn gluten feed and oilseeds comprise 75% of total availability, which makes these components the most relevant to analyse. Corn[69] gluten feed is a by-product of the starch extraction industry obtained by corn wet milling after most starch has been removed. The process starts when shelled corn is cleaned and washed; it is then grinded and screened. The resulting bran, after being dried, is corn gluten feed (Shroeder, 2004). It is composed of a moderately high proportion of protein (20-25%), starch (20%), many digestible fibres and a small amount of oils (Blasi *et al.*, 2001). Portugal mainly imports corn gluten feed from the United States of America (IACA, 2004).

One should, however, notice that the exact ingredient used, in each of the categories shown in Table 79, depends on the species of animal, its current age and phase of development, since its nutritional needs also vary (IACA, 2004). Therefore, the composition of the feed itself is different from animal to animal, as well as the quantity sold and the price.

Nevertheless, there are some common characteristics among feeds. Maize is normally the most used cereal. In general, the quantity of cereals in feeds in 2003 is shown in Table 80 (IACA, 2003).

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[69] Since corn gluten feed is fully imported from the United States of America, the American designation of maize is used when referring to this product.

Table 80 – Main cereals' quantity used in feeds.

Product	Quantity (t)
Oats	901
Barley	150 088
Maize	989 935
Sorghum	2 292
Wheat	355 742
Triticale	29
Others	5 586

Scenarios studied

Knowing which ingredients are generally used in feeds, we chose three cases of feeds for the same type of animal and age, in this case finishing calves, to analyse:

4. A grain maize-based feed (Feed 1), which will reflect optimized choices in production zones and methods;
5. A silage maize-based feed (Feed 2), also reflecting optimized choices;
6. The average national feed, which will reflect the current situation in terms of composition and ingredients' origins.

Table 41 gives the composition of each feed. The first type of feed is a conventional maize grain-based feed. It is given to the animal in a quantity of 2.5% of its live weight per day, plus a constant amount of 1.5 kg of wheat straw (Alfredo Sendim; personal communication). The second type of feed is maize silage-based. The animals are fed with 3% of their live weight of this composition per day (Alfredo Sendim; personal communication). Since many feeds are used for each type of animal and age, it is impossible to use a given feed composition as representative (Fernando Anjos; personal communication). Therefore, the average national feed for finishing calves was obtained from the fodder balance (IACA, 2004; INE, http://www.ine.pt, 2005), and so there is a large uncertainty in its composition. This feed should only be considered as a benchmark for the other two.

Table 81 – Base composition of the feeds studied.

Ingredient	% (kg ingredient / 100 kg feed)		
	Feed 1	Feed 2	Average feed
Maize (silage)		58.6	
Maize (grain)	20.0	12.5	13.7
Corn Gluten Feed	20.0	6.6	27.0
Wheat (grain)	19.0	6.6	4.9
Barley	10.0		2.1
Soy meal (44% protein)	6.0	9.1	16.9
Manioc	5.0		
DPG	5.0		
Palm kernel cake	5.0		
Sunflower	4.8		
Carbonate	2.1		
Fats	0.9		
Bicarbonate	0.8		
Salt	0.6		
Premix	0.5	0.8	
Urea	0.3		
Others			36.3[70]
Wheat (straw)	(1.5 kg.day^{-1})[71]	5.8	

[70] The "others" are undisclosed cereals and by-products. Due to lack of information, it was not possible to determine what they are.

The animals are fed from the age of 6 to 8 months (180 to 200 kg live weight) to the age of 12 to 14 months (360 to 400 kg live weight), and considering the average value in each of those, it may be shown that the first type of feed, which provides a fixed quantity of straw and 2.5% of the animal's live weight, is given to the animal in larger amounts in the beginning, and smaller in the end, than the second type of feed, which varies the quantity of straw within the base feed (Alfredo Sendim; personal communication).

Here, we considered two intervals of 2.4 months: (1) from 7.2 to 9.6 months, when the animals are fed in 60% by commercial feed and 40% from pasture; (2) from 9.6 to 12 months, animals are confined to stables and completely fed with the commercial feed. Feed 1 and the average feed allow the animals to grow more rapidly, at the rate of 1.5 $kg.day^{-1}$, whereas Feed 2 provides a slower growth rate of 1.2 $kg.day^{-1}$. Therefore, animals fed with Feed 2 end the second period with less weight than those fed in the other cases (Table 42) (Alfredo Sendim; personal communication).

Table 82 – Animal weights when fed with each feed

Age (months)	Weight 1 and average (kg)	Weight 2 (kg)
7.2	190	190
9.6	298	276.4
12	406	362.8

Ingredient origin

In Portugal, productivity of each cereal depends on geographical location, as shown in Table 83 for 2002 (INE, http://www.ine.pt, 2005).

Table 83 – Cereal production by agricultural zone

Agricultural zones	Area (ha)			Production (t)			Productivity ($t.ha^{-1}$)		
	Wheat	Maize	Barley	Wheat	Maize	Barley	Wheat	Maize	Barley
Entre-Douro e Minho	80	34 645	10	65	129 581	6	0.8	3.7	0.6
Trás-os-Montes	10 493	8 415	586	16 652	14 359	475	1.6	1.7	0.8
Beira Litoral	1 215	30 605	127	2 282	123 575	136	1.9	4	1.1
Beira Interior	1 835	11 366	122	2 012	33 477	131	1.1	2.9	1.1
Ribatejo e Oeste	13 607	33 817	1 538	33 890	333 077	3 028	2.5	9.8	2
Alentejo	200 562	19 428	7 486	354 481	153 356	15 043	1.8	7.9	2
Algarve	2 832	1 239	1 328	3 589	7 191	1 195	1.3	5.8	0.9

The imported cereals come mainly from France (38.4%), Germany (12.6%), Ukraine (12.4%), Argentina (11.1%), Spain (6.8%) and the United Kingdom (6.7%) (INE, http://www.ine.pt, 2005). The most imported cereal is wheat (45% of all cereals), mainly imported from France (40.4% of all wheat), followed by maize (38% of all

[71] Straw is given to the animals in a fixed quantity, which does not depend on the quantity of feed also given. It could not be determined whether the average feed contains straw, but since its fibre content is equal to that of Feed 1 (as shown next) it is plausible to assume that it does not (straw is mainly used for fibre).

cereals), from Argentina (42% of all maize) (FAOSTAT, http://faostat.fao.org, 2005; INE, http://www.ine.pt, 2005; IACA, 2004).

Soybeans and its derivates is the most imported oilseed (78% of all oilseeds), followed by sunflower (10%) and palm kernel cake (9%) (INE, http://www.ine.pt, 2005; IACA, 2004). Soybeans and soybean meal are mainly imported from Brazil (51.1% of all imports) and soy oil from Argentina (81.8%). However, while soy is fully imported, sunflower is, to some extent, produced within the country. Palm kernel cake is mainly imported from Malaysia and Indonesia. Corn gluten feed is fully imported from the United States of America (IACA, 2004).

Imports are mainly transported by road, for intra-UE importation, or by sea (otherwise, mainly America). Transport by railroad or airplane is not significant (INE, http://www.ine.pt, 2005).

Industrial processing

Licencing by the IPPC European directive is mandatory for all facilities dedicated to the processing of animal feeds with a finished product production capacity superior to 300 tons per day. The directive establishes the correct environmental practices that the industry should follow. This regulation also exists in Portuguese legislation, through DL 194/2000. In 2002, there were 22 licensed companies with 27 industrial facilities, 13 of which in Lisbon and Tagus Valley[72], 10 in the centre of the country, 3 in the north and 1 in Alentejo. Total production was 3 478 890 tons, and the business volume was 6.4E08 € (FIPA[73]; APA[74]).

According to FIPA and APA, the most common production scheme starts with the reception, discharge and storing of the raw materials (solid and liquid). Next, these are mixed and crushed, and then, in granulation, the feed is assembled and given its form. It is then cooled and bagged. In all of these operations, the main impacts are the high electrical energy consumption (materials transport indoors), the products resultant from the gas combustion in the boiler (used to form water vapour, which aggregates the ingredients during the granulation stage), other gas emissions (for example, from fossil fuel burning) and dust emissions (from the cereals). Since most materials arrive in bulk, the formation of solid waste from packaging is small. Therefore, most solid waste, as well as wastewater, produced within the facility are of domestic nature (from bathrooms, eventually canteens).

The net emissions considered were based on available environmental licences, regarding two factories: Racentro, in Beira Litoral, and SAPJU, in Alentejo. The data in each case is presented in Table 84.

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[72] Lisbon and Tagus Valley corresponds to agricultural region Ribatejo e Oeste.

[73] Federação das Indústrias Portuguesas Agro-Alimentares. http://www.fipa.pt, 2005.

[74] Agência Portuguesa do Ambiente (Portuguese Environmental Agency). http://www.apambiente.pt, 2005.

Table 84 – Animal feeds sector data

		Racentro	SAPJU
Available capacity ($t.day^{-1}$)		500	590
Electric energy ($kWh.yr^{-1}$)		1 802 625	22 700
Propane gas ($t.yr^{-1}$)		148.64	-
Gas emissions (maximum values) ($mg.Nm^{-3}$)	CO	1000	1000
	Organic volatile compounds	50	50
	NO_x	1500	1500
Water spent ($m^3.yr^{-1}$)		-	140
Waste water	pH	6.5 – 8.5	-
	Hydrocarbons ($mg.L^{-1}$)	10	0

Based on these values, an average first estimation of the process was determined, in order to assess the impacts of such methods in the life cycle of the feed. In SAPJU the water effluent has domestic characteristics, as in Racentro, but in the latter the effluent presents additionally a certain level of hydrocarbons, from the maintenance shop. The difference in electrical energy use is striking, but no justification could be found. Therefore, we used the highest value in all calculations, to assure the worst case scenario is considered.

After industrial processing, the feed is transported to regional storehouses or directly to farms themselves, usually by road (INE, http://www.ine.pt, 2005).

Software and inventory used

Life cycle assessment was performed using SimaPro 6.0, developed by the National Reuse of Waste Research Programme and Pré Consultants of the Netherlands, which is widely used in assessing environmental performance in all areas. Two aggregated weighing methods were used: "Ecoindicator 95" and "Ecoindicator 99". Even though conclusions drawn are often similar (Luo *et al.*, 2001), it is important to use both, as they consider different themes and a different conception. Based on studies like Pluimers *et al.* (2000), we considered that the most important themes from Ecoindicator 95 in the case of agriculture were eutrophication, acidification, greenhouse gas emissions, heavy metals and energy use. The themes chosen from Ecoindicator 99 were climate change, acidification/eutrophication and land use. As results will show, these are also the themes to which a highest percentage of global impact is attributable. The aggregated single score values of both Ecoindicators consider not only these themes, but also all those available.

In all crops, nutrient leaching was considered as presented in the last Appendix (for maize) in Table 75, based on van der Werf *et al.* (2005).

NO^-_3 leaching values vary with crop, production method, soil characteristics and content in nutrients and rain occurrence, and therefore a local analysis should clearly study these parameters and their relation to leaching. We did not find any data directly referring to the zones studied. Therefore, in order to assess whether the value indicated above is adequate for this paper's conditions, several Spanish studies relative to maize production were consulted. Spain is especially relevant, given its similarity with Portugal in terms of soil and climate characteristics.

Moreno *et al.* (1996) studied nitrate leaching under irrigation in Spain and reached values of 150 and 43 kg $N.ha^{-1}$ leached, corresponding respectively to 500 and 170 kg $N.ha^{-1}$ applied. Villar-Mir *et al.* (2002), for an N application of 250 to 340 kg $N.ha^{-1}$, measured 60 kg $N.ha^{-1}$ leached. Diez *et al.* (1997) studied the effect of various choices in fertilizer and irrigation, emphasising that with convenient irrigation it is possible to diminish leaching in more than 90% for each fertilizer used. These results confirm that a value of 40 kg $NO_3^-.ha^{-1}$ is plausible as a first approach.

As for wheat, EC (1997) indicates a reference leaching of 35 kg $N.ha^{-1}$ for loamy soils, and 45 kg $N.ha^{-1}$ for sandy soils, under a recommended fertilization level of 180 kg $N.ha^{-1}$.

In this paper, since the main objective is to obtain a comparison between average zones for each ingredient, the generic value was used as indicated above, but an uncertainty analysis was made in order to verify whether results would change significantly from the variation in input parameters.

Water consumption was also analysed, by determining the water used to produce the amount of each cereal that is used in each feed. The direct occupation of space was also calculated, equal to the area needed to produce 1 t of each cereal. The latter is an approximation of Ecoindicator 99's theme land use. However, water use is not usually considered in life cycle analysis, but is of great relevance in agriculture, and so we integrated it in direct occupation of space, and from this in land use. Since land use is measured in a specific Ecoindicator unit ($PDF.m^2.year$) to which no relation exists, water use could not be integrated directly. In order to do so, we used the annual average precipitation in Portugal, estimated as 8890 $m^3.ha^{-1}.yr^{-1}$ (EEA, http://reports.eea.eu.int/92-9167-056-1/en/page003.html, 2005), which allowed us to transform water volume in area and vice-versa. For other countries, we used the world surface average precipitation, estimated as 10 500 $m^3.ha^{-1}.yr^{-1}$ [75]. Then, we converted direct occupation of space to Ecoindicator 99's theme land use, and those to aggregated single score values. This method assumes water use has the same Ecoindicator wheighting as land use. Since crops use both irrigation (except rainfed crops) and rain water, and we only considered in water use the first one, this analysis also allowed us to consider rain water in the global water used by the crop. Such method is, however, a mere estimate, since rain water is not necessarily correspondent, in its temporal and spatial distribution, to the water used for irrigation. Furthermore, infiltration and evaporation would have to be considered in a more accurate estimate, and this approach is not valid when two cultures are sown in the same year.

Data sources

In order to find the environmentally best feed, the first step is to optimize the choices of ingredients. Generically three types of ingredients are the most used - cereals, oilseeds and by-products. Therefore, from these groups, the ones that constitute the feeds were analysed. It is then important to determine the best option in terms of production zones, since they present different methods, inputs and productivities. The data was collected in crop fact sheets provided by the GPP (2001) model for the whole country, and the chosen zones were those with higher productivity and/or production for each ingredient. The sheets analysed and the generic information gathered within them is presented in Table 89. Comparing productivity values therein with the

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[75] Value from http://www.physicalgeography.net/fundamentals/8g.html.

averages in Table 83, it may be seen that sheets are not always representative of the corresponding region. In every case of wheat the sheets overestimate the productivity, and therefore results may still be compared, as this paper intends. The same happens for maize, except in Ribatejo e Oeste. Barley's sheets have productivities closer to the average for the regions considered.

The data in the crop fact sheets was the used to determine the production impact of each cereal. All values for maize were used according to Teixeira *et al.* (2007). The environmental impact of each type of barley (common and malts) and wheat products (grain and straw) was allocated according to its economic value.

As for oilseeds, the most used and imported one is soy, and it comes mainly from Argentina as soy oil and Brazil as soy meal. Therefore, it is important to characterize its production in its country of origin. Since no information could be found about Brazil, it was considered that all soybeans are imported from Argentina, which is also a massive exporter of soy, since there it already occupies more land than all other crops taken together (Dros, 2004). There are basically two types of soy produced: type I, which is grown as a monoculture in rotation (bi-annual) with maize, and type II, which is grown together (at the same time) with wheat.

The production of soy in Argentina, commonly in the northeast of the country (Dros, 2004), usually does not include any fertilization, because traditionally it is considered by farmers that the crop's production is not responsive to fertilizers. According to MADRP[76], soybeans also capture some atmospheric nitrogen (Table 85), supplying about 75% of its needs (Table 86).

Table 85 – Quantity of N fixed by soy.

Plant	Fixed N ($kg.ha^{-1}.yr^{-1}$)	
	Common interval	Typical value
Soy (*Glycine soja*)	65 - 179	112

Table 86 – Quantity of N assimilated by soy.

Plant	Product	Production ($t.ha^{-1}$)	Assimilation ($kg.ha^{-1}$)		
			N	P_2O_5	K_2O
Soy (*Glycine soja*)	Grain	2	150	35	60

N – Nitrogen; P_2O_5 – Phosphorus oxide; K_2O – potassium oxide.

Therefore, with the depletion of the soil's nutrient fertility, it should be expected that the situation changes (Gutierrez-Boem *et al.,* 1999). So, we created a scenario in which we considered the level of fertilization needed, according to Galarza *et al.* (2001). Those quantities are shown in Table 87.

[76] Ministério da Agricultura, do Desenvolvimento Rural e das Pescas (Portuguese Ministry of Agriculture), http://www.min-agricultura.pt.

Table 87 – Nutrient intake by soybeans' plant

Nutrient	Quantity (kg.t^{-1} soy)
N	54,0
P	5,4
K	15,7
Ca	2,3
Mg	2,3
S	3,4
B	0,026
Cu	0,012
Fe	0,131
Mn	0,025
Mo	0,005
Zn	0,039

Since Mn, Mo and Zn are not considered in SimaPro's database, and their quantity is very small, these elements were not considered. Soy transportation distances from Argentina to Portugal were taken to be the same as for maize.

It should also be noticed that type II has some supplementation granted by the fertilization of wheat, and therefore the extra nutritional needs are not the same, but, for simplification, they were considered to be so in this paper.

Allocation

The use of by-products, like corn gluten feed, as well as some cereals and oilseeds that have to be processed industrially, raises the allocation problem. Values for mass and price allocation between by-products were considered according to Cederberg and Mattsson (2000), (Table 88). Allocation may as well consider energy, as in Kim and Dale (2002). In this paper price allocation was used, but the effect of such choice was studied by also calculating some results for mass allocation.

Table 88 – Mass and price allocation for ingredients used

Crop	Products	Mass allocation (%)	Price allocation (%)
Soybean	Oil	20	31
	Meal	80	69
Corn for starch production	Starch	63	78
	Corn gluten feed	20	8
	Corn gluten meal	5	10
	German meal	7	4
Palm oil	Crude palm oil	77	83
	Crude palm kernel oil	10	14
	Kernel expel	13	3
Sunflower	Oil	31	63
	Meal	68	37

Table 88 shows that the by-product "kernel expel" is also referred to as "palm kernel cake", used in Feed 1. Following Cederberg and Mattsson (2000), since the market value allocation of this product is only 3%, only the impact of transportation was considered. As the country of origin is Malaysia, transportation by sea and land was considered. DPG, a by-product of barley malt from the beer production process, is fully imported by Portugal from the United States of America, but due to lack of information from the process of barley production in the country of origin, an alternative fully produced in Portugal was considered.

Functional unit

Given the situation described above, the functional unit for the evaluation of each feed had to be relative to the weight variation of the animals, and so all results regarding feeds will be compared with the unit impact.kg^{-1} gained by the animal.

Expected results

The analysis on this paper was structured according to the following steps:

- Determine the impact of each crop's production through life cycle assessment;
- Determine the impact of transportation and the feeds' industrial processing;
- Integrate the latter information to determine each feed's impact;
- Study optimization possibilities for given ingredients;
- Study the relevance of transportation in global impact;
- Verify if impact allocation of by-products influences the final result;
- Study and integrate water use in life cycle analysis results;
- Perform an uncertainty analysis;
- Determine the nutritional composition of the feeds;
- Study if any ingredient with a relevant impact can be substituted.

Table 89 – Production zones and methods analysed

Cereal	Sheet	Production zone	Quantity Produced ($kg.ha^{-1}$) *	Space occupation ($ha.t^{-1}$)	Market value (€.kg^{-1})	Irrigation method**	N fertilizer ($kg.ha^{-1}$)	P_2O_5 fertilizer ($kg.ha^{-1}$)	K_2O fertilizer ($kg.ha^{-1}$)	Water quantity ($m^3.ha^{-1}$)	Water per t ($m^3.t^{-1}$)	Number of activity months	Total cost (€.ha^{-1})	Production value (€.ha^{-1})	Revenue (€.ha^{-1})
Barley ***	cev1	Alentejo	1 400 + 1 500	0.714 + 0.667	0.13 + 0.07	Rainfed	38	24	10	0	0	7	505.9	297.28	-208.62
	cev2	Alentejo	1 800 + 1 800	0.556 + 0.556	0.13 + 0.07	Rainfed	38	24	10	0	0	7	505.9	372.6	-133.30
	cev3	Ribatejo e Oeste	2 500 + 2 000	0.400 + 0.500	0.13 + 0.07	Rainfed	75	72	30	0	0	10	473.09	473.86	0.77
Maize	QF	Beira Interior	6 500	0.154	0.13	Gravity, Pumping	246	172	262	9 000	1 385	8	1 087.76	845.00	-242.76
	mil8	Beira Interior****	6 500	0.154	0.14	Sprinkling	69	56	56	1 200	185	8	1 461.48	914.3	-547.18
	mil14	Ribatejo e Oeste	10 000	0.100	0.14	Furrows, gravity	219	105	105	6 000	600	4	1 578.12	1 406.61	-171.51
	mil15	Beira Litoral	10 000	0.100	0.14	Sprinkling	165	70	70	2 400	240	4	1 505.43	1 406.61	-98.82
	ARG	Argentina	8 540	0.117	0.04	Gravity	69	9	23	2 470	290	-	135.44	362.72	227.28
	Silage	Alentejo	42 000	0.023	0.03	Gravity, Furrows	240	168	168	4000	95	-	35.98	30.00	-5.98
Wheat	tri1	Trás-os-Montes	1 500 + 1 500	0.667 + 0.667	0.13 + 0.07	Rainfed	55	42	42	0	0	12	436.26	306.76	-129.50
	tri5	Ribatejo e Oeste, Alentejo	3 000 + 2 500	0.333 + 0.400	0.13 + 0.07	Rainfed	69	60	25	0	0	7	558.88	568.63	9.75
	tri7	Ribatejo e Oeste	4 500 + 1 500	0.222 + 0.667	0.14 + 0.07	Rainfed	145	108	45	0	0	10	566.11	729.49	163.38
	tri10	Ribatejo e Oeste, Alentejo	5 000 + 3 500	0.200 + 0.286	0.13 + 0.07	Pivot	134	84	35	1 500	175	7	713.09	897.84	184.75
Sunflower	gir1	Beira Interior, Alentejo	2 500	0.400	0.2	Sprinkling	90	63	63	2 400	960	7	1 011.15	505.03	-506.12
	gir2	Ribatejo e Oeste	2 500	0.400	0.21	Rolling	69	63	63	1 500	600	7	711.6	517.5	-194.10
Soybeans	I	Argentina	3 250	0.308	0.09	-	0	0	0	-	-	-	63.24	295.63	232.39
	II	Argentina	2 310	0.433	0.09	-	-	-	-	-	-	-	86.48	143.85	57.36

*** When two values are shown in the same column the first refers to wheat and the second to straw. ** For more on irrigation methods, and its environmental and energetic evaluation, see Esteves et al. (1995) and Vieira et al. (2005). *** First sheet is common barley and others barley malts. **** Maize production in Beira Interior additionally requires 20 000 $kg.ha^{-1}$ of manure.**

Results

Crop production

Maize

Results for maize were considered according to Appendix III – Environmental analysis of maize production. Silage maize is produced in Alentejo (where the feed is given to the animals), while grain maize is transported from Ribatejo e Oeste (the environmentally best option for production).

Wheat

Wheat (*Triticum aestivum*) production results in two separate parts: grain and straw (in variable proportions), both of which are used in animal feeds. It may, then, be shown in Table 90 that irrigated grain wheat has the lower impact in Ecoindicator 95, but rainfed straw in RO and ALE wheat has a lower impact in that case. This happens because the rainfed crop in that produces more grain wheat in proportion to straw, and so a smaller percentage of impact is attributed to straw.

Table 90 – Relative contributions in each environmental category, allocated to each product by economic value (Ecoindicator 95).

Production zone	Single score ($Pt.ha^{-1}$)	Product type	t	$€.t^{-1}$	€	%	Single score ($Pt.ha^{-1}$)	Single score ($Pt.t^{-1}$)
Irrigated Wheat (RO, ALE)	9.7	Grain	5.0	130	195	65	6.3	1.3
		Straw	3.5	70	105	35	3.4	1.0
Rainfed Wheat (RO)	14.4	Grain	4.5	130	390	69	9.9	2.2
		Straw	1.5	70	175	31	4.5	3.0
Rainfed Wheat (RO, ALE)	8.9	Grain	3.0	130	585	85	7.6	2.5
		Straw	2.5	70	105	15	1.3	0.5
Rainfed Wheat (TM)	6.4	Grain	1.5	130	650	73	4.7	3.1
		Straw	1.5	70	245	27	1.7	1.2

RO – Ribatejo e Oeste; ALE – Alentejo; TM – Trás-os-Montes.

Barley

As for wheat, each of the types of barley (*Hordeum vulgare* L.) has two products. Common barley is used directly in animal feeds (principal product), while from barley malts, which is used to produce beer, only the secondary product may be used in animal feeds. The impact again had to be distributed according to economic value, and major results are shown in Table 91.

Table 91 – Relative contributions in each environmental category, allocated to each product by economic value (Ecoindicator 95).

Production zone	Single score ($Pt.ha^{-1}$)	Product type	t	$€.t^{-1}$	€	%	Single score ($Pt.ha^{-1}$)	Single score ($Pt.t^{-1}$)
Common Barley (ALE)	5.0	Principal	1.4	130	182	63	3.2	2.3
		Secondary	1.5	70	105	37	1.9	1.2
Barley Malts (ALE)	5.0	Principal	1.8	130	234	65	3.2	1.8
		Secondary	1.8	70	126	35	1.7	1.0
Barley Malts (RO)	6.4	Principal	2.5	130	325	70	4.5	1.8
		Secondary	2.0	70	140	30	1.9	1.0

ALE – Alentejo; RO – Ribatejo e Oeste.

Sunflower

Sunflower (*Helianthus annum L.*), mainly produced in Portugal, is the second most used oilseed in feeds (10%). The environmentally best option in this case is grown in Beira Interior or Alentejo, with an Ecoindicator 95 single score result of 1.85. The other option studied was Ribatejo e Oeste, but the single score value is 1.95.

National ingredients' best production zones

Performing the same analysis ("Ecoindicator 95") for the other cereals and sunflower, all of which are produced in Portugal, the environmentally best national options were found and are indicated in Table 92. Therefore, not regarding transportation:

Table 92 – Best production zones for each cereal produced in Portugal

Product	Best production zone
Grain maize	Ribatejo e Oeste
Wheat (grain)	Ribatejo e Oeste, Alentejo (irrigated)
Wheat (straw)	Ribatejo e Oeste, Alentejo (rainfed)
Barley	Alentejo
Sunflower	Beira Interior, Alentejo

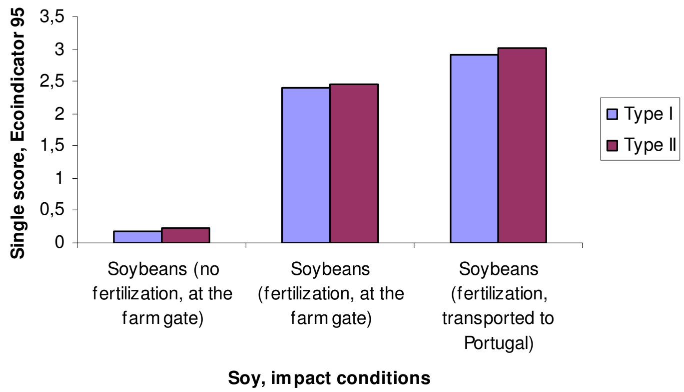
Considering that the factory where the feed shall be produced is located in Lisbon and Tagus Valley (where most are), the impact of transportation from each zone of each ingredient was considered as for maize in Teixeira *et al.* (2007), and average distances between regions were assumed to be 75 km for Ribatejo e Oeste and 150 km for Alentejo. It was then concluded that transportation does not change the results, and the best production zones in Portugal remain the same referred above.

Soy

In the case of soybeans (*Glycine max* L.), produced in and imported from Argentina, the total impact was determined when there is no fertilization and no transportation (growing as is), when there is fertilization and no transportation (growing as it will be) and when both occur. The results in both ecoindicators show that the environmental impact of growing soybeans of both types is, in great part, provided by fertilization. The impact of fertilization is even greater than that of transportation to Portugal. Soybeans of type II are less adequate than type I when compared in the same scenario, but since type II is produced together with wheat it is expected that fertilization will not have to be so high, and so it may be an even better option. Since no information could be gathered about such type of soy, this option was not used in this paper.

Further results may be explained by the fact that, even though they fix a large quantity of N, soybeans still require fertilization. Results for Ecoindicator 95 are shown in Figure 51.

Figure 51 – The impact of cultivating soy in Argentina, with and without fertilization, and with and without considering transportation (Ecoindicator 95)



<u>Corn Gluten Feed</u>

In order to determine the impact of corn gluten feed, data on impacts of the process of U.S.A. corn production and subsequent starch extraction from the SimaPro 6.0 database was directly used, and then allocation according to economic value was done. Ecoindicator 95 single score value, not regarding transportation, is 1.72.

Industrial processing and transportation to animal farm

The impacts of processing the feed (***P***) were based on average maximum production capacity and emission limits, and therefore the worst case scenario was considered, and typical specific emissions were taken to be those of Racentro (Table 84). However, as most facilities are located in Lisbon and Tagus Valley, we considered the processing factory was there. Ecoindicator 95 single score value is 1.8E-02. Since the animal farm was considered to be in Alentejo, additional transportation of the final feed was considered (***F***).

Aggregated feeds

<u>Feed 1</u>

Given the values shown before, we then calculated the impact of each feed. In the case of the grain maize-based feed, overall impact is obtained by using Equation 2. Information for 89.7% of the feed was used.

$$I_{\text{Feed 1}} = \sum_{\substack{\text{i = Grain maize,} \\ \text{Grain wheat,} \\ \text{Barley,} \\ \text{Sunflower,} \\ \text{Soy,} \\ \text{Corn Gluten Feed,} \\ \text{Straw}}} (I_i + T_i) + P_{\text{Feed 1}} + F_{\text{Feed 1}} \qquad (2)$$

Considering the total amount of each ingredient used[77], we determined the global impact of production and transportation multiplying each quantity and the unitary impact (Table 93). Finally, the value obtained for the total impact using SimaPro's Ecoindicator 95 is divided by the variation of weight of the animal during the period (Table 96).

Table 93 – Ingredients' impacts over the period of time analysed (I and T) for Feed 1 (Ecoindicator 95).

Ingredient	Quantity (kg ingredient.t^{-1} feed)	Quantity used in 4.8 months (kg)	Unitary productio n impact ($Pt.t^{-1}$)	Impact I (Pt)	Unitary transporaio n impact ($Pt.t^{-1}$)	Impact T (Pt)
Maize (grain)	200	200.03	1.2	0.244	0.066	0.0133
Corn Gluten Feed	200	200.03	0.1	0.028	0.412	0.0825
Wheat (grain)	189	189.03	1.3	0.238	0.066	0.0125
Barley	100	100.01	2.5	0.252	0.133	0.0133
Soy (40% protein)	60	60.01	1.7	0.100	0.514	0.0309
DPG	50	50.01	0.3	0.016	0.022	0.0011
Palm kernel cake	50	50.01	0	0.000	0.514	0.0257
Sunflower	48	48.01	1.1	0.052	0.133	0.0064
Straw	1.5 $kg.day^{-1}$	7.20	0.3	0.002	0.066	0.0005
Total:	897	904.32		0.932		0.186

Feed 2

In the case of the silage maize-based feed, considering 99% of its composition for which information was obtained, overall impact is determined using Equation 3.

$$I_{\text{Feed 2}} = \sum_{\substack{\text{i = Grain maize,} \\ \text{Silage maize,} \\ \text{Grain wheat,} \\ \text{Soy,} \\ \text{Corn Gluten Feed,} \\ \text{Straw}}} (I_i + T_i) + P_{\text{Feed 2}} + F_{\text{Feed 2}} \qquad (3)$$

Silage maize is not processed nor transported, since it is usually grown in the same place where it is consumed. So, in each ton of Feed 2, only the other ingredients are considered in those steps of the life cycle. Once, again the total quantity of each ingredient used was determined (Table 94), and then the aggregated impact (Table 96).

[77] According to the amount per day in "Composition of the feeds".

Table 94 – Ingredients' impacts over the period of time analysed (I and T) for Feed 2 (Ecoindicator 95).

Ingredient	Quantity (kg ingredient.t^{-1} feed)	Quantity used in 4.8 months (kg)	Unitary production impact (Pt.t^{-1})	Impact I (Pt)	Unitary transporaion impact (Pt.t^{-1})	Impact T (Pt)
Maize (silage)	586	635.94	0.5	0.293	0	0
Maize (grain)	125	135.65	1.2	0.165	0.066	0.0090
Soy (44% protein)	91	98.76	1.7	0.164	0.514	0.0508
Wheat (grain)	66	71.62	1.3	0.090	0.066	0.0047
Corn Gluten Feed	66	71.62	0.1	0.010	0.412	0.0295
Wheat (straw)	58	62.94	0.3	0.018	0.066	0.0042
Total:	992	1076.54		0.741		0.098

<u>Feed 3</u>

The average national feed's impact is determined according to Equation 4. Oly 65% is accounted for.

$$I_{\text{Average Feed}} = \sum_{\substack{\text{i = Grain maize,} \\ \text{Grain wheat,} \\ \text{Common barley,} \\ \text{Soy,} \\ \text{Corn Gluten Feed}}} (I_i + T_i) + P_{\text{Average Feed}} + F_{\text{Average Feed}} \qquad (4)$$

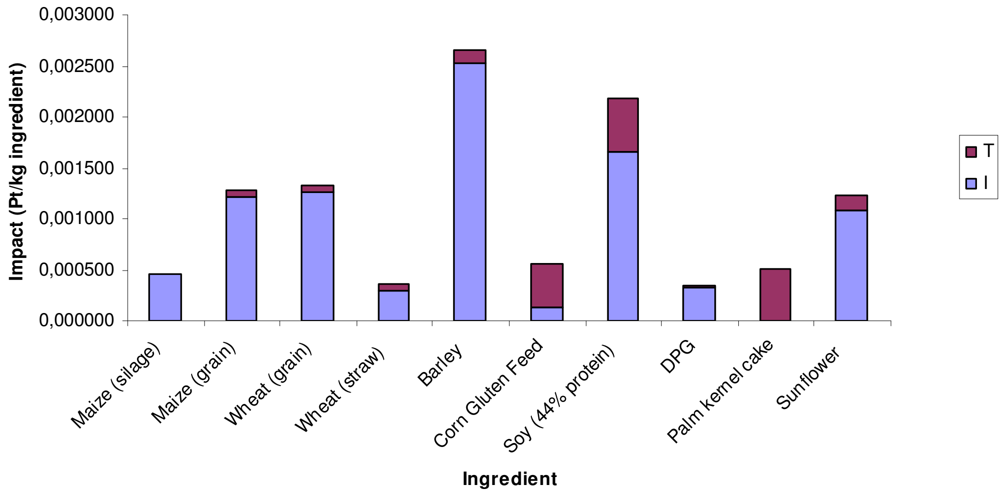
In this case, it was considered that the feed is given to the animals in 2.5% of their weight during the 4.8 months period, in order to calculate the aggregated impact (Table 95 and Table 96).

Table 95 – Ingredients' impacts over the period of time analysed (I and T) for the average national feed (Ecoindicator 95).

Ingredient	Quantity (kg ingredient.t^{-1} feed)	Quantity used in 4.8 months (kg)	Unitary production impact (Pt.t^{-1})	Impact I (Pt)	Unitary transporaion impact (Pt.t^{-1})	Impact T (Pt)
Barley	23.9	21.44	2.5	0.054	0.1326	0.0028
Maize (grain)	158.2	141.90	1.2	0.173	0.0663	0.0094
Wheat (grain)	56.9	51.02	1.3	0.064	0.0663	0.0034
Corn Gluten Feed	467.5	419.42	0.1	0.058	0.4122	0.1729
Soy	293.6	263.35	1.7	0.438	0.5142	0.1354
Total:	1000.0	897.12		0.787		0.324

In order to observe the relation between impact and feed composition, we first determined the impact of each unit (in weight) of the ingredients. Results are presented in Figure 26. Barley and soy are the ingredients with the highest unit impact. Silage maize and wheat straw have the lowest unit impact, along with corn gluten feed, DPG and palm kernel cake, which are by-products.

Figure 52 – Unit impact of each of the ingredients (Ecoindicator 95).



Summary of results

Results in each step of the life cycle are summarized in Table 96. As for Feed 1, the impact obtained in Ecoindicator 95 is mostly due to the production of the ingredients, which is the critical parameter. Transportation has a smaller impact on Feed 2. This happens because silage maize is not incorporated industrially in the feed. Usually, silage is grown close to the animal farm and then given to the animals together with the feed. The average feed is the one where transportation is more important, because the ingredients that have to be transported from a greater distance, namely corn gluten feed and soy, are in a larger proportion than in Feeds 1 and 2. Industrial processing has a very small relevance in all cases.

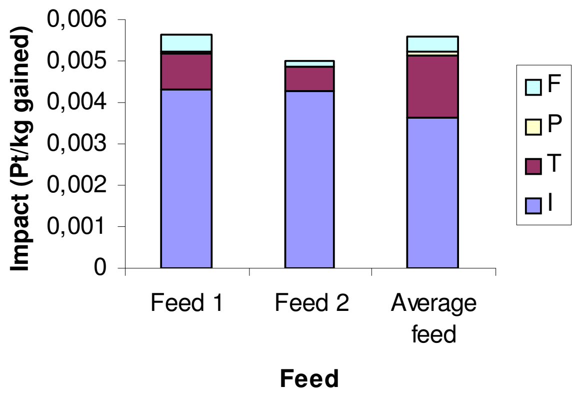
Table 96 – Impact summary for each feed, by step in the life cycle.

	Feed 1		Feed 2		Average Feed	
	Impact ($Pt.kg^{-1}$ gained)	%	Impact ($Pt.kg^{-1}$ gained)	%	Impact ($Pt.kg^{-1}$ gained)	%
I	4.3×10^{-3}	77	4.3×10^{-3}	85	3.6×10^{-3}	65
T	8.6×10^{-4}	15	5.7×10^{-4}	11	1.5×10^{-3}	27
P	7.0×10^{-5}	1	3.0×10^{-5}	1	7.0×10^{-5}	1
F	3.7×10^{-4}	7	1.3×10^{-4}	3	3.7×10^{-4}	7
Total	5.6×10^{-3}	100	5.0×10^{-3}	100	5.6×10^{-3}	100

I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Figure 27 shows that Feed 2 has a 10% lower environmental impact than Feed 1, which is a slightly better option than the national average feed. However, all results concerning the average feed have to be interpreted carefully, since the full composition is not known.

Figure 53 – Overall impact of each feed ($Pt.kg^{-1}$ gained), Ecoindicator 95



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Table 97 shows the impact accountable for each ingredient in the analysed feeds. In Feed 1, maize and barley are the ingredients with the most impact, which is proportionally greater than its weight in the feed. Barley is very demanding, since for input levels equivalent to other cereals its productivity is very small. As for Feed 2, silage maize has the bigger contribution, even though its unit impact is the smallest (Figure 26), since it is used in larger amounts (58.6% of the feed in mass). Maize is responsible for a smaller impact than soy for the reasons stated in the case of the first feed. In the average feed, soy has the bigger contribution, not only because of its unit impact (Figure 26) but also because it is used in large amounts.

Table 97 – Impact of the amount of each ingredient used in each feed (production and transport).

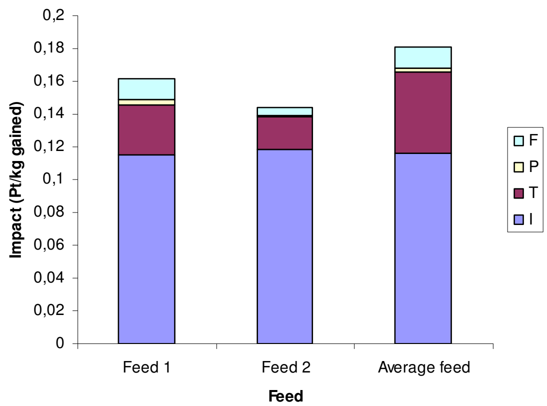
Ingredient	Feed 1		Feed 2		Average feed	
	I + T ($Pt.kg^{-1}$ gained)	%	I + T ($Pt.kg^{-1}$ gained)	%	I + T ($Pt.kg^{-1}$ gained)	%
Maize (silage)	-	-	1.7×10^{-3}	34.9	-	-
Maize (grain)	1.2×10^{-3}	23.0	1.0×10^{-3}	20.8	8.5×10^{-4}	16.4
Wheat (grain)	1.2×10^{-3}	22.4	5.5×10^{-4}	11.3	3.1×10^{-4}	6.1
Wheat (straw)	1.2×10^{-5}	0.2	1.3×10^{-3}	2.7	-	-
Barley	1.2×10^{-3}	23.7	-	-	2.6×10^{-4}	5.1
Corn Gluten Feed	5.1×10^{-4}	9.8	2.3×10^{-4}	4.7	1.1×10^{-3}	20.8
Soy (44% protein)	6.0×10^{-4}	11.7	1.2×10^{-3}	25.6	2.7×10^{-3}	51.6
DPG	8.0×10^{-5}	1.5	-	-	-	-
Palm kernel cake	1.2×10^{-4}	2.3	-	-	-	-
Sunflower	2.7×10^{-4}	5.2	-	-	-	-
Total:	5.2×10^{-3}	100	4.9×10^{-3}	100	5.1×10^{-3}	100

I – impact of the production of ingredients; T – impact of the transportation of ingredients.

Using Ecoindicator 99, results point out Feed 2 as the best option and the average feed as the worst, this time by a large margin (20%), as shown in Figure 54. This indicates that Ecoindicator 99 may be more sensitive to the impact of transportation, since in this case the impact of the ingredients' production (I) is smaller for the average feed, as in

Ecoindicator 95 (although in this case the difference was larger), but in the remaining life cycle the impact is much bigger, since more imported products are used, and therefore that difference is more significant in this case. Still, Feed 2 is better than Feed 1 in the same margin, about 10%.

Figure 54 – Overall impact of each feed ($Pt.kg^{-1}$ gained), Ecoindicator 99

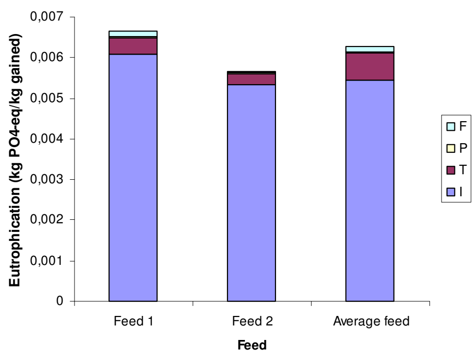


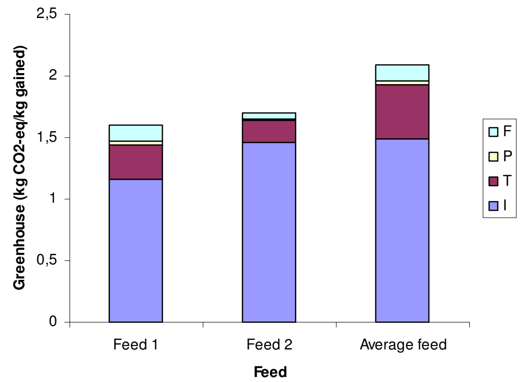
I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

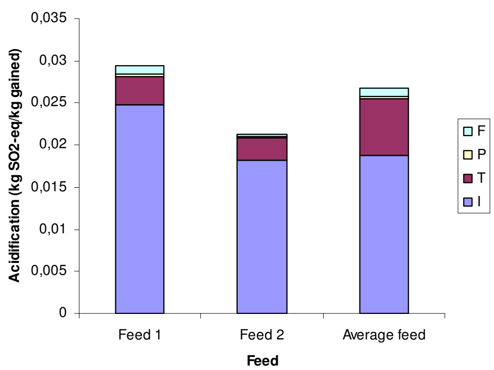
Results per environmental theme

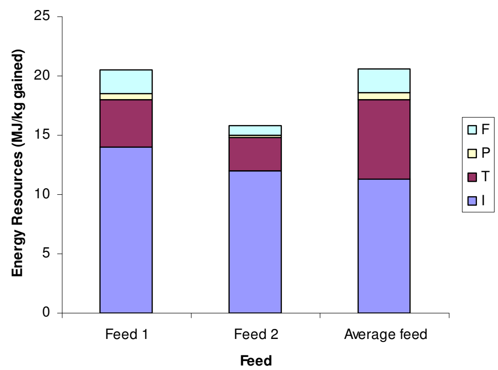
As for particular categories of life cycle analysis contributing to Ecoindicator 95, it may be shown that Feed 2 has a lower impact in eutrophication, acidification and energy resources (especially in the latter, since it requires less transportation). In those themes, Feed 1 is always the worst option. However, in greenhouse gas emissions, Feed 1 has the lower impact, and the average feed is the worst. This may be explained by the fact that such feed uses the smaller amount of soy, and that is the ingredient with the biggest contribution. That also explains why greenhouse gas emissions and energy use results are not equivalent. In fact, results are similar except in what respects ingredients' impact (I) for Feed 1. Therefore, a higher energy use implies more emissions, but in the ingredient case there are other sources that change the results. The average feed is better in terms of heavy metals, since impact comes mainly from ingredient production. In this theme, however, Feed 2 is still better than Feed 1. All these results are shown in Figure 55 to Figure 55.

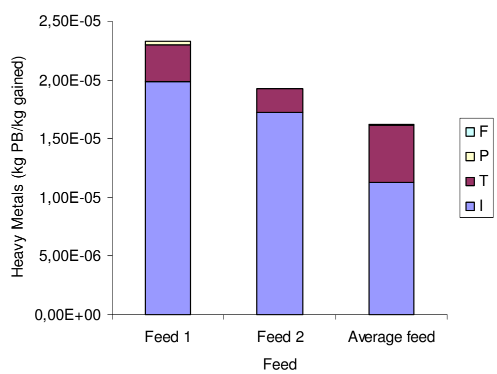
Figure 55, Figure 56, Figure 57, Figure 58 and Figure 59 – Impact per theme of each feed per kg of weight gained by the animal.







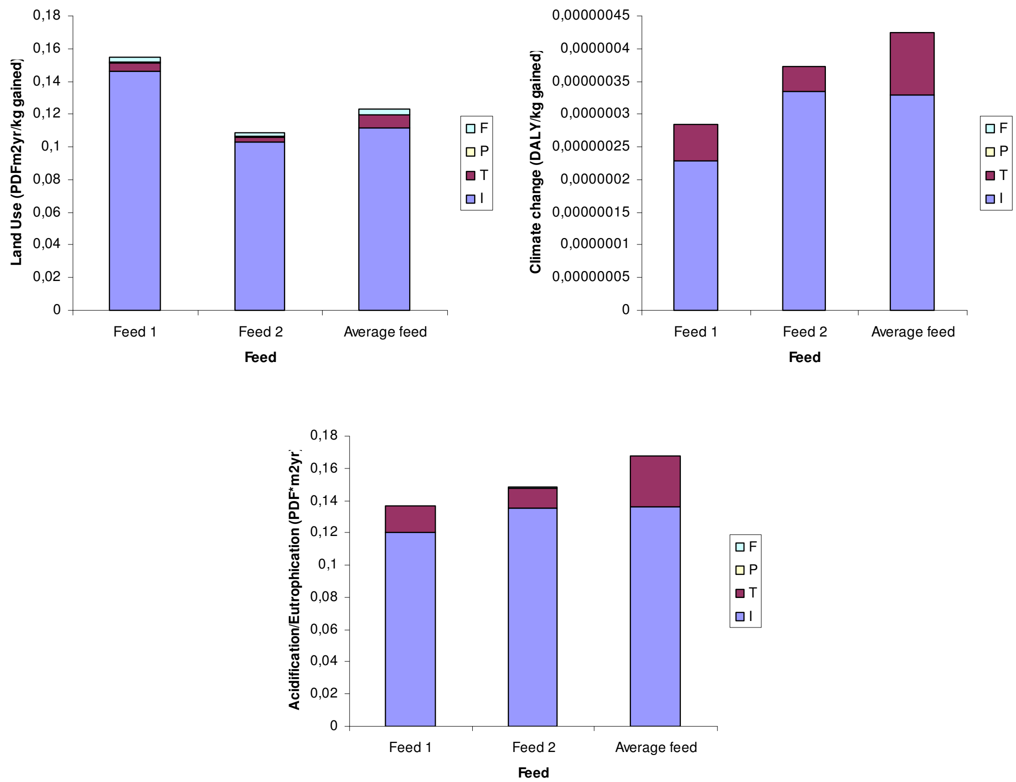




Life cycle assessment themes – eutrophication (top left), greenhouse gases' emissions (top right), acidification (middle left), energy resources spent (middle right) and heavy metals (bottom). I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Themes contributing for Ecoindicator 99 were also analysed, and it was determined that Feed 1 uses more land than Feed 2 (since silage maize requires little space), but contributes less to climate change, acidification and eutrophication. The average feed is in every case the worst option, as shown in Figure 60 to Figure 60.

Figure 60, Figure 61 and Figure 62 – Impact per theme of each feed per kg of weight gained by the animal.

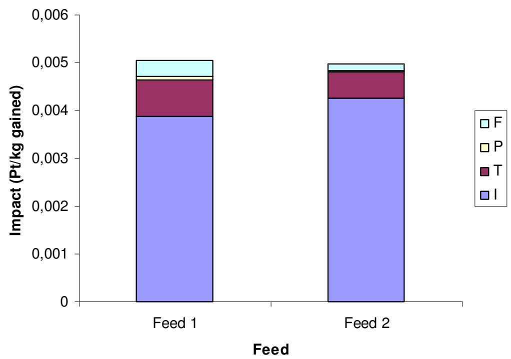


Life cycle assessment themes – land use (on the top left), climate change (on the top right) and acidification/eutrophication (on the bottom).
I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Of the above, land use is of special relevance in agriculture. It may be seen that Feed 1 clearly requires more land, while Feed 2 is the one that requires the less. If we only consider the part due to ingredient production, the average feed has almost the same land use as Feed 2. The results are due to the fact that large productions of silage maize require small amounts of land, even though inputs are small for the productivity, and therefore silage is usually related to extensive systems. This double advantage makes silage maize a very sustainable option.

Therefore, it may be concluded that Feed 2 is a better option for feeds. However, it should be noticed that, in Feed 1, only 90% of the ingredients are considered, and the results are normalized for 100%. Since the other 20% are mainly by-products, Feed 1 may have its impact over-estimated, since it is not to expect that those by-products weigh as much as products to which all of the impact is allocated. If no normalization is considered, and therefore only 90% of the impact is considered, the results are shown in Figure 63.

Figure 63 – Overall impact of Feeds 1 and 2 (Pt.kg^{-1} gained), Ecoindicator 95, only 80% of Feed 1 considered.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

This means that both feeds are probably more equal in impact than it seems if the known composition of Feed 1 is considered the whole composition; however, the fact is that, even considering only 90% of its composition, Feed 1 is already worse than Feed 2. The same process cannot be used for the average feed, since it is not known if the 35% left out are by-products or not.

Uncertainty analysis

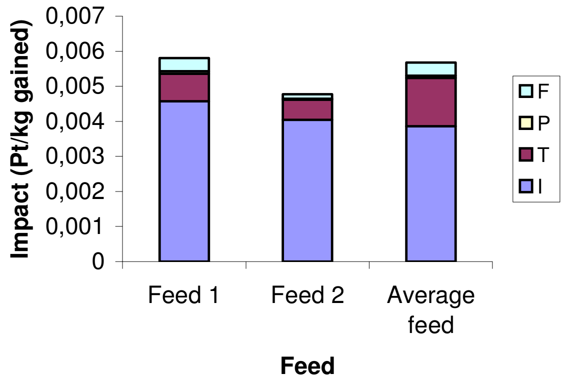
The possibility of silage maize transportation

Silage maize is usually not transported, but these results raise the question of, if transported, how far is still better to use a silage-based feed. Considering that transportation occurs by road, and given the single score differences in Ecoindicator 95, one may note that Feed 2 has the same environmental performance as Feed 1 if silage maize is produced at a distance of 190 km of the animal farm. To be just the same as the average feed, silage may be transported from 200 km. Therefore, even though results from Ecoindicator 95's impact assessment seem very similar, the fact is that the consideration of the transportation of silage maize shows that its use when produced in the same farm where animals grow is the best option. In Ecoindicator 99, for Feed 2 to have the same impact as Feed 1, silage would have to be transported 150 km, and in relation to the average it would still pay off to transport silage from 320 km.

Possibility of silage maize production enhancement

Another important analysis is, since silage maize is usually produced locally or nearby, if the production was controlled so that it could have an environmental performance that, for example, would make nitrate leaching, in time, null, through optimised irrigation practices and direct seeding. In that case, results would be as shown in Figure 64.

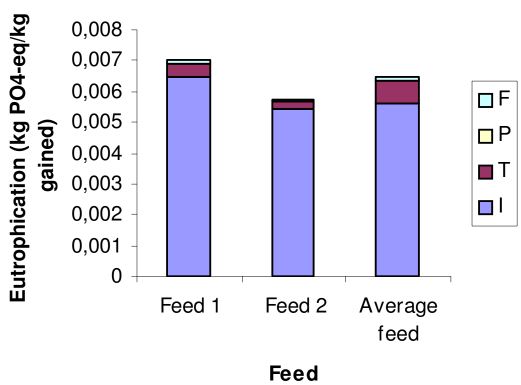
Figure 64 – Overall impact of each feed (Pt.kg-1 gained), Ecoindicator 95, no leaching in silage and no-tillage.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

However, in the case of eutrophication, differences would be very significant, as shown in Figure 65.

Figure 65 – Eutrophication contribute of each feed (Pt.kg-1 gained), Ecoindicator 95, no leaching in silage.

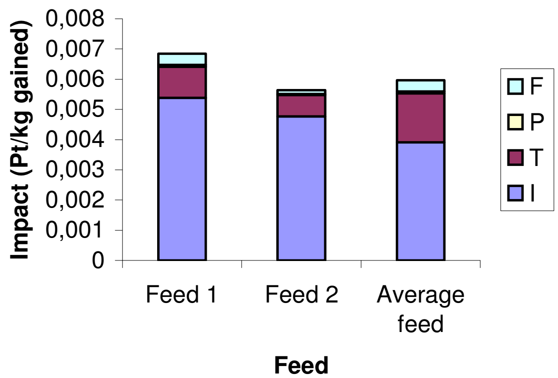


I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

The impact of non-optimized feeds

It is also important to notice that, in this paper, the most appropriate places for each ingredient to be produced (when choice was available) were considered for calculations, but this scenario is very unlikely, since no beef cattle producer can possibly assure that the commercial feeds he buys are composed by the best possible environmental options, and therefore it is not possible to legislate that agriculturalist's activity. The only ingredient that is not processed is silage maize, and therefore that is the only one where some control may exist. Given that, a new analysis was made (with normalized values), this time considering that ingredients are produced in the most common regions, which means that part of grain maize was considered to come from Argentina, and wheat was considered not to be irrigated (since usually is not). However, to guarantee a fair comparison, fertilization corrections in maize and soy were kept. New results are shown in Figure 66.

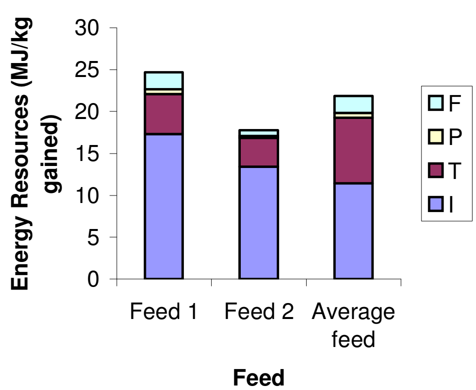
Figure 66 – Overall impact of all feeds ($Pt.kg^{-1}$ gained), Ecoindicator 95.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

These results translate the fact that the impact of the feed with the highest content in maize and wheat (Feed 1) is the most altered by the zone optimization, and therefore it further increases the distance between them. Feed 1 is now almost 20% worst than Feed 2, and more than 10% worst than the average. Feed 2 is the best option, but only by 5% from the average. This confirms that the average feed, since it is mainly composed with by-products, is a good environmental option. Still, since in all the results a market value allocation method was used, this feed is very dependent on prices fluctuations. Furthermore, in particular categories, like those which directly reflect transportation, as energy resources, the average feed is much worse than Feed 2, as shown in Figure 67.

Figure 67 – Overall impact of all feeds in energy resources ($MJ.kg^{-1}$ gained).



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

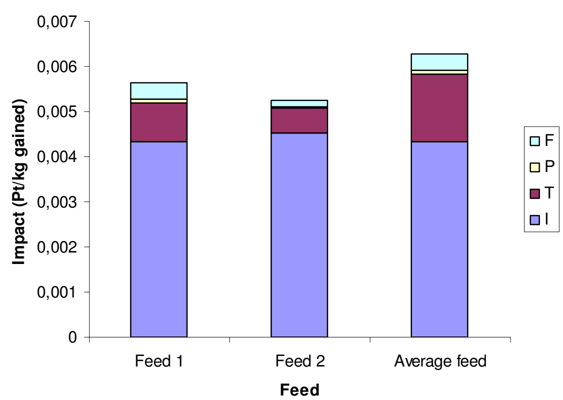
The impact of transportation

Another important analysis is whether the choice of truck for transportation is relevant. If transportation in a 3.5 t van (instead of a 16 t truck) was considered, then the overall impact of Feed 1 rise 11%, for Feed 2 6% and for the average feed 4%, thus aggravating the impact of transportation by far, especially in the case of Feed 1, where all ingredients are transported and most of them by road. Then, it is important that transportation occurs in the largest capacity truck possible.

The impact of allocation choice

Finally, since feeds contain a high percentage of by-products (INE, http://www.ine.pt, 2005), it is important to determine how impact should be allocated. Sometimes, allocation changes results more than any other parameter, as shown by Kim and Dale (2002). In this paper we used economic value, but to determine if that choice influences results an analysis using mass allocation was done, and results are shown for Ecoindicator 95 in Figure 68. The main difference is that mass impact allocation is greater than economic value allocation for soybeans and corn gluten feed (Table 88). Since the feed that used them the most is the average, it increases its overall impact the most. Notice that the impact of ingredients becomes about the same in every case, and therefore Feed 2 has the advantage because it requires less transportation.

Figure 68 - Overall impact of all feeds (Pt.kg^{-1} gained), Ecoindicator 95.



I – impact of the production of ingredients; T – impact of the transportation of ingredients; P – impact of the feed production; F – impact of the transportation of the feed to the animal farm.

Water use

Since water use is not considered in SimaPro's life cycle analysis, we determined how much water each crop, and therefore each feed, spends. Water uses for industrial processing of both crop and aggregated feeds have also been considered.

Feed 1 and Feed 2 require more total water than the average, as shown in Figure 69. The average feed uses much soybean meal and corn gluten feed, and to each of those only a given percentage may be attributed. Figure 70 shows the unit consumptions of each ingredient.

Figure 69 – Water consumption ($m^3.day^{-1}$) in the composition of each feed.

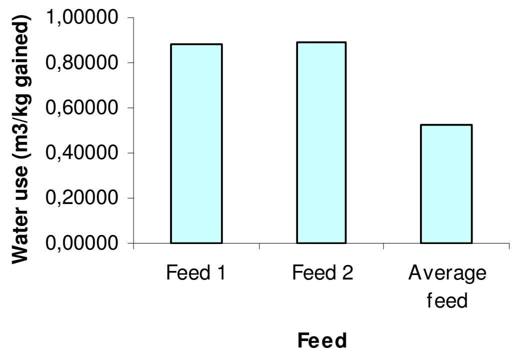
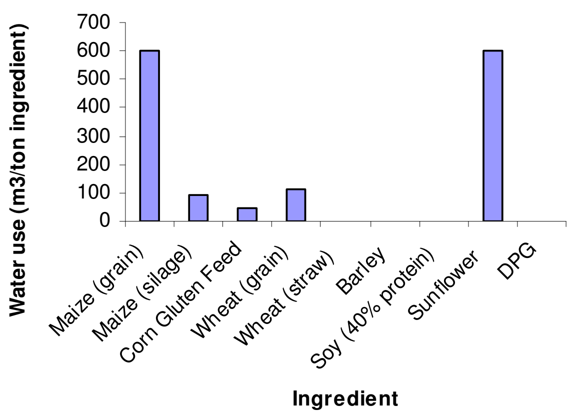


Figure 70 – Water used to produce 1 t of each ingredient.



If irrigated wheat wasn't used, the environmental impact of the feeds would be greater, but less water would be used. In fact, large uses of water when applied incorrectly may be very inefficient, since they favour the drench of soils and consequent tendency for aggravated leaching.

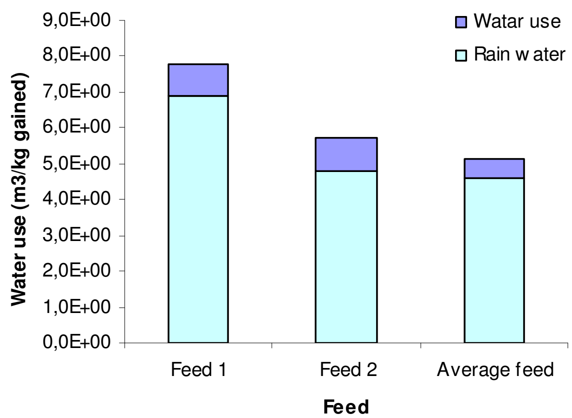
It should be noticed that this analysis is incomplete, since no irrigation data for soybeans was found. If that had been considered, Feed 2 would have a higher water consumption than Feed 1, since it uses more soy. The average feed would surely show a water consumption equivalent to that of the other feeds. Silage irrigation is also uncertain, since the value used in the original grain maize sheet was used. However, it is known that silage maize requires less water and is irrigated fewer times, since it is harvested earlier than grain.

To analyse the importance of both uncertainties, it was considered that soy requires the same level of fertilization that sunflower. Results shown that silage maize would have to be irrigated 50% less than grain for total water consumption to be the same as in Feed 1.

However, according to IDRHa[78], silage maize only requires less 10% irrigation water than grain maize, and therefore Feed 2 surely requires more water than Feed 1.

In order to determine the rain water that the crop receives, we converted land use into water, considering the average precipitation. This does not significantly change results, as shown in Figure 71. Even though Feed 2 requires more water, the difference is not sufficient to compensate the excess of space occupation by the ingredients in Feed 1, since Feed 2 requires less water, which means ingredients receive less rain water.

Figure 71 – Water use due to irrigation and estimated rain water.



To incorporate this global water use in life cycle analysis, we converted irrigation water to direct occupation of space. Then, we converted the new value to land use units, and from there to Ecoindicator 99 single score value. The conversion factors were obtained by dividing the corresponding SimaPro results. Table 98 shows that the aggregated value only changes in about 1%. Feed 2 is still the one with the lower environmental impact, even though more water is spent in producing its ingredients.

Table 98 – Ecoindicator 99 single score value considering water use.

		Feed 1	Feed 2	Average feed
Water use	$m^3.kg^{-1}$	8.83E-01	8.88E-01	5.14E-01
Occupation of space equivalent to water use	$ha.kg^{-1}$	9.90E-05	1.00E-04	5.60E-05
1 $PDF.m^2.year$	Ha	4.92E-03	4.76E-03	3.95E-03
Land use equivalent to water use	$PDF.m^2.year.kg^{-1}$	2.00E-02	2.09E-02	1.42E-02
1 $PDF.m^2.year$	Pt	7.80E-02	7.80E-02	7.80E-02
Single score equivalent to water use	$Pt.kg^{-1}$	1.56E-03	1.63E-03	1.11E-03
Single Score (without irrigation water)	$Pt.kg^{-1}$	1.61E-01	1.44E-01	1.83E-01
Single Score (with irrigation water)	$Pt.kg^{-1}$	1.63E-01	1.46E-01	1.84E-01

Uncertainty Analysis

Performing an uncertainty analysis for single score results in Ecoindicator 95, results were compared between feeds, and are as shown in Table 77. Results are obtained

[78] http://www.idrha.min-agricultura.pt/hidrologia/necessidades/inec.htm

through a Monte Carlo analysis with SimaPro, where random numbers are generated to determine the parameters in the uncertainty dominium of all values in the database used.

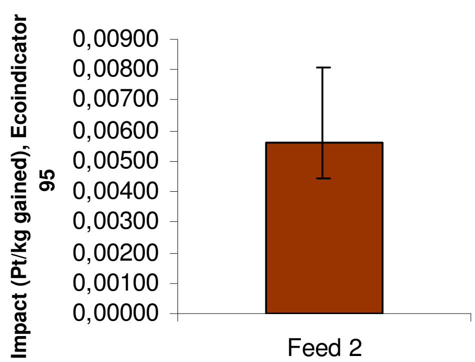
Table 99 – Sensibility analysis' results for all feeds; indicated as a percentage are the number of times impact of A > impact of B (Ecoindicator 95).

B \ A	Feed 1	Feed 2	Average feed
Feed 1		0%	81%
Feed 2	100%		100%
Average feed	19%	0%	

It was then determined that Feed 2 has, in fact, always a lower impact than Feed 1. The same happens in particular categories, except greenhouse (99% less impact for Feed 1) and winter smog (99% less impact for Feed 2). The average feed is relatively worse than Feed 2 in 100% of the cases, and worse than Feed 1 in 78% of the cases.

The variability within each value of feed's impact is illustrated in Figure 47, in the case of Feed 2. It is visible that the error associated is not symmetric in relation to the average, and the average value does not coincide with the value obtained through calculations with the given parameters (Teixeira *et al.,* 2005).

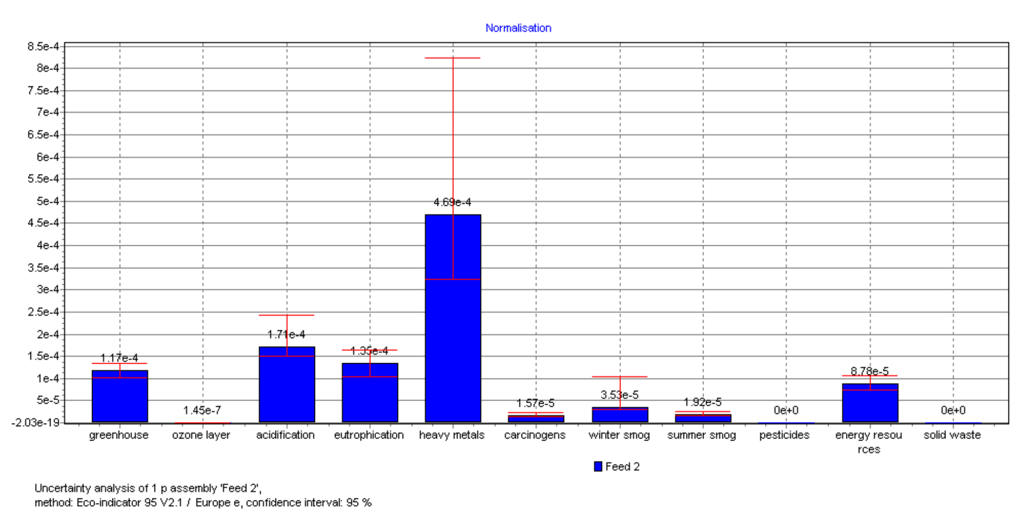
Figure 72 – Overall results, with the error bars indicated.



It should be noticed that crop fact sheets do not have any uncertainty data. Thus, all inputs therein have not been considered to vary. One, however, of particular importance is productivity, which may be very different between regions, in the same zone or even depending on the climatic characteristics of the year. We incorporated that variability in the results above by observing time series of average productivity values for each region (INE, http://www.ine.pt, 2005). Since the values in the fact sheets are not necessarily the average values, the intervals of variation could not be thoroughly defined. Still, values of 10% above the normal for a good year and 30% below for a bad one seem plausible from data observed (maintaining all inputs constant). These were introduced in the analysis with a triangle distribution of probability. Only ingredients produced in Portugal were considered.

In this analysis, heavy metals and winter smog contribute the most to the uncertainty of the final result, as may be seen in Figure 73. As for heavy metals, the biggest contributions to emissions come from fertilizers (uncertainty not considered) and agricultural machinery. Therefore, the most uncertain parameters are the parcel of the field cultivated and the portion of the machine spent in the work hours. The first varies as referred above, with a triangle distribution of probability. For the second one the SimaPro lognormal distribution was used, with a standard deviation square (σ^2) of 1.11. Correspondent emissions of cadmium, lead and zinc also present a lognormal distribution, σ^2 of 1.52, as others like nickel, copper or cadmium, with σ^2 of 5.05. As for winter smog, the largest contributions come from fertilizers and transoceanic transportation. The latter presents a lognormal distribution of probability with σ^2 of 1.5.

Figure 73 – Uncertainty in each theme of Ecoindicator 95 analysis after Single Score normalization, for Feed 2.



This analysis was also the criteria for results' precision. Since uncertainty is proportionally high for all themes, and particularly for the aggregated single score as shown in Figure 47, only two significant algorisms were used.

The choice of functional unit

In order to assess the nutritional equivalence between feeds, the aggregated crude protein content and digestible energy of each were determined. The composition of each ingredient is as shown in Table 100 (Stanton, 2004).

Table 100 – Dry matter, crude protein, digestible energy, net energy for growth and maintenance and crude fibre of each ingredient.

Ingredient	DM (%)	CP (%)	DE ($Mcal.kg^{-1}$ DM)	NE_m ($Mcal.kg^{-1}$ DM)	NE_g ($Mcal.kg^{-1}$ DM)	CF (%)
Maize (silage)	26	8	2.73	1.50	0.88	26
Maize (grain)	92	10	3.92	2.16	1.48	3
Corn Gluten Feed	90	26	3.62	1.94	1.30	9
Wheat (grain)	89	13	3.92	2.16	1.48	3
Barley	89	12	3.66	1.96	1.32	6
Soy (40% protein)	89	50	3.70	2.01	1.34	6
Sunflower	93	50	2.87			
Maize (silage)	26	8	2.73	1.43	0.82	12
Wheat (straw)	88	4	1.94	0.95	0.02	42

DM – Dry Matter; CP - Crude Protein; DE - Digestible Energy; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth; CF - Crude Fibre.

Regarding each feed's composition, their characteristics are shown in Table 101.

Table 101 – Dry matter, crude protein, digestible energy, net energy for growth and maintenance and crude fibre of each feed.

Feed	Mass (kg/4.8 months)	DM (%)	CP (%)	DE ($Mcal.kg^{-1}$ DM)	NE_m ($Mcal.kg^{-1}$ DM)	NE_g ($Mcal.kg^{-1}$ DM)	CF (%)
Feed 1	904.32	84.2	20.4	3.72	2.01	1.35	6.0
Feed 2	1076.54	51.8	17.3	3.30	1.79	1.11	14.9
Average feed	897.12	89.4	27.5	3.73	2.02	1.36	6.4

DM – Dry Matter; CP - Crude Protein; DE - Digestible Energy; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth; CF - Crude Fibre.

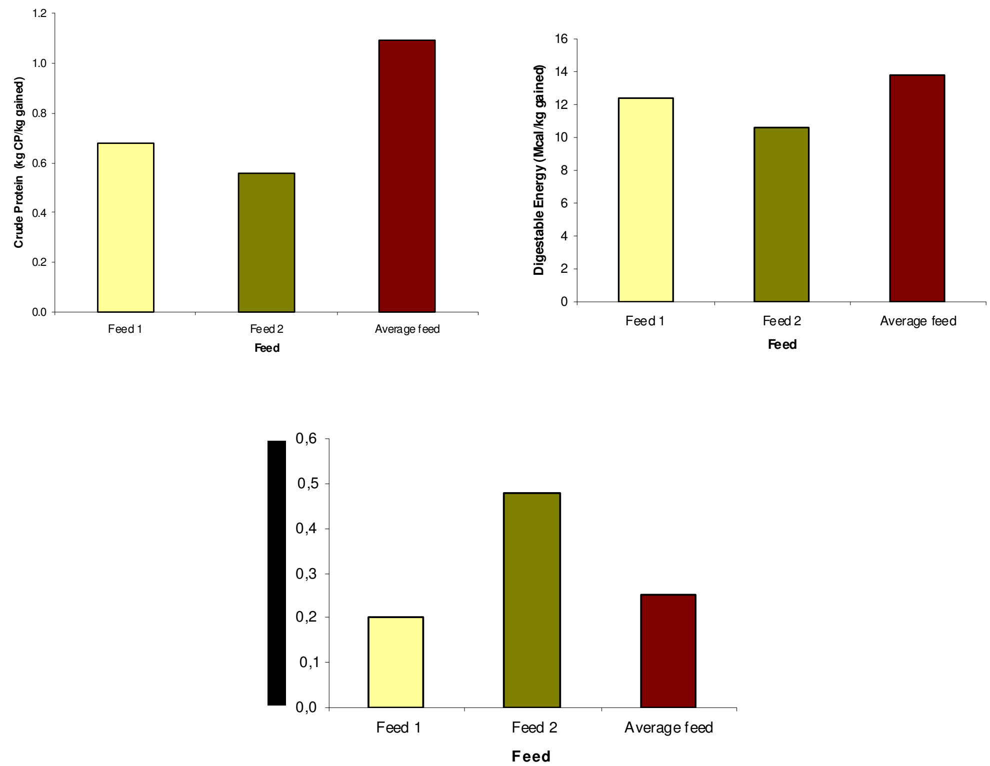
However, as for impacts, values should be compared in relation to the weight gained by the animals during the 4.8 months period. Therefore, values for comparison are shown in Table 102 and illustrated in Figures 74 to Figures 74.

Table 102 – Crude protein, digestible energy, net energy for growth and maintenance and crude fibre of each feed in each kg of weight gained by the animal.

Feed	CP (kg $CP.kg^{-1}$ gained)	DE ($Mcal.kg^{-1}$ gained)	NE_m ($Mcal.kg^{-1}$ gained)	NE_g ($Mcal.kg^{-1}$ gained)	CF (kg $CF.kg^{-1}$ gained)
Feed 1	0.718	13.105	7.103	4.764	0.211
Feed 2	0.559	10.660	5.789	3.597	0.480
Average feed	1.019	13.847	7.507	5.064	0.238

CP - Crude Protein; DE - Digestible Energy; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth; CF - Crude Fibre.

Figures 74, 75 and 76 – Crude protein, digestible energy and crude fibre of each feed.



From here, it may be concluded that Feed 2 has a lower protein value than Feed 1 and even the average, but higher fibre content.

From the impact (I + T) of each ingredient, the impact of each unit of protein (Figure 77), energy (Figure 78) or fibre (Figure 79) may be determined, and therefore assess which ingredients are better used for each purpose. Corn gluten feed has a small impact in all of them. Other ingredients with low impacts are: soy and sunflower for each unit of crude protein; silage maize, straw and sunflower for fibre; straw for digestible energy.

Figure 77 – Impact of each unit of crude protein in each ingredient analysed (Single score, Ecoindicator 95).

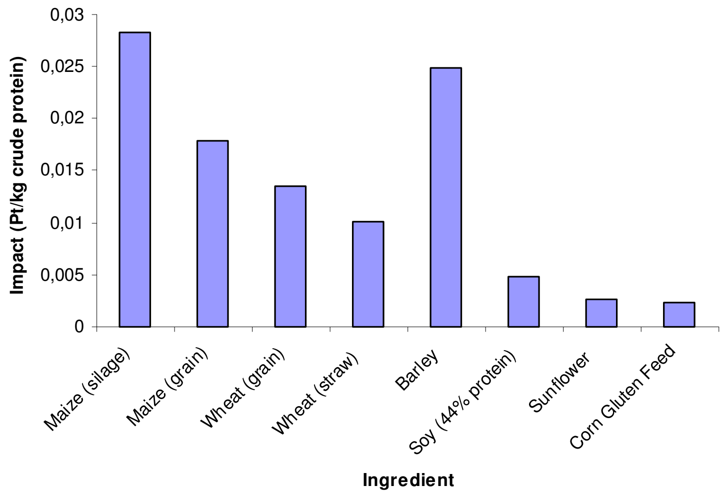


Figure 78 - Impact of each unit of crude fibre in each ingredient analysed (Single score, Ecoindicator 95).

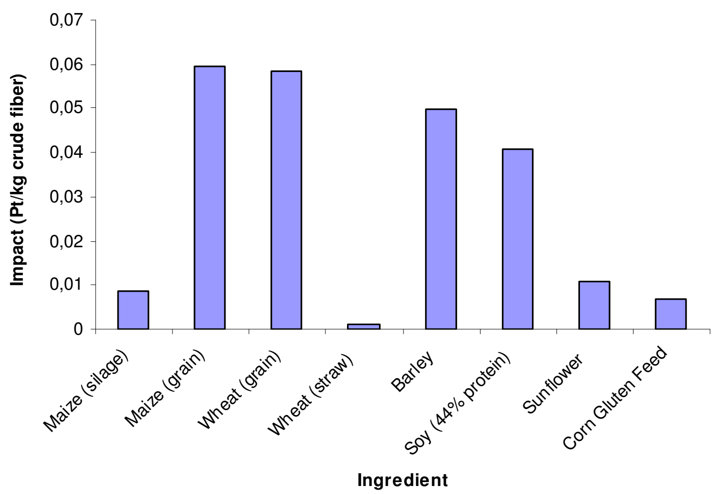
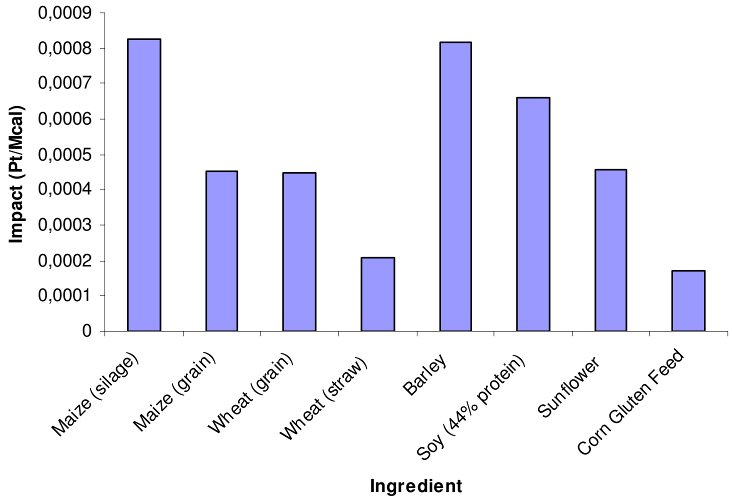


Figure 79 - Impact of each unit of digestible energy in each ingredient analysed (Single score, Ecoindicator 95).



It is also important to determine if the analysed feeds satisfy the animals' needs. In this stage of development, in terms of protein, net energy for grown and maintenance beef calves require the amounts indicated in Table 103 (NRC, 1996). We considered the average weight in the middle of the total period (at 9.6 months), which corresponds to the average quantities in each parameter, as shown in Table 103. Comparing both, Feed 2 has about the necessary quantities of protein and energy, but Feed 1 and the average feed exceed by far that number. For example, the animals use for maintenance 5.55 $Mcal.day^{-1}$, but both feeds provide about the double.

Table 103 – Feed composition and requirements in protein and energy for maintenance and growth.

Feed	CP ($g.day^{-1}$)	NE_m ($Mcal.day^{-1}$)	NE_g ($Mcal.day^{-1}$)	Average weight (kg)
Feed 1	1076.9	10.66	7.15	298
Feed 2	670.2	6.95	4.32	276.4
Average feed	1529.2	11.26	7.60	298
Requirement for maintenance and 1.2 $kg.day^{-1}$ growth	614	5.2	4.22	275
Requirement for maintenance and 1.5 $kg.day^{-1}$ growth	716	5.55	5.74	300

CP - Crude Protein; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth.

Therefore, in the analyzed categories, by-products seem good options regarding its inclusion in feeds, since the environmental impact attributed to them is smaller, and they still provide the protein, fibre or energy needed for animal growth (as shown by Feed 1 and the average feed). However, they are insufficient in other parameters, as for example in what respects to amino acids (Alfredo Sendim; personal communication). For example, while grain maize has an overall 4.49% content of 12 amino acids (Church and Pond, 1988), corn gluten feed has 0.94% of 4 amino acids (Schroeder, 2004).

Substitution between soy and alfalfa

Since soybeans are one of the ingredients with the highest environmental impact, and furthermore are usually imported from Argentina and often transgenic, their substitution by dehydrated alfalfa was studied. Alfalfa and soybeans have different contents in protein, fibre and energy, but since both play in the feed the role of main protein source, the equivalence was determined regarding that parameter.

Dehydrated alfalfa is obtained from drying of the collected and conditioned fresh alfalfa (*Medicago Sativa* L.), by forced ventilation (Infoagro[79]). The nutritional characteristics of the product remain roughly the same, but the dry matter content is much higher, as shown in Table 104 (Stanton, 2004). In this paper, the impact of fresh alfalfa was determined, and, since at that stage the dry matter content is 26%, it was considered that alfalfa was dried until its dry matter content was 92% (which means that for each ton of dehydrated alfalfa there were 92/26 tons of fresh alfalfa produced). The impact of this final product was used for the comparison.

Table 104 – Nutritional characteristics of alfalfa and soybean meal.

Feedstuff	DM (%)	CP (%)	CF (%)	DE (Mcal.kg^{-1})	NE_m (Mcal.kg^{-1})	NE_g (Mcal.kg^{-1})
Alfalfa dehydrated 17%	92	19	26	0.55	0.28	0.14
Alfalfa fresh	26	19	27	0.54	0.27	0.14
Soybean meal solvent 44% protein	89	50	6	0.76	0.41	0.28

DM – Dry Matter; CP - Crude Protein; CF - Crude Fibre; DE - Digestible Energy; NE_m - Net Energy for maintenance; NE_g - Net Energy for growth.

Table 104 also shows that for the protein content to be about the same in the feed there should be about 2.5 times more alfalfa than there would be soy.

Alfalfa usually is imported from Spain (IACA, 2004), and the region that produces the most quantity is Aragón, from which a transportation distance of 600 km by road was estimated. Therefore, it may be concluded that, using the single score value of Ecoindicator 95 an equivalent amount of each (in protein), alfalfa and soy have about the same impact (both before and after transportation), but in Ecoindicator 99 soybeans have a much larger impact, as shown in Table 105.

Table 105 – Single score (Pt) impact in Ecoindicators 95 and 99 for production and transport of alfalfa and soybeans.

	Dehydrated alfalfa (Pt.t^{-1})	Soybeans (Pt.t^{-1})
Ecoindicator 95	1.7	1.7
Ecoindicator 99	31.3	54.4
Ecoindicator 95 + Transport	2.2	2.2
Ecoindicator 99 + Transport	50.1	72.6

Since alfalfa drying usually involves electricity consumption, which in this case was not considered, its impact is underestimated. Still, these results show that alfalfa is a valid option relatively to soybeans, since its environmental impact is equal or inferior, and it has the additional advantage of being produced in Spain, where correct production techniques and food safety are easier to guarantee. It should also be noted that alfalfa has a much higher crude fibre content than soybeans, and so changing these ingredients has necessarily implications on the remaining content of the feed.

[79] Infoagro, 2005. http://www.infoagro.com.

Another valid option for soy substitution is the use of peas. In this paper such option was not considered, since Portugal mainly imports peas from France and no information could be gathered. Peas capture more nitrogen than soy (requiring lesser fertilization) (MADRP[80]), but pea fields have greater levels of nitrate leaching (van der Werf *et al.,* 2005).

Conclusions

In this paper we analysed different options regarding animal feeds, optimizing the alternatives and verifying whether they would still constitute a nutritionally balanced feed.

From the results obtained it may be concluded that the use of silage maize is an environmental adequate option. Even though feeds consisting of silage maize have to be composed of a high percentage (in mass) of such ingredient, its impact is significantly reduced, especially when compared with the alternative, which is a grain maize-based feed.

The silage-based feed is also environmentally better when compared to the average national feed, and transportation becomes less relevant since silage maize is usually grown nearby or even in the same farm where it is given to animals. In ecological terms, since in this case the unit impact is small, this choice seems plausible. However, even if silage maize is transported, the feed that it constitutes is still better for distances up to 160 to 230 km, depending on the analysis. This means that the use of silage presents large advantages at the farm (less impact in every theme per unit of mass produced), and that clearly overcomes the impact of an eventual transportation.

The most important analysis in which silage maize is not the environmentally better feed is water use. However, to understand if that disadvantage could significantly change the life cycle analysis, we transformed that water in land use. Even though such analysis is only an approximation, it allowed noticing that water use values are important in themselves but, when aggregated, their influence is not enough to change the global impact.

Furthermore, since silage maize is not processed industrially with the remaining feed, its impact is also smaller in this part of the life cycle. However, the assembling of the aggregated feed has a very small overall impact (1%), and production of the ingredients accounts for about 80% of global impact.

But the most important characteristic of silage maize is that the fact that it is produced nearby where it is given to animals, or even in the same farm, makes it easier to control whether its production practices and techniques are those with a least environmental impact. When a farmer buys an average commercial feed, ingredients come from many places and countries (not only these optimised places considered), and therefore the only variable that may be easily controlled through legislation is silage maize.

The fact that the average feed is also a good option is a direct consequence of its large use of by-products. By-products are a valid option for animal feed, since they are usually residues of other industrial processes, and therefore the impact allocation based on

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[80] Ministério da Agricultura, do Desenvolvimento Rural e das Pescas (Portuguese Ministry of Agriculture), http://www.min-agricultura.pt.

market value is always low, even though large quantities of by-product are produced, and therefore a mass allocation would attribute then a larger impact (for example, corn gluten feed - Table 88). However, by-products are deficient in essential amino acids. Furthermore, usually in Portugal by-products are imported, which means that transportation becomes the largest impact in the process, with large contribution to specific themes, like greenhouse of energy use.

Still, the results obtained indicate the strong possibility that the environmentally best feed would be an alternative to all the studied feeds composed by both silage maize and by-products. Such a feed would avoid the direct use of all other crops produced specifically for this end, since those present the highest environmental impact. This option would, however, have to respect all criteria of healthy animal nutrition, and therefore this paper suggests the determination of its exact composition as further work.

Since the goal of this paper is not a single score objective classification of the impact in a given production system or feed, but the comparison between feeds, the uncertainty analysis were very important, since they validated the assumptions and simplifications made. By comparing feeds in similar conditions, the analysis assured that the results obtained always stand.

In what respects to each ingredient, the ones with the highest impact are soybeans and common barley. In the case of soybeans, used as protein sources, a good substitution option must be found. Alfalfa has a slightly higher impact, but is not transgenic, and therefore is a valid ingredient. As for common barley, it may be concluded that producing the cereal for animal feeds has a great impact, but if by-products from other barley products are used the overall impact of the feed may be favoured. Irrigated grain wheat has a low impact, even though in Portugal rainfed cultures are the current trend. Therefore, the trade-off between environmental impacts and water use are of great importance to decide between options. That justifies the method used in this paper for water use internalization on the impact of aggregated feeds. Still, in the case of wheat, we did not consider foreign importation options, because of lack of information. Rainfed production could possibly have a lower impact there, even though transportation would have to be considered. If wheat was imported from France, it would probably be transported by road, and so the global impact could rise considerably.

This paper also did not consider the economic evaluation of the optimization, but such is a very relevant study that may determine which of the trade-offs here suggested pay off. We also did not consider alternative occupations of space, since life cycle assessment methods used in this paper assume that choices do not change the whole agricultural sector enough for such effect to be relevant. LCA results have a very low special and temporal resolution, and do not regard social and economical aspects, as noted by Owens (1997) and Udo de Haes (2004).

Still it is important to notice that because of price regulation, if farmers decide to produce silage maize and use Feed 2, other ingredients this feed uses less or not at all would be produced in smaller amounts. The real environmental impact is dependent on which alternative use of space land formerly used to produce the other cereals and oilseeds used in traditional feeds would have. Furthermore, by-products are often residues from other industries, and if they are not used in feeds they must be given a different end. Real environmental impact comes not only from the direct impact here studied, but also from all other impacts, positive or negative, that choices imply. This means opportunity costs must always be addressed, as noted by Jungk *et al.* (2002) and Manbiot (2004).

Appendix V – Economic balances for steer production in SBPPRL

We show next the calculations for the SBPPRL economic balance[81] in each scenario (with and without support from the PCF, with and without installation support, and for each stocking rate).

Sowing in 2009

Table 106 – Economic balance (values in €.ha^{-1}); steer sold for 250 €, sowing in 2009.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-252	-213	-161	-110	-32	98
	Installation with support	-233	-195	-143	-91	-13	116
With PCF	Installation without support	-235	-196	-144	-92	-14	115
	Installation with support	-216	-177	-125	-73	4	134

Table 107 – Economic balance (values in €.ha^{-1}); steer sold for 375 €, sowing in 2009.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-233	-176	-99	-22	93	285
	Installation with support	-215	-157	-80	-4	112	304
With PCF	Installation without support	-216	-158	-81	-5	111	303
	Installation with support	-197	-139	-63	14	129	321

Table 108 – Economic balance (values in €.ha^{-1}); steer sold for 500 €, sowing in 2009.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-215	-138	-36	65	218	473
	Installation with support	-196	-120	-18	84	237	491
With PCF	Installation without support	-197	-121	-19	83	236	490
	Installation with support	-178	-102	0	102	254	509

Sowing in 2010

Table 109 – Economic balance (values in €.ha^{-1}); steer sold for 250 €, sowing in 2010.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-252	-213	-161	-110	-32	98
	Installation with support	-233	-195	-143	-91	-13	116
With PCF	Installation without support	-240	-201	-149	-98	-20	110
	Installation with support	-221	-183	-131	-79	-1	128

[81] If the value is negative, the costs are higher than the revenue, and if it is possible revenue is higher than costs.

Table 110 – Economic balance (values in €.ha^{-1}); steer sold for 375 €, sowing in 2010.

Scenario/Stocking rate (LU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-233	-176	-99	-22	93	285
	Installation with support	-215	-157	-80	-4	112	304
With PCF	Installation without support	-221	-164	-87	-10	105	297
	Installation with support	-203	-145	-68	8	124	316

Table 111 – Economic balance (values in €.ha^{-1}); steer sold for 500 €, sowing in 2010.

Scenario/Stocking rate (CU.ha^{-1})		0.15	0.30	0.50	0.70	1.0	1.5
Without PCF	Installation without support	-215	-138	-36	65	218	473
	Installation with support	-196	-120	-18	84	237	491
With PCF	Installation without support	-203	-126	-24	77	230	485
	Installation with support	-184	-108	-6	96	249	503