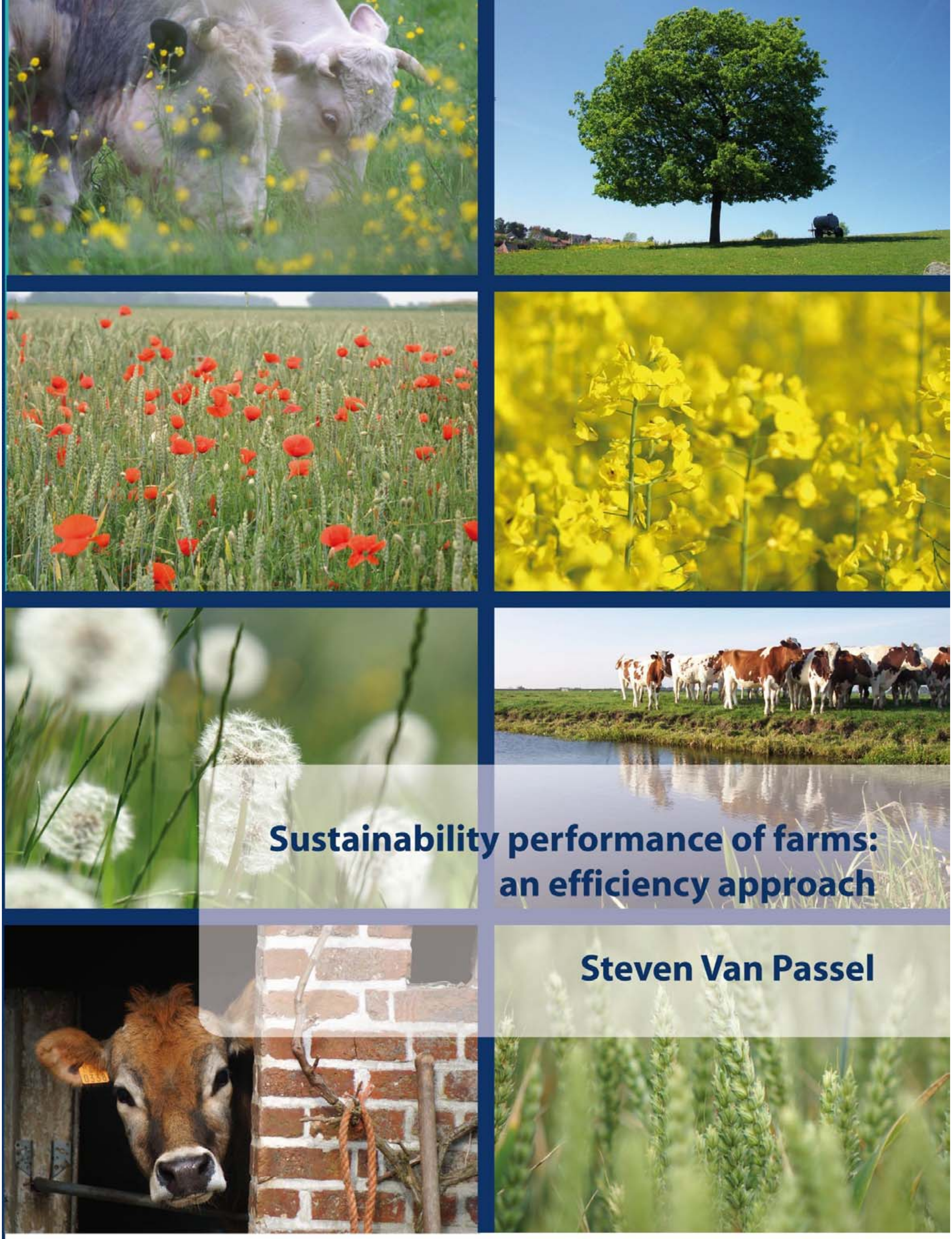

Sustainability performance of farms: an efficiency approach

Steven Van Passel

UNIVERSITEIT GENT

FACULTEIT BIO-INGENIEURSWETENSCHAPPEN

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Assessing Sustainability Performance of Farms: An Efficiency Approach

Thesis submitted in fulfillment of the requirements for the degree of Doctor (PhD) in Applied Biological Sciences

Dutch translation of the title:
Beoordeling van Duurzaamheidsprestaties van Landbouwbedrijven:
Een Efficiëntiebenadering

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Voorwoord / Preface

Alles kost moeite, anders was het de moeite niet waard
—

Kiezen is verliezen
—

Een koe leer je niet blaffen
—

Een expert is iemand die steeds meer weet over steeds minder
—

Twijfel is een eerbetoon aan de waarheid
—

This work would never have been possible without help and support of friends, colleagues and family.

(Diegene die deze zin niet voldoende vinden als voorwoord mogen gerust verder lezen)

Het voorwoord schrijven is ongetwijfeld de lastigste en meestal dan ook nog de laatste opdracht bij het maken van een doctoraat. Een dankwoord schrijven is altijd ondankbaar. Mensen persoonlijk vermelden en bedanken is eigenlijk vanzelfsprekend maar mensen vergeten te vermelden is ongepast, onbeleefd, dom, onverstandig, ... Daarom wil ik in de eerste plaats iedereen bedanken die ik hieronder vergeten ben te vermelden (en die natuurlijk verdienen vermeld te worden). Bedankt en sorry!

Ik heb veel kansen gekregen en daarvoor ben ik vele mensen dan ook zeer dankbaar. Vanuit de bosrand (de prachtige lokatie van het Steunpunt Duurzame Landbouw) heb ik mogen samenwerken met de Stedulezen (logisch toch?) en promotoren, de vakgroep landbouweconomie in Gent, de afdeling landbouw- en voedseleconomie in Leuven, het voormalige centrum voor landbouweconomie in Brussel (en daarna het ILVO in Merelbeke), de afdeling monitoring en studie (Ministerie van Landbouw en Visserij) en de Nederlanders van het LEI.

Allereerst, wil ik mijn promotoren persoonlijk bedanken: Guido en Erik, bedankt voor de vrijheid, steun en inspiratie bij het maken van dit doctoraat. Verder wil ik ook Frank (het Stedula opperhoofd), waar ik heel veel van heb geleerd, speciaal bedanken. Ook Jan heeft een belangrijke impact gehad in mijn eerste jaren als Stedula-onderzoeker: bedankt. Verder heb ik altijd graag samengewerkt met verschillende Stedulezen (en met sommigen nu nog): bedankt, Marijke, Rein, Elke, Annelies (x2), Karin, Joost, Ignace, Xavier en Luk. Verder wil ik ook alle L&M'ers bedanken die met mij sinds januari 2007 de nieuwe eenheid landbouw & maatschappij bevolken en mij dus al meer dan een half jaar hebben kunnen verdragen (hoop ik (meestal) toch). Ludwig wil ik hierbij speciaal bedanken voor de aangename samenwerking en de bilaterale gesprekken, dit al sinds de start van mijn onderzoeksloopbaan. Furthermore, I enjoyed writing the SVAPPAS project proposal and the first months of working with all the project members, special thanks to Frank, Tobias and Timo.

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Verder wil ik ook mijn ouders, schoonouders, broer en schoonbroers en hun huisdieren bedanken voor hun steun en advies en hun zorgen. Daarnaast wil ik al mijn vrienden bedanken voor al die uren ontspanning. Leve tennis, zeilen, badminton, squash, voetbal, bord games en al de rest. En op uitdrukkelijk verzoek zal ik J.(e)F. vermelden omdat hij zo lief was spellingsfouten en vooral vreemde zinsconstructies uit dit doctoraat te verwijderen.

Tot slot, geniet van het lezen van het doctoraat (of toch in ieder geval van Calvin & Hobbes op de volgende pagina) !


HEY! WHAT HAPPENED TO THE TREES HERE? WHO CLEARED OUT THE WOODS?
THERE USED TO BE LOTS OF ANIMALS IN THESE WOODS! NOW IT'S A MUD PIT!
THIS SIGN SAYS, "FUTURE SITE OF SHADY ACRES CONDOMINIUMS."
ANIMALS CAN'T AFFORD CONDOS!
"SHADY ACRES"? THE ONLY SHADE I SEE IS FROM THAT BULLDOZER.
WATTERSON
WHERE ARE ALL THE ANIMALS SUPPOSED TO LIVE NOW THAT THEY CUT DOWN THESE WOODS TO PUT IN HOUSES ??
BY GOLLY, HOW WOULD PEOPLE LIKE IT IF ANIMALS BULLDOZED A SUBURB AND PUT IN NEW TREES ?!?
WATTERSON
NO GOOD. THEY DIDN'T LEAVE THE KEYS.
IT TOOK HUNDREDS OF YEARS FOR THESE WOODS TO GROW, AND THEY LEVELED IT IN A WEEK. IT'S GONE.
AFTER THEY BUILD NEW HOUSES HERE, THEY'LL HAVE TO WIDEN THE ROADS AND PUT UP GAS STATIONS, AND PRETTY SOON THIS WHOLE AREA WILL JUST BE A BIG STRIP.
EVENTUALLY THERE WON'T BE A NICE SPOT LEFT ANYWHERE.
I WONDER IF YOU CAN REFUSE TO INHERIT THE WORLD.
I THINK IF YOU'RE BORN, IT'S TOO LATE.

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Assessing Sustainability Performance of Farms: An Efficiency Approach

Chapter 1

General introduction: problem statement and objectives

We have not inherited the world from our forefathers, we have borrowed it from our children

—

1.1 Towards a sustainable agriculture

Over the past twenty years,[1] the idea of *sustainable development* has come to the forefront of the scientific debate. Sustainability proved a remarkably difficult concept to define and use precisely. Moreover, real measurement of sustainability is fraught with difficulties of principles and practice, so there are understandably, though disappointingly, few published empirical papers (Pezzey and Toman, 2002*a*). The aim of this dissertation is to assess sustainability performance with a focus on agriculture.

Since the end of World War II, agriculture has undergone a radical transformation in industrial countries. Rapid technological change and the changing demands of consumers have led to the creation of an industrialized food and agricultural system with intensive forms of production. These intensive forms have typically been associated with high population densities, productive land,

[1] In 2007, the publication of *Our Common Future* (World Commission on Environment and Development, 1987) celebrates its twentieth anniversary. This report put forward the ideas of sustainable development for the first time in an international context

and rapid technological progress (Chavas, 2001). Agricultural sectors in most advanced economies have come under severe criticism for lacking the characteristics of *sustainability* (Hartridge and Pearce, 2001). The enormous structural changes in agriculture have raised food output, changed the nature of farm structure, and increased the environmental and social consequences of farming (Filson, 2004). In the past, agricultural policy has mostly been interested in productive aspects. Policy interventions aiming to promote agricultural modernization stimulated structural change resulting in several negative side-effects, such as increased pollution, landscape depletion and deepened regional disparities (Andreoli and Tellarini, 2000).

Nowadays, an important objective of European agricultural policy is to have a **sustainable, efficient** farming sector which uses safe, clean, environmentally-friendly production methods providing quality products to meet consumers' demand. The sustainability goal is thereby seen as a key element towards a profitable long-term future for farming and rural areas. Policy makers aim for strong economic performance hand in hand with the sustainable use of natural resources in the field of agriculture (European Commission, 2004, Fisher Boel, 2005).

These policy objectives are formulated as resounding words. However, several issues remain unclear and vague. The following questions may be raised: - What is a sustainable farming sector? - Can we measure it? - Is it self-evident that sustainable farmers are efficient farmers? - How can we make agricultural practices more sustainable? - Can we measure the progress towards sustainability?

Hence, without a clear framework to assess sustainability and without empirical work measuring, explaining and evaluating contributions towards sustainability, the policy objectives will remain hollow or empty sounding words.

1.2 Farming in Flanders

As all the empirical applications in this dissertation are using Flemish farm accounting data, we first provide a short overview of agricultural production in Flanders. Flanders is an interesting case because of its densely populated area and highly intensive agricultural structures. Although almost every agricultural structure is represented in Europe (Mann, 2006), several common characteristics of EU agriculture can still be described: (i) exit of labor from agriculture, (ii) changes in numbers of farms and their average size, (iii) the importance of family farms, (iv) the combination of farming with other activities, (v) family ownership of land, (vi) personal characteristics of farm managers (age, schooling, etc.).

Agricultural production in Flanders is dominated by intensive livestock production and horticulture. In addition, mixed farming combines cereals, industrial crops, horticulture and livestock farming. Flemish agriculture is confronted with declining economic importance, considerable fragmentation of the useable land, large income disparity, an ageing population with succession problems and pollution caused by intensive agricultural methods and by pressure from urban areas on the rural environment. However, there is a large diversity in agricultural production, a high specialization in livestock farming, horticulture and dairy production, an advantageous geographical location in the middle of Western Europe, and a well-developed transport infrastructure.

There are a lot of different farm types in Flanders and many farmers combine several agricultural activities (mixed farms). On the other hand, farms evolve toward more product specialization. The fact that most farms are multi-product firms suggests that their benefits are significant in agriculture. The first benefit is the presence of economies of scope reflecting the reduced cost associated with producing multiple outputs. The second benefit are the risk-reducing effects of diversification (Chavas, 2001).

The agricultural sector, including fishery and horticulture, has a share of 1.3% in total gross domestic product (GDP) in Flanders (2004). Only 2% of the working force works in the agricultural sector (Flemish Government, 2005). Hence, 2% of the working population creates only a share of 1.3% of the GDP, resulting in a productivity deficit. The share in GDP and in employment of agriculture declines year after year. This evolution is apparent in all member states of the European Union. The productivity deficit is even larger in several other EU countries. For example, in Poland where 17.6% of the working population creates only 2.9% of the GDP (Balmann, 2006). Contrarily, the agricultural sector is an important activity in rural areas. Even in a densely populated area as Flanders the agricultural sector uses about 46% of the area (Flemish Government, 2005).

Blandford and Hill (2005) argue that to provide insight into the sustainability of agriculture, the focus must be on the institutional units in which production takes place: the farms responsible for bringing together land, labor and capital. The number of Flemish farms (production units) has decreased from 70 000 in 1983 to 35 000 farms in 2004 (FOD, n.d.). Hence, in those 20 years the number of farms halved. As agricultural area in Flanders stayed more or less the same, the utilized land per farm doubled to nearly 18 hectares per farm. Furthermore, the utilized land per unit of labor also decreased during the observed period (1983-2004). However, the decrease in number of farms is higher than the decrease in labor. In particular, the number of small farms has decreased, while the number of large farms has even increased in Flanders. On average, Flemish farmers are old with no successor available on the farm and with a low education level. The average age of a Flemish farmer is 47.8 year (2004), only 13.7% of farms with a farm manager older than 50 year have a potential successor. In Flanders (2004), 58% of all farmers only have practical experience, 21% of all farmers followed a basic agricultural training while only 21% of all farmers followed a full agricultural training after their basic training (Flemish Government, 2005).

Besides social and economic aspects, also environmental aspects characterize current agricultural structures. Agriculture is a major user of natural resources (e.g., land, water, etc.) and one needs to maintain the quantity and quality of those resources in order to remain viable. Important environmental themes are: soil, water and air quality, use of pesticides, energy use, water use, nutrients emissions and biodiversity.

In Flanders (as in several other European regions), nutrient emissions are a major concern in agricultural practice. Inefficient use of nitrogen and phosphorus can result in potentially detrimental losses to the environment. Nevens et al. (2006) found a decrease of the average farm gate nitrogen surplus of Flemish specialized dairy farms between 1989 and 2001, but there is still room for further improvement of the nitrogen use efficiency in Flemish dairy farming (Nevens et al., 2006). Meul et al. (2007a) analyzed the changes in energy use and energy use efficiency, and they found for Flemish specialized dairy, arable and pig farms a decrease in total energy use per hectare between 1990 and 2000. Further, pesticide use (taking into account eco-toxicology) and greenhouse gas emissions from agriculture have decreased in Flanders during the period 1990-2004 (Flemish Government, 2005). In general, there has been an overall improvement in the environmental performance of agriculture, but this masks a number of severe local and regional problems, while future global pressures on land and water resources will be significant (OECD, 2004).

1.3 Objectives and outline

It is generally acknowledged nowadays that a sustainable farming sector is necessary. Moreover, sustainable development, including its economic, environmental and social elements, is a key goal of all decision makers (Islam et al., 2003). It is not only essential to recognize the importance of sustainability of agricultural systems but also the need to develop appropriate ways to measure sustainability (Tellarini and Caporali, 2000). Sustainability should be measured using a more comprehensive approach, taking into account the whole range of impacts caused by agriculture (Andreoli and Tellarini, 2000). In other words, it is important to develop and use tools capable of assessing farm performance taking into account all relevant impacts of farm activities. In fact, an integrated view, translated into well-defined methods and procedures for weighting economic, social and environmental aspects is necessary for policy development, political opinion formation and well-considered private and public action. Without them, it is difficult to say what is very good, good and not so good, beyond the simple situation where improvement is possible without cost (Hubbes and Ishikawa, 2007).

This dissertation consists of two major parts: a conceptual and theoretical framework (part I, p. 11 et seq) and an empirical analysis (part II, p. 111 et seq).

In the first part of the dissertation, an overview is given of the most common methods to measure performance under different assumptions and visions. In this way, we will try to partly solve the sustainability paradox between theory and practice. It is important to understand the several existing notions and related assumptions of sustainability before analyzing contributions towards sustainability in practice. The literature overview of this dissertation consists of two major parts: one about concepts and notions of sustainability and the other about measuring performance. In the past, performance was defined in terms of creating value added and return on economic capital. This view on performance can be broadened. In this broader view, performance is similar to sustainability: measuring sustainability is interpreted as measuring performance, all kind of returns and costs should be considered.

In the second part of the dissertation, several empirical applications about measuring performance and sustainability are presented. Our analysis is situated at farm level with the focus on (Flemish) agriculture. Although sustainability is a global concept, we opt for a farm level analysis because a farm can be seen as the unifying institution to achieve sustainability (Hagedorn, 2003). Examining activities at the organization level is helpful to a broader debate concerning the uptake of tools that can make the sector sustainable. While legal regimes, policy frameworks and cultural values in society have an impact on how an organization behaves, there are various influences which operate at the level of the firm (e.g., firm mission, strategic orientation) (Bebbington et al., 2007). In the empirical applications of this dissertation, economic and sustainability performance are measured in terms of efficiency (*doing things right*). Efficiency relates the used resources to the obtained results. One should not confuse efficiency with effectiveness. This latter concept compares the results to the desired outcomes or objectives (*doing the right things*). We are aware that efficiency alone is not sufficient to measure economic and sustainability performance. However, efficiency improvement can be seen as a first important and necessary step towards higher (sustainability) performance. Therefore that we speak about an efficiency approach.

The objectives of the second part of this dissertation are to measure farm performance in a consistent way and to analyze differences in farm performance. The first application measures performance as technical efficiency. Environmental and social aspects are not taken into account in this traditional economic performance measure. Several managerial and structural characteristics explain the differences in measured efficiency (a proxy for economic performance).

Furthermore, the impact of farm performance on structural change can be analyzed. To measure performance in a more complete way, environmental and social considerations should be integrated into the calculation of agricultural performance. In the following applications farm performance is measured in terms of sustainable value. Again, the differences in farm sustainability can be explained by differences in managerial and structural characteristics. In a final application, the sustainable value methodology used to measure farm sustainability is combined with efficiency analysis to construct the necessary benchmarks. In this case, the production theoretical underpinnings of efficiency analysis enrich the sustainable value approach.

Hence, the second part of the dissertation will focus on economic and sustainable farm performance. Put in a nutshell, this dissertation tries to make several performance and sustainability concepts operational. Empirical results can help decision takers (policy makers, farmers,...) in their aim to create a sustainable and efficient farming sector. Putting sustainability concepts into practice will contribute to the search for a more sustainable agriculture.

The general objectives of this dissertation can be summarized as:

★ to give an overview of the different notions of sustainability (part I, chapter 2 and 3);

★ to give an overview of the most important approaches to measure (sustainability) performance on different levels given the different notions of sustainability (part I, chapter 4);

★ to measure farm efficiency as an indicator of economic performance (part II, chapter 5);

★ to explain differences in farm efficiency (part II, chapter 5);

★ to link farm performance with structural change (part II, chapter 6);

★ to measure the sustainable value of farming as an indicator of sustainability performance (part II, chapter 7);

★ to explain differences in farm sustainability (part II, chapter 7);

★ to improve sustainability assessment using production frontier benchmarks (part II, chapter 8).

Part I

Conceptual and theoretical framework

Introduction (part I)

Sustainability means different things to different people
—Talbot Page

The debate on sustainability is obscured by a number of misunderstandings. First, there is an ongoing dispute concerning different visions about the limits of economic growth and the carrying capacity of the Earth. Second, there is a clear discrepancy between theoretical sustainability and practical sustainability (Van der Hamsvoort, 2006). Hence, bringing sustainability into practice deserves our full attention.

We will first discuss the underlying concepts of sustainability, before discussing the measurement of sustainability and evaluating differences in sustainability indicators. Without defining the concepts of sustainability itself, the selection and use of indicators tends to have an intuitive and pragmatic appeal (Patterson, 2006). To operationalize sustainability, it is therefore necessary to understand the several notions of sustainability. In chapter 2, the main concerns for sustainability and some definitions and assumptions of sustainability are explained. In the subsequent chapter 3 different notions and their underlying assumptions are discussed. In chapter 4, an overview is given of different methods to measure (sustainability) performance. After a short overview of some economic performance measures, a review is given of both methods and assumptions to measure weak and strong sustainability. Finally, some sustainability methods to measure firm sustainability are discussed.

This first part of the dissertation gives a theoretical framework for the subsequent part where some empirical analyses are performed. Understanding the different existing notions of sustainability is essential before making sustainability operational using empirical applications.

Chapter 2

Concern for sustainable development

Hurtling into the future, without any brakes and in conditions of zero visibility accurately describes my concerns and those of many people I know
—John Peet

2.1 Introduction

Kenneth Boulding describes his concern for sustainable development with his image of the *Earth as a space ship*:

> Earth has become a space ship, not only in our imagination but also in the hard realities of the social, biological, and physical system in which man is enmeshed. In what we might call the *old days*, when man was small in numbers and Earth was large, he could pollute it with impunity, though even then he frequently destroyed his immediate environment and had to move on to a new spot, which he then proceeded to destroy. Now man can no longer do this; he must live in the whole system, in which he must recycle his wastes and really face up to the problem of the increase in material entropy which his activities create. In a space ship there are no sewers (Boulding, 1966).

Boulding (1966) describes the transition from a *cowboy economy* without limits to a *spaceman economy*, without unlimited reserves. He states that we have to minimize the throughput of material in the economy and to try to produce as efficiently as possible. As a general rule, the higher the capital stock on board (of the space ship), the better. The throughput is the inevitable cost of maintaining the stocks of people and artifacts and should be minimized subject to the maintenance of a chosen level of stocks.

Closely related to Boulding's space ship image is the *Steady State Economy* by Herman Daly. A steady state economy is defined by constant stocks of physical wealth and a constant population, each maintained at some chosen, desirable level by a low rate of throughput (Daly, 1974). Daly (1974) also sees our economy as a subsystem of the Earth, and the Earth apparently as a steady-state system. A subsystem cannot grow beyond the frontiers of the total system and that subsystem must at some point conform to the steady-state mode, otherwise it will disrupt the functioning of the total system. An economy may be functioning very efficiently from the point of view of production in isolation, but this may be beyond the capacity of the environment. Daly (1991*a*, p. 35) draws an analogy with a nautical plimsoll line:

> Optimal allocation of a given scale of resource flow within the economy is one thing. Optimal scale of the whole economy relative to the ecosystem is an entirely different problem. The micro allocation problem is analogous to allocating optimally a given amount of weight in a boat. But once the best relative location of weight has been determined, there is still the question of the absolute amount of weight the boat should carry. The absolute optimal scale of load is recognized in the maritime institution of the Plimsoll line. When the watermark hits the Plimsoll line the boat is full, it has reached its safe carrying capacity. Of course, if the weight is badly allocated, the water line will touch the Plimsoll mark sooner. But eventually as the absolute load is increased, the watermark will reach the Plimsoll line even for a boat whose load is optimally allocated. Optimally loaded boats will sink under too much weight, even though they may sink optimally!

In 1972 Meadows et al. (1972) built a world model to investigate five major trends of global concerns: (i) accelerating industrialization, (ii) rapid population growth, (iii) widespread malnutrition, (iv) depletion of nonrenewable resources and (v) deteriorating environment. The main conclusions of this research team associated with the Massachusetts Institute of Technology were (Meadows et al., 1972):

1. If the present growth trends in world population, industrialization, pollution, food production, and resource depletion continue unchanged, the limits to growth on this planet will be reached sometime within the next one hundred years. The most probable result will be rather sudden and uncontrollable decline in both population and industrial capacity.

2. It is possible to alter these growth trends and to establish a condition of ecological and economic stability that is sustainable far into the future. The state of global equilibrium could be designed so that the basic material needs of each person on Earth are satisfied and each person has an equal opportunity to realize his individual human potential.

This report to the Club of Rome (*The Limits to Growth report*) received a great deal of publicity and was the subject of much public discussion. Although many critics believed that the authors had overdramatized their conclusions, especially the one that predicted a rapid cessation of growth, almost all agreed that they had truly identified trends of global concern, notably population growth and environmental degradation (Cameron and Neal, 2003).

During the United Nations Conference on Human Environment of 1972, all members agreed with several principles which considered the need for a common outlook to inspire and guide the peoples of the world in the preservation and enhancement of the human environment (United Nations, 1972)

In 1987 the World Commission on Environment and Development, also known as the *Brundtland Commission*, published its report *Our Common Future*. This highly quoted report placed *sustainable development* in the forefront (World Commission on Environment and Development, 1987, p. 8-9) and called for action:

> The concept of sustainable development does imply limits, not absolute limits but limitations imposed by the present state of technology and social organization on environmental resources and by the ability of the biosphere to absorb the effects of human activities.(...)
> Yet in the end, sustainable development is not a fixed state of harmony, but rather a process of change in which the exploitation or resources, the direction of investments, the orientation of technical development, and institutional change are made consistent with future as well as present needs. We do not pretend that the process is easy or straightforward. Painful choices have to be made. Thus, in the final analysis, sustainable development must rest on political will.

The Brundtland report was innovative in content and procedure (Spangenberg et al., 2002b). This report was followed by the United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro in 1992, bringing together diplomats, politicians and experts in environment and development from 172 member states of the United Nations and more than 1100 Non Governmental Organizations. The Rio Declaration can be seen as the frame of reference for sustainable development (Hens, 1996). Therefore, the first of the 27 principles states that:

> Human beings are at the center of concerns for sustainable development. They are entitled to a healthy and productive life in harmony with nature (United Nations, 1992).

The actual Rio Declaration reflects a very delicate balance of principles which both industrialized and developing countries consider important. In fact, the UNCED 1992 was an important benchmark in developing an international basis for sustainable development (Hens, 1996).

In 2002 the commitment to sustainable development was reaffirmed at the World Summit on Sustainable Development in Johannesburg (United Nations, 2002):

> We commit ourselves to act together, united by a common determination to save our planet, promote human development and achieve universal prosperity and peace.

Not everyone is convinced of the usefulness of the sustainable development concept. Beckerman (1994) argues that sustainable development has been defined in such a way as to be either morally repugnant or logically redundant. He states that the only development that is sustainable now is development that enables people to live together peacefully (Beckerman, 2003). However Beckerman (1994) assumes unlimited capital-resource substitutability and his argument that weak sustainability offers nothing beyond traditional economic welfare maximization is incorrect (Pezzey and Toman, 2002*a*, Neumayer, 2003). Lomborg (2001) too is not worried. He sees great overall improvements within the environmental area in the developed world. Although Bjørn Lomborg claims it will make good sense to invest even more in sound environmental management, his publication (*the Skeptical environmentalist*) got a lot of critical reviews, e.g., Grubb (2001); Pimm and Harvey (2001); Ege and Christiansen (2002). Pimm and Harvey (2001) states that Lomborg's work is a mass of poorly digested material, deeply flawed in its selection of examples and analysis.

In a nutshell one can say that the concern for sustainability derives from an ethical concern for future generations (Perman et al., 2003). It is clear that the concern for a sustainable development (including environmental concern) is obvious and even natural. However, further theoretical and methodological research is necessary. It will be an enormous challenge to detect and respond in time to any potential threat to sustainability (Pezzey, 1992*a*). In addition, empirical work to make sustainability operational is required in our pursuit towards more sustainable production and consumption. Even Wilfred Beckerman, known as fierce opponent of the concept of sustainable development, states that (Beckerman, 1994, p. 192):

> Economic policy has tended to ignore environmental issues, particulary those having very long run consequences. It is right, therefore, that they should now be given proper place in the conduct of policy.

2.2 Definitions and assumptions of sustainable development

A first step toward clarity would be to make the distinction between growth and development. Economic growth can be defined as increasing aggregate consumption or output. Growth ignores environmental quality and other social factors, and also ignores the distribution of income (Pezzey, 1992*b*). Economic growth, which is an increase in quantity (e.g. in consumption), cannot be sustainable on a finite planet. Economic development, which is an improvement in the quality of life without necessarily causing an increase in quantity of resources consumed, may be sustainable (Costanza et al., 1991). In short, growth is a quantitative increase on a physical scale, while development is qualitative improvement or unfolding of potentialities. The term a sustainable growth should be rejected as a bad oxymoron (Daly, 1990).

As in each text about sustainable development, we will start with the most known definition of sustainable development:

> Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs (World Commission on Environment and Development, 1987, p. 43).

This definition can be seen as the standard definition when judged by its widespread use and frequency of citation (Kates et al., 2005). Already in 1989, two years after the introduction of sustainable development by the Brundlandt Commission, dozens of verbal definitions existed and were listed by Pezzey (1992*b*). But in fact, although the Brundlandt definition of sustainable development captures the essence of sustainable development, it is hard to use in economic analysis because of the difficulty of the concept of *need* (Pezzey, 1992*a*). In his definition of sustainability, Pezzey (1992*b*, p. 14) tries to relate most aspects of sustainability to the economic concepts of production functions and utility functions:

> Sustainability is non-declining utility of a representative member of society for millennia into the future.

Neumayer (2003, p. 7) proposes the following definition for the economic concept of sustainable development:

> Development is sustainable if it does not decrease the capacity to provide non-declining per capita utility for infinity.

This last definition is not utilitarian (in contrast with the Pezzey (1992*a*) definition), because here sustainable development is defined in terms of maintaining the capacity and not the utility itself. Also it leaves space for free choice (Neumayer, 2003). Definitions often contain several aspects. For example, Stavins et al. (2002, p. 3) interpret sustainability in terms of efficiency plus intergenerational equity. Their economic definition of sustainability is:

> An economy is sustainable if and only if it is dynamically efficient and the resulting stream of maximized total welfare functions is non-declining over time.

Many other definitions are possible, but three remarks apply to almost all sustainability criteria: (i) they are long term criteria, (ii) most criteria derive from a common school of ethical principles regarding intragenerational and/or intergenerational fairness or justice, (iii) they are mostly constraints rather than maximizing criteria such as for instance optimality (Pezzey, 1992*b*). Sustainable development has come to be associated with several normative principles. Baker (2006) distinguish the following normative principles of sustainable development: (i) common but differentiated responsibilities, (ii) inter-generational equity, (iii) intra-generational equity, (iv) justice, (v) participation, and (vi) gender equality. The principle of common but differentiated responsibilities provides a way for distributing responsibilities and tasks more fairly among all countries (developing versus developed countries). Intra-generational equity refers to equity within our own generation, while inter-generational equity refers to equity between generations.

It is clear that there is no universally agreed definition of the concept of sustainability. One finds a variety of definitions, meanings and interpretations (Perman et al., 2003). Furthermore, there is no universal definition of sustainability that may be applied at all times and all places (Köhn et al., 1999). The construct of sustainable development is fundamentally infused with multiple objectives and ingredients, complex interdependencies, and considerable moral thickness (Gladwin et al., 1995). As a consequence, some scholars forecast that the notion of sustainable development will remain fuzzy, elusive and contestable (Levin, 1993, Beckerman, 1994, Dasgupta and Mäler, 1995). This has led some observers to call sustainable development an oxymoron: fundamentally contradictory and irreconcilable. Further, if anyone can redefine and reapply the term to fit to their purposes, it becomes meaningless (Kates et al., 2005).

But definitional diversity can be expected during the emergent phase of any potentially big idea of general usefulness (Gladwin et al., 1995), each definitional attempt is an important part of on ongoing dialogue (Kates et al., 2005). Bell and Morse (1999) argue that the flexibility of the meaning of sustainability can be a great strength in a diverse world. People differ in the environmental, social and economic conditions within which they have to live and having a single definition that one attempts to apply across this diversity could be both impractical and dangerous. In fact, sustainable development draws much of its resonance, power, and creativity from its very ambiguity (Kates et al., 2005). Moreover, the theoretical criticism raised by Beckerman (1994) and Dasgupta and Mäler (1995) that the concept of sustainable development is not based on any clearly defined concept of social welfare has been answered by Howarth and others (Howarth, 1995, 1997, Pezzey, 1997, Norton and Toman, 1997), who argue that the incompatibility of sustainability with standard economics indicates narrowness in neoclassical economic theory, rather than any inconsistency in the inherently normative concept of sustainability as intergenerational justice (Harris, 2003). Solow (2000) summarizes this by stating that sustainability is not meaningless, it is just inevitably vague. Hence, a more useful exercise than providing an overview of several definitions or constructing a new one, is identifying the major issues characterizing sustainability.

The Board on Sustainable Development of the U.S. National Academy of Science focused in its report *Our Common Journey: A Transition toward Sustainability* on the distinction between what advocates and analysts sought to sustain and what they sought to develop. They identified three categories to be sustained: (i) nature (Earth, biodiversity, ecosystems), (ii) life support (ecosystem services, resources, environment), and (iii) community (cultures, groups, places). Similarly, there were three categories to be developed: (i) people (child survival, life expectancy, education, equity, equal opportunity), (ii) economy (wealth, productive sectors, consumption), and (iii) society (institutions, social capital, states and regions) (National Research Council, 1999).

Gladwin et al. (1995) distinguish several important components. They suggest that sustainable development is a process of achieving human development in an inclusive, connected, equitable, prudent and secure manner. Inclusiveness implies human development over time and space. Connectivity entails an embrace of ecological, social, and economic interdependence. Equity suggests intergenerational, intragenerational, and interspecies fairness. Prudence connotes duties of care and prevention: technologically, scientifically, and politically. Security demands safety from chronic threats and protection from harmful disruption (Gladwin et al., 1995). Key points of sustainability which emerge are (Pezzey, 1992*b*): (i) sustainable resource use, (ii) attention to the needs of the current (intragenerational) as well as the future (intergenerational) poor, and (iii) the geographical and temporal context.

An important aspect of the application of sustainability is the emphasis on multidimensionality (economic, social and environmental). According to the multidimensional concept of sustainability, sustainability can be characterized as the long-term preservation of the viability of the overall system and its components (Spangenberg et al., 2002b). The *three-legged stool* model of sustainability consists of the three mutually reinforcing and critical aims of sustainable development: (i) the improvement of human well-being (economic security), (ii) more equitable distribution of resource use benefits across and within societies (social equity), and (iii) development that ensures ecological integrity over intergenerational scales (ecological integrity) (Sneddon et al., 2006). Furthermore, the general objective of sustainable development is to maximize these aims across the social, economic and ecological systems through a process of trade-offs, because it is not possible to maximize all the objectives all the time. For example, as the economic process of production is dependent on resource use, increasing even useful goods and services may conflict with ensuring the economic security and productivity and the genetic diversity of the ecological and resource system (Barbier, 1987). Martens (2006) distinguishes separate underlying principles for each aspect of sustainable development: efficiency plays a primary role in economic sustainable development, whereas justice plays an important role in social sustainable development, and resilience or capacity for recovery plays a primary role in ecological sustainable development. It is essential to analyze these economic, social and environmental dimensions in a balanced manner.

Furthermore, sustainability can be recognized on multiple layers ranging from supra-national, national, regional, sectoral and firm level (Bebbington et al., 2007). The achievement of sustainability requires an effective integration of these multiple levels and systems (Starik and Rands, 1995). Sustainability is addressed within systems that are nested in space and time (Pearson, 2003). Hence before achieving sustainability, one has to answer the question (Bell and Morse, 1999): *over what space and time is sustainability to be achieved?* Sustainability is clearly dynamic, in fact it can be described as a process of sustainable qualitative improvement (Lawn, 2001). In other words, the resulting sustainability state of a system is not a static balance of all dimensional concepts, but a dynamic, evolutionary process of permanent change (Spangenberg et al., 2002a).

Costanza and Patten (1995) formulate the basic idea of sustainability as a system that survives or persists. They separated the problem of defining sustainability from three basic questions: (i) What system(s) or subsystems or characteristics of systems persist? (ii) For how long? and (iii) When do we assess whether the system or subsystem has persisted? Costanza and Patten (1995) state that to answer the *what question* a nested hierarchy of systems over a range of time and space must be considered. To answer the question for how long systems should survive, they state that a system is sustainable if it attains its full expected life span within the nested hierarchy of systems within which it is embedded. In fact, some finite time horizon is present in almost all sustainability writing[1], given a finite life for our sun it is impossible for our planet's civilization to be sustained forever (Pezzey and Toman, 2002*b*). The assessment of the persistence of systems (the *when* question) can only be done after answering the what question. So it is important to predict what configurations will persist, and to develop policies and instruments to deal with remaining uncertainty (Costanza and Patten, 1995).

Hence, within the sustainability debate three concepts are essential: (i) natural resources are finite and there are limits to the carrying capacity of the Earth's ecosystem, (ii) economic, environmental and social goals must be pursued within these limits, (iii) there is a need for inter- and intragenerational equity (Farrell and Hart, 1998). It is clear that a precise definition of sustainable development will remain an ideal, elusive and probably unreachable goal. A less ambitious but more focused strategy is to make development more sustainable (Munasinghe, 2002). This incremental method is more practical, because many unsustainable activities can be recognized and eliminated (Islam et al., 2003), as explained in the following section.

2.3 Towards more sustainable development

To move towards sustainable development, several approaches can be considered. The process of working towards more sustainable development can be translated in several steps (section 2.3.1) and an efficiency improvement can be seen as an important first step (section 2.3.2). Next, a transition towards sustainability is necessary; in that case sustainable development is defined as a long-term, complex and drastic process of change (section 2.3.3).

2.3.1 Steps towards sustainability

Absolute sustainability may not be feasible for any particular pattern of industrial development, but one can still find a pattern that is relatively more sustainable. In many instances of sustainable development, one is talking about transforming a system that was previously unsustainable into one that is at least relatively sustainable (Barbier, 1987).

[1] This finite horizon is often not explicit

It can be useful to see the transition of current production practices to more environmentally friendly production as an overlapping three-stage process - efficiency, substitution and redesign (Hill et al., 1999). A similar approach is described by Pretty (1998), he sees the transition to sustainability as a three step process (Pretty, 1998)[2]:

Step 0 Conventional production

Step 1 Improved economic and environmental efficiency;

Step 2 Integrating regenerative technologies;

Step 3 Redesign with communities.

Regarding technology, the rule of sustainable development would be to emphasize technologies that increase resource productivity (the amount of value extracted per unit of resource) rather than technologies that increase the resource throughput itself (Daly, 1990).

Somewhat differently, Kuhndt and Seifert (2004) distinguish several phases for sustainable business development: (i) green entrepreneurship, (ii) efficient entrepreneurship and (iii) responsible entrepreneurship. The focus of green entrepreneurship is on internal environmental improvements, first output-orientated, then process-orientated and finally system-orientated. In contrast with the internal focus, the efficient entrepreneur pays attention to the economic relevance of environmental issues, including external effects within markets (chain-oriented). Hence, in a cooperative way, win-win situations are created for the entire chain. Finally, for responsible entrepreneurship (stakeholder-oriented), companies base their vision and policy on stakeholder expectations, as a sense of responsibility towards society (Kuhndt and Seifert, 2004).

2.3.2 Efficiency as a first step towards sustainability

Callens and Tyteca (1999) adopt the view that economic, social and environmental efficiency is a necessary (but not sufficient) step towards sustainability. Improving efficiency might be important towards more sustainability because it could allow the often conflicting environmental and economic objectives to be achieved simultaneously. Increasing the output per unit of input would not only reduce environmentally harmful emissions but also increase income by saving costs or increasing output (De Koeijer, 2002). Moreover, improving end use efficiency of resources is desirable regardless of whether the resource is renewable or nonrenewable (Daly, 1990).

[2] An application can be found in Webster (1997).

However, efficiency will not necessarily lead to sustainability. Efficiency puts society on the utility possibilities frontier, but sustainability is also a matter of distributing assets across generations (Howarth and Norgaard, 1992). In theory, there is an infinite number of efficient states. Some of these will be sustainable, while many others use resources in a manner that will leave later generations with much diminished economic opportunities. The efficiency criterion, however, does not help us to distinguish between sustainable and unsustainable time paths (Woodward and Bishop, 1995). Advances in resource efficiency can be overcompensated because higher efficiency may lead to increased use of (environmental) resources. This is called the rebound effect (Mayumi et al., 1998, Herring and Roy, 2002). But efficiency is a precondition for any morally acceptable resource use, because inefficiency implies waste (Ruth, 2006). Hence, it is true that restoring efficiency is not sufficient to produce sustainability (Perman et al., 2003, Tietenberg, 2003), but efficiency improvements will generally result in an improvement in sustainability (Tietenberg, 2003). De Koeijer et al. (2002) also state that enhancing efficiency may support sustainability. Stavins et al. (2002) suggest that a broadly-accepted and normatively useful notion of sustainability can be better understood by breaking it into two components, both of which are well defined in economics: dynamic efficiency and intergenerational equity. Sustainability is not only about intergenerational equity, sustainability encompasses elements of both efficiency and distributional equity. We can conclude that efficiency is a necessary but not a sufficient condition for sustainability.

2.3.3 Transition towards sustainability

Most transitions involve trade-offs, but these trade-offs need not always be harmful. Pretty (2003) explains that it is possible to produce more food whilst protecting and improving nature, and that it is possible to have diversity in both human and natural systems without undermining economic efficiency. Dealing with trade-offs is a major factor in assessments and related decision making (Gibson et al., 2005). Furthermore, measuring sustainability involves dealing with trade-offs (see section 4.3).

To achieve sustainability, fundamental changes are needed. These changes are denoted in terms like system innovation and transition. In fact, the realization of sustainability can be seen as a process of social innovation (Rotmans, 2005). Substantial improvements in environmental efficiency (with a *Factor 2*), may still be possible with innovations of an *incremental* kind (Geels et al., 2004), this can be called system optimization. Larger jumps in environmental efficiency (possibly by a *Factor 10*) may only be possible by system innovations. Transition can be defined as a profound process of change on different social levels in the long run. Historical examples include: (i) from hunting to agriculture, (ii) from sailing ships to steamships, or (iii) from mechanics to informatics. The transition towards sustainability is somewhat different because there is a clear postulated target. The transition from one socio-technical system to another can be called system innovation, which is wider than product and process innovation. Such a system innovation is required for a successful transition to a sustainable system. A transition is complex, drastic and takes time. Moreover, there is also a lot of uncertainty. Transition management is a brand new discipline which analyzes transitions and system innovations and which also tries to influence transition processes. A conceptual framework for the transition towards sustainable systems is depicted in figure 2.1.

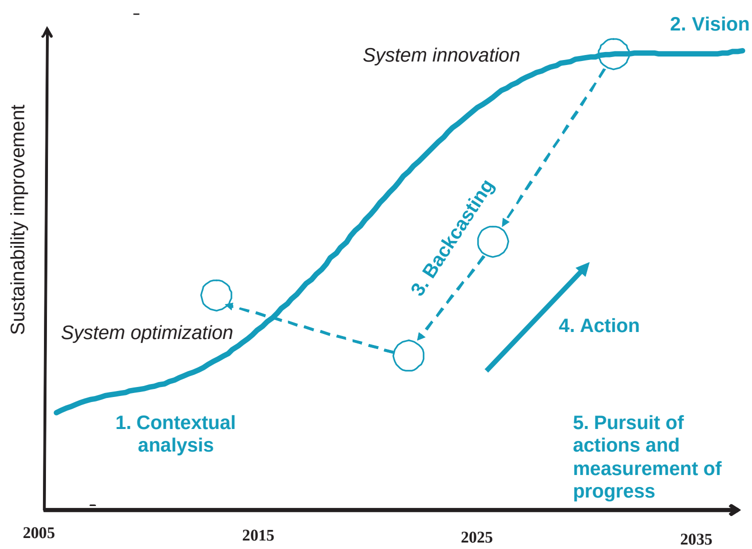

Figure 2.1: Conceptual framework for the transition towards sustainable systems (Source: Based on Nevens et al. (2007))

Transition management is a visionary, evolutionary learning process, which is progressively constructed by undertaking several steps (Martens, 2006). Figure 2.1 shows the different steps to achieve sustainability: (i) contextual analysis, (ii) vision development, (iii) backcasting and strategy choice, (iv) action, and (v) process monitoring. The first step, the contextual analysis, forms the basis of the problem outline and helps to create a first image of the desirable outcome. In the next step, this desirable image is worked out in a well-considered vision. In the third step, a strategy is developed. A strategy shows a possible path from the current situation towards the envisioned future and serves as a decision-base for taking actions. A way to achieve the desirable ideal is to use back casting. The technique of back casting can be used to determine milestones, as intermediate stop-overs in our way to achieve sustainability. In the short term existing system are adjusted and improved (system optimization), in the long term existing systems are changed in totally new systems with new functions (system innovation). The fourth step is to take action. Once the final goal and milestones are known, it is possible to take specific and goal-oriented actions. Note that uncertainty still exists and there are always early adopters with success and also with failure. The fifth step in this framework is the monitoring of the progress made and thus the measurement of sustainability using indicators. A detailed overview of several methods to measure (sustainable) performance is given in chapter 4. Evaluation methods are necessary to assess performance and to adjust the actions towards the sustainability (vision). It can be useful to formulate and execute local experiments that could contribute to the transition to sustainability. Afterwards, these experiments should be evaluated and we should assemble the vision and the strategy for sustainability based on what has been learned. More information about system innovation and the transition to sustainability can be found in Elzen et al. (2004), Geels (2005) and Rotmans (2005). An application of the transition approach in agriculture can be found in Steunpunt Duurzame Landbouw (2006) and in Nevens et al. (2007).

2.4 Sustainability and economics

2.4.1 Sustainability during the history of economic thinking

In the course of history many scholars have contributed to economic theory. Aristoteles[3], the scholastics[4], the adherents of mercantilism[5], the physiocrats[6] and many more have all added important contributions to the development of the economic science. In 1776 Adam Smith published his famous book *An Inquiry into the Nature and Causes of the Wealth of Nations*. Smith is often regarded as the father of economics, and his writings have been enormously influential. The birth of economics as a discipline is therefore traditionally associated with the work of Adam Smith (Edwards-Jones et al., 2000). His work marked the breakthrough of an evolutionary approach which has progressively displaced the stationary Aristotelian view. The belief in the efficacy of the market mechanism is still a fundamental organizing principle of the policy prescriptions of modern economics, including resource and environmental economics (Perman et al., 2003). Smith was writing at a time in which the availability of natural resources was not a constraint on the economy. His per-

[3] Important works of his economic insights are *Ethica Nicomachea* and *Politica*

[4] For example Thomas van Aquino with his work *Summa Theologica*

[5] For example the Frenchman Jean-Baptiste Colbert and the Englishman Thomas Mun

[6] The most important physiocrat was François Quesnay with his work *Tableau Œconomique*

ception on the value of nature was instrumental, meaning nature was valuable only if it serves human wants:

> The Earth furnishes the means of wealth, but wealth cannot have any existence, unless through industry and labor which modifies, divides, connects, and combines the various production of the soil, so as to render them fit for human consumption (Smith, 1776).

Adam Smith was extremely favorable to growth. One of the first persons who stated that economic growth wouldn't go on without limits was Thomas Robert Malthus[7]. In his *Essay on the Principle of Population* Malthus stated that:

> I think I may fairly make two postulata. First, that food is necessary to the existence of man. Secondly, that the passion between the sexes is necessary, and will remain nearly in its present state. These two laws ever since we have had any knowledge of mankind, appear to have been fixed laws of our nature. (...)
> Assuming, then, my postulata as granted, I say, that the power of population is indefinitely greater than the power in the Earth to produce subsistence for man. Population, when unchecked, increased in a geometrical ratio. Subsistence increases only in an arithmetical ratio (Malthus, 1798).

Malthus's arguments can be challenged. The law of diminishing returns is only applicable under the assumption that the state of technology remains constant. Furthermore, as income and education levels have improved, family size has diminished (Edwards-Jones et al., 2000). But he was the first to express the limitations imposed by the finite nature of the natural resources.

Natural resources were seen as important determinants by classical economists (Dietz et al., 1994). The natural resource *land* was viewed as limited. Moreover, land was a necessary input to production and exhibited diminishing returns. The early classical economists came to the conclusion that economic progress would be a transient feature of history. They saw the inevitability of a stationary state, in which the prospects for the living standard of the majority of people were bleak (Perman et al., 2003). This notion of a steady state was extended by David Ricardo in his *Principles of Political Economy and Taxation (1817)*. John Stuart Mill adopted a broader view of the roles played by natural resources than his predecessors. He states that natural resources ultimately impose restrictions on production. The available natural resources are limited and furthermore they show diminishing returns:

[7]The basics of his theory was also written down earlier by Giovanni Botero in 1592. His work was titled *Delle Cause della Grandezza della Citta*

> After a certain, and not very advanced, stage in the progress in agriculture, it is the law of production from the land, that in any given state of agricultural skill and knowledge, by increasing the labor, the produce is not increased in equal degree; doubling the labor does not double the produce; or to express the same thing in other words, every increase of produce is obtained by a more than proportional increase in the application of labor of the land (Mill, 1848).

An interesting feature in Mill's thinking was the argument that the quality of living space is an important part of economic well-being. According to Mill, a world where the environment is used completely for industrial and agricultural purposes is not an ideal world (Tahvonen, 2000).

Since the 1870s several important works were published as what became known as neoclassical economics. Neoclassical economics has become and still is the dominant form of economic analysis. Previous notions of absolute scarcity and value were replaced by the concept of relative scarcity. Furthermore, the technique of marginal analysis was adopted. The general equilibrium theory was developed by Leon Walras[8] and this provided a foundation for the concepts of efficiency and optimality (Perman et al., 2003). The concern with the level and the growth of economic activity has been largely ignored during the period where neoclassical economics has been developed. Also classical limits-to-growth arguments, based on a fixed land input, did not have any place in early neoclassical growth modeling. Land or natural resources were absent in the production functions used in the early neoclassical growth models (Perman et al., 2003). The introduction of natural resources into neoclassical models of economic growth didn't occur until the 1970s (Perman et al., 2003).

Nowadays, several economic subdisciplines study the interaction of economics and our environment (i.e. sustainability), for example natural resource economics, environmental economics and ecological economics. The precise dividing line between these disciplines is a little fuzzy (Cropper and Oates, 1992). Natural resource economics focuses on the intertemporal allocation of renewable and nonrenewable resources. Environmental economics analyzes environmental issues by extending the neoclassical economic tools and principles. Ecological economics tries to understand and to address environmental problems in an interdisciplinary way. In the following sections, a short overview is given of the several economic subdisciplines studying sustainability.

[8] The proof of the existence of the equilibrium was given in 1954 by Arrow and Debreu (1954)

2.4.2 Natural resource economics

Resource economics is concerned with the use of natural resources as inputs to economic processes; it deals with issues such as the extraction rate of minerals from the Earth, the harvesting of fisheries and forestry and the management of other resources such as water and renewable energy sources. It analyzes how prices regulate the quantities produced of desired environmental goods (Fullerton and Stavins, 1998, Edwards-Jones et al., 2000).

The origin of natural resource economics can be found in the seminal paper by Hotelling (1931). In Hotelling's model, the owner of a nonrenewable resource has two options: extract now, and earn interest on the proceeds of sale, or keep the resource in the ground, with the expectation that the price will rise (Edwards-Jones et al., 2000). The theory of natural resource economics applies dynamic control methods of analysis to problems of intertemporal resource usage (Cropper and Oates, 1992). Nonrenewable resources (e.g., oil, gas, coal, copper, silver) are formed by processes over million of years and exist as fixed stock which, once extracted, cannot be renewed. Some depletable resources are recyclable (e.g., copper). A recyclable resource is one which, although currently being used for some particular purpose, exits in a form allowing its mass to be recovered once that purpose is no longer necessary or desirable (Tietenberg, 2003). Natural resources economists search for the optimal extraction path over time for any particular nonrenewable stock (Perman et al., 2003). Environmental resources are renewable when they have a capacity for reproduction and growth, such as biological organisms (e.g., fisheries and forests) and atmospheric systems (e.g., water). Renewable resources are differentiated from depletable resources primarily by the fact that natural replenishment augments the flow of renewable resources at a non-negligible rate (Tietenberg, 2003). Most of the literature on the economics of renewable resources is about two research areas: the harvesting of animal species (hunting and fishing) and the economics of forestry (Perman et al., 2003). Agriculture could also be thought of as a branch of renewable resource harvesting, but in this case the environmental medium is designed and controlled.

2.4.3 Environmental economics

Already in 1919, King (1919) stated that a high national income is not the same as a high level of wealth. A high national income in industrialized countries often goes hand in hand with the existence of scarcity of natural resources (King, 1919). However, the real revolution in environmental economics did not start before the late 1960s (Cropper and Oates, 1992). Before 1970 economists saw pollution as the consequence of an absence of prices for certain scarce environmental resources, and they prescribed the introduction of surrogate prices in the form of unit taxes or effluent fees. The continuous growth of industrial production and the changing social climate resulted in environmental problems as an everyday phenomenon (Dietz et al., 1994). Pollution and environment became part of the general public awareness and the scientific interest for environmental issues increased during the seventies.

The three functions of the environment for the economy are (Røpke, 2004) (i) resources for production, (ii) assimilative capacity to absorb pollution and (iii) direct utility related to the enjoyment of nature (amenity value). Environmental economists are interested in pollution and other externalities (Fullerton and Stavins, 1998, Walter, 2002). Environmental economics was traditionally concerned with the use of the environment as a sink for the waste products of economic activity, and in the beginning researchers in this field were therefore mainly concerned with pollution and its control. Hence, this work was concerned with how prices failed to regulate the use of the environment, and how to correct those prices. Over time other related issues such as the conservation and management of biodiversity, environmental decision making, agriculture, forestry and soil conservation were also studied by environmental economists (Edwards-Jones et al., 2000).

The theory of externalities is essential in the economic analysis of environmental aspects (Cropper and Oates, 1992). An externality exists whenever the welfare of some agent, either firm or household, depends not only on his or her activities, but also on activities under the control of some other agent (Tietenberg, 2003). Early work in the analysis of externalities and market failure can be found in Marshall (1890). The first systematic analysis of pollution as an externality can be found in Pigou (1920). Pigou's most famous contribution is that of *internalizing externalities* associated with environmental damages (Edwards-Jones et al., 2000). Pigou (1920) proposed to impose taxes on polluting firms in proportion to the output of the pollution, known as the Pigouvian tax. Arthur Pigou can be seen as the most influential writer in the interventionist tradition (Edwards-Jones et al., 2000). An important aspect of the Pigouvian solution to pollution was the ability of the polluter to *get away* with pollution because there are no defined property rights to environmental resources (Edwards-Jones et al., 2000). The importance of property rights within the analysis of environmental issues was highlighted by Coase (1960). The Pigouvian neoclassical tradition still continues to dominate the analytical foundation of environmental economics (Cropper and Oates, 1992, Venkatachalam, 2006). The external effect is an untraded and unpriced product which arises because the victim has no property rights that can be exploited to obtain compensation for the external effect. However, the absence of a price for a resource does not mean that it has no value (Perman et al., 2003). Environmental economists developed several valuation techniques such as the contingent valuation, the contingent ranking, the hedonic values and the avoidance expenditures (Tietenberg, 2003). Other important topics and seminal papers are the second best solution in the area of pollution control (Baumol and Oates, 1988), non-market valuation within micro cost-benefit analysis (Smith, 1993) and environmental accounting (Ahmad et al., 1989) within macroeconomics of the environment (Munasinghe, 2002). Note that sustainable development is also an important topic within environmental economics (Venkatachalam, 2006).

Environmental economics has been a busy field over the past three decades. Environmental economists have reworked existing theory, making it more rigorous and clearing up a number of ambiguities. New methods for the valuation of benefits from improved environmental quality were developed. Numerous empirical studies to measure the costs and benefits of actual or proposed environmental programs were undertaken (Cropper and Oates, 1992).

2.4.4 Ecological economics

Ecological economics is a relatively new transdisciplinary field of study that addresses the relationships between ecosystems and economic systems in the broadest sense[9]. The basic observation in ecological economics is that the human economy is embedded in nature, and economic processes are always natural processes as they can be seen as biological, physical or chemical processes and transformations (Røpke, 2004). Ecological economics is in fact not a totally new discipline but rather a synthesis of many separate disciplines (Edwards-Jones et al., 2000). Important is the fact that economics and ecology are seen as the two disciplines most directly concerned with what can be seen as the central problem: *sustainability* (Perman et al., 2003). Economics is of central concern within ecological economics for three reasons (Edwards-Jones et al., 2000): (i) it can be a foundation for public decision making, (ii) it can explain and predict behavior in relation to natural resources through cost-benefit analysis and (iii) it can develop possible improvements to decision-making in both these contexts through specific tools of analysis. Ecological science also has a pivotal role in ecological economics (Edwards-Jones et al., 2000): (i) it provides data on important ecological parameters[10], (ii) it identifies and predicts changes occurring in natural systems in response to human-caused stresses and it thereby provides guidance on the management of human economic activity to sustain environmental quality in the long term, and (iii) it also generates and communicates a deeper and richer understanding of the immense diversity of life on Earth (Edwards-Jones et al., 2000).

The basic idea of ecological economics is that the economy ought to be studied also, but not only, as a natural object, and that economic processes should consequently also be conceptualized in terms usually used to describe processes in nature (Røpke, 2004). Kenneth Boulding, one of the founding fathers

[9] Overall, ecological economics is already a success story in the establishment of a new scientific field (Røpke, 2005)

[10] For example data about the sustainable harvesting rates for biological resources

of ecological economics, describes the transiton from *cowboy economy* without limits to a *spaceman economy*, without unlimited reserves in his *Earth as a space ship* (see section 2.1). Boulding states that we have to minimize the throughput of material in the economy and to try to produce so efficiently as possible. Ayres and Kneese (1969) state that in economic literature externalities are viewed as exceptional cases. They may distort the allocation of resources but can be dealt with using appropriate simple ad hoc arrangements. Ayres and Kneese (1969) find externalities a normal, inevitable part of the consumption and production process. Daly (1968) shares the same opinion and suggests to recast economics as a life science because the ultimate subject of biology and economics is one: *the life process*.

It could be said that ecology is the study of nature's housekeeping, while economics is the study of human housekeeping. Ecological economics could then be said to be the study of how these two sets of housekeeping are related to one another (Perman et al., 2003). Ecological economists do not take the view that resource and environmental economics are wrong in dealing with environmental problems, rather they stress the need to put environmental problems in the proper context, one where the economic system is seen as a subsystem of a larger system (Perman et al., 2003). Therefore, ecological economics is a transdisciplinary alternative to mainstream environmental economics, and ecological economists take a rather pluralistic approach (Forstater, 2004). Hence, ecological economic models of economic behavior try to encompass consumption and production in the broadest sense, including their ecological, social and ethical dimensions, as well as their market consequences (Gowdy and Erickson, 2005).

2.4.5 Industrial ecology, human ecology and bioeconomics

Industrial ecology also tries to address the need for biophysical reality in the analysis of human-environment interactions. Industrial ecology has been guided by the quest for production and consumption processes that minimize waste generation and thus, environmental impact (Ruth, 2006). Industrial ecology is industrial because it focuses on product design and manufacturing processes (Lifset and Graedel, 2002). Industrial ecology is ecological because it looks at non-human natural ecosystems as models for industrial activity (Frosch and Gallopoulos, 1989) and it places human technological activity in the context of the relevant larger ecosystems (Lifset and Graedel, 2002).

As in ecological economics, industrial ecology is a cluster of concepts, tools, metaphors and exemplary applications and objectives (Lifset and Graedel, 2002). More information can be found in Ayres and Ayres (2002) and in the *Journal of Industrial Ecology* or in the journal *Progress in Industrial Ecology*.

Human ecology investigates how humans and human societies interact with nature and with their environment (Park, 1936). Human ecology can be seen in terms of merging sociology and ecology. An historical overview of the development of human ecology can be found in Gross (2004).

Bioeconomics investigates the problem of specifying the process of natural production. In other words, it studies the process underlying the services nature provides in supplying raw material and disposing waste. Hence, bioeconomics primary theme is the inclusion of natural processes within natural production (Walter, 2002).

2.4.6 Studying sustainability within economics

Environmental economics, natural resource economics, bioeconomics, industrial ecology and ecological economics are all subdisciplines of economics that are particularly relevant in analyzing sustainability. All share the common objective of understanding the human-economy-environment interaction in order to redirect the economies towards sustainability (Costanza et al., 1991, Edwards-Jones et al., 2000, Venkatachalam, 2006).

Environmental and resource economics have offered important insights for governance for sustainable development, especially the notion of opportunity costs and the consequent imperative valuing of natural assets, based on their multifunctional contribution to human welfare (Dietz and Neumayer, 2006*b*). The strength of environmental economics lies in its analytical rigor and in its ability to provide concrete, firsthand solutions to major environmental problems. Its weakness is that it adopts a narrow approach which has prevented us from thinking about the larger features of the environmental and ecological issues (Lazear, 2000). Ecological economists try to incorporate those larger features within several frameworks using alternative economic standpoints (different from the neoclassical analytical approach). Large-scale environmental problems are characterized by significant risk, uncertainty and ignorance, by very long run effects, by threat of major, discontinuous and irreversible changes and often by fundamental irreplaceability of the asset in question (Dietz and Neumayer, 2006*b*). Ecological economics offers viable alternatives to the theoretical foundations and policy recommendations of environmental economics (Gowdy and Erickson, 2005, Gowdy, 2005).

Gowdy and Erickson (2005, p. 218-219) argues in favor of ecological economics and summarizes it as follows:

> It is often argued that economists must follow the narrow path of neoclassicism because there is no well developed alternative. However there is, but it requires abandoning the flawed grand unification

> theory of neoclassical welfare economics. Rather than a theory of everything, we appear to need theories of theories of things. Understanding the human economy requires an appreciation of the importance of hierarchies, contingency and self-organisation, and recognition of the fragility of market economies in biophysical space and cultural specificity.

Several authors argue in favor of new subdisciplines within economics to study sustainability, for example *natural economics* by Ruth (2006) and *sustainability economics* by Walter (2002). Ruth (2006) reviews insights from natural resource and environmental economics, ecological economics and industrial ecology to understand and promote sustainable development. He identifies four major themes for a *natural economics*: (i) the need to build on concepts from nature, (ii) the roles of efficiency and effectiveness in decision making, (iii) the need for adaptive and anticipatory management and (iv) the need for holistic impact assessments. Walter (2002) defines *sustainability economics* as the study of the use of resources for the achievement of an ongoing high quality of life, individual and social, within a context of co-stewardship of natural and human communities. Stewardship can be seen as a concern and action regarding the justice, healthiness and continuance of communities. O'Hara (1998) argues that sustainability cannot be achieved without a radical shift in our perception of the relation between economic activity and its social and ecological context. Therefore she calls for an internalizing of economics into the real world instead of internalizing the external effects of economic activity back into the conceptual framework of economics.

One can say that specialization in environmental economics has gone too far, while in contrast ecological economics adopts too many approaches (Røpke, 2005). But I would like to speak in positive terms: all economic subdisciplines studying sustainability (environmental and ecological economics, industrial ecology,...) can be seen as complementary in our aim to understand the human-economy-environment interaction. Tisdell (1994, p. 147) argues for a pluralistic approach as:

> The world is complex and we shall never know it fully. The best we can hope for is to explore it from different limited angles. Given that a pluralistic approach to theory is desirable, significant new perspectives such as those provided by sustainability considerations should be welcomed.

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2.5 Lessons learned

Our concern for a sustainable development is obvious and justifiable. On the one hand, there is no universally agreed definition of sustainability, on the other hand, we often find one or several of the following concepts in the description of the numerous notions of sustainability: (i) natural resources are finite and there are limits to the carrying capacity of the Earth's ecosystem, (ii) economic, environmental, and social goals must be pursued within these limits and (iii) there is a need for inter- and intragenerational equity. The definitional diversity does not mean that sustainability is meaningless, in fact the flexibility of the meaning of sustainability is rather a strength in a diverse world. Therefore, it is not surprising that several economic subdisciplines pretend to be relevant in analyzing sustainability. Note however that to make sustainability operational, it is essential to define sustainability in a clear way, indicating the underlying assumptions. For example, we see efficiency as a first step towards sustainability, but we are aware that efficiency will not necessarily lead to sustainability. We describe our efficiency approach as a necessary but not a sufficient condition for sustainability.

Chapter 3

Different notions of sustainable development

A principle of sustainable development is that there is no single blueprint for achieving such development which is universally applicable to all places and peoples
—Gareth Edwards-Jones

Within economics of sustainability, two main opposing paradigms of sustainability can be distinguished. The mainstream neoclassical view has come to be known as *weak sustainability*. This view states that substitutability of human-made capital for environmental resources is more or less unlimited, while proponents of the *strong sustainability* view find that capital-resource substitutability is either a self-evidently impossible concept, or subject to strict and fairly imminent limits. Note that *weak* and *strong* views are really views about the fact of substitutability and not about the goals of sustainability (Pezzey and Toman, 2002*b*). Because the main difference derives from starkly contrasting assumptions about the substitutability of natural capital, Neumayer (2003) calls the weak sustainability approach the substitutability paradigm and the strong sustainability approach the non-substitutability paradigm. Other descriptions found in literature of weak sustainability are the conventional economic optimistic view or technological optimism. Strong sustainability is sometimes described as the environmental pessimist vision or technological pessimism (Van der Hamsvoort, 2006)[1]

A somewhat different concept of sustainability focuses on consensus building and institutional development. Sustainability as defined in Agenda 21 has four dimensions: the social, economic, environmental and institutional one.

[1] We will mainly focus on weak versus strong sustainability, which both have their roots in economics (Van der Hamsvoort, 2006). There are also other descriptions for example the ecologist's definition of sustainability (Gatto, 1995, Van der Hamsvoort, 2006)

Institutions are not only organizations but also the systems of rules governing the interaction of members of a society (Spangenberg et al., 2002b). Despite the theoretical and practical progress, it is clear that there are significant barriers to the implementation of sustainable development on a broad scale. They arise especially as a result of issues of power, knowledge, and institutional structures (Harris, 2003). For example, uneven or unbalanced power relationships between farmers and non-farmers, between farmers and governments, and between farmers of large and small farms are a powerful explanation for environmental problems that so often arise in agriculture (Paarlberg, 2003). So far, there has been only limited work on institutions for sustainability (Spangenberg et al., 2002b). The institutional concept of sustainable development focuses on processes rather than looking at outcomes or constraints as do the weak and strong sustainability approaches (Perman et al., 2003). Since not all constraints are known, the sustainability of a given socio-environmental system cannot be assessed in advance (de Graaf et al., 1996). de Graaf et al. (1996) state that sustainable development is a development of a socio-environmental system with a high potential for continuity because it is kept with economic, social and cultural, ecological and physical constraints. Therefore, they propose a strategy that regards sustainable development as a development on which the people involved have reached consensus. In other words, they propose consensus building through negotiations. It is not clear yet what this negotiations process will exactly consist of (Perman et al., 2003).

Furthermore, the formal economic literature on sustainability has mostly focused on defining and justifying it or on measuring it, rather than on finding policies to achieve it (Pezzey, 2004). An exception is Pezzey (2004) who makes the distinction between sustainability policy and environmental policy. They not only have different goals, but also different instruments to achieve them. Environmental policy reflects a dynamic, governmental intervention to maximize social present value, by internalizing the social values of environmental stocks and flows that agents ignore[2] when they privately maximize present value. By contrast, sustainability policy aims to achieve some social improvement in intergenerational equity, such as making utility forever constant, non-declining or sustainable. Pezzey (2004) calls the combination of sustainable policy with environmental policy an optimal sustainability policy, because the resulting economy is efficient. The sustainability policy component can be represented as a shift from the representative agent's individual utility discount rate to some other, probably lower sustainability discount rate path.

Besides policy aspects, our main focus will lie on the production side and not on the consumption side of sustainability[3]. Sustainable consumption targets consumers, sustainable production is related to companies and organizations

[2] called externalities

[3] There does not exist a lot of literature about the link between consumption (preferences) and sustainability. Some examples of studies analyzing consumption and sustainability are Stern (1997), Norton et al. (1998), Köhn et al. (1999) and Wagner (2006)

that make products or offer services (Veleva and Ellenbecker, 2001). Furthermore, in this part we give minor attention to sectoral sustainability. Further, intragenerational equity is not discussed in detail and we left out discussions on the link between economic growth and the environment and discussions of population growth.

The degree of substitution between economic and natural capital has turned out to offer a level of abstraction at which many find it attractive to discuss different perspectives on sustainability (van den Bergh, 1999). Therefore we will discuss the arguments of both (weak and strong sustainability) approaches (sections 3.2, 3.3 and 3.4). Further, we will briefly discuss equity aspects in section 3.5. We will now explain how resources can be described to assess the sustainability of a system (section 3.1), we will also clarify the notion of *(natural) capital*.

3.1 Describing resources: the capital approach

Central to sustainable development is how we use the Earth's natural resources and the processes by which they are transformed. This can be described in terms of capital stocks (Cochrane, 2006), flows and the organization of these stocks and flows. Capital can be defined as all aspects needed for the production of valuable goods and services. For something to be considered a capital stock it must have the potential to produce something that is economically desirable (Goodwin, 2003). Each capital type provides flows, for example natural capital can provide a flow of natural goods such as oil, fish, wood or drinking water, but the flow of natural goods also includes ecosystem services (Van der Hamsvoort, 2006).

Traditional economic theory defines capital as labor, land and human-made capital. Human made capital or manufactured capital are physical items, such as tools and buildings used in the production process. Land could cover any parts of natural capital, but for the most part only land itself was seen as in limited supply, the rest of nature was assumed to be limitless. In its representation of production functions, neo-classical economics omits land and only focuses on labor and capital. With the increase in environmental awareness, some production functions have been extended to include energy and material inputs as well (Ekins et al., 2003b). Hence, nowadays capital is increasingly defined in a more broad sense, meaning any economically useful stock (Pezzey, 1992*a*). Perman et al. (2003) distinguish four different capital forms: (i) natural capital as any naturally provided stock[4], (ii) physical capital, (iii) human capital as stocks of learned skills in particular individuals and, (iv) intellectual

[4]Natural capital refers to the various ways that the environment powers production and supports most aspects of human existence (Costanza et al., 1991)

capital as disembodied skills and knowledge. In this way any economically useful stock is included. In fact, natural capital provides a major extension of the concept *land*, one of the classical factors of production in economic theory (Ekins et al., 2003a). Note that when we speak of natural capital or human capital, we might imply that nature, and human beings, are important only as productive resources. The term human capital is not intended to be a synonym for human beings, and natural capital is not everything we care about in nature. The capital terms refer to much more limited subsets of the broader concepts with which they are linked (Goodwin, 2003). Most economic activity involves the direct or indirect use of common-property environmental resources that are transformed from a natural state to a degraded condition (Considine and Larson, 2006). Ayres and Kneese (1969) argue that material inputs should be defined more broadly, considering a material balance principle. van den Bergh (1999) states that a neoclassical production function is not necessarily inconsistent with mass balance, but it provides little information on what types of mechanisms de-link the value of output from material inputs.

Natural capital has become a foundational concept of economics over the past decade or more (England, 2006). Two broad types of natural capital can be differentiated. Nonrenewable natural capital refers to the fossil fuel and mineral deposits that do not renew themselves on a time scale close to the rate that human use them, while renewable natural capital (e.g., trees) is active and self-maintaining using energy from the sun and the Earth's core (Ayres et al., 1996). Note that other classifications are possible. For example, Smith (2004) distinguish four categories of natural capital: renewable and nonrenewable natural capital, land and ecosystems[5]. Ekins et al. (2003b) found that natural capital[6] is a complex category of four different environmental functions: (i) the provision of resources for production, (ii) the absorption of wastes from production and consumption, (iii) the provision of basic life-support functions, such as those producing climate and ecosystem stability and shielding of ultraviolet radiation by the ozone layer, and (iv) the provision of amenity services such as the beauty of wilderness.

To assess sustainability, a much broader interpretation of the concept of capital than the one traditionally used by economists, is needed (Dyllick and Hockerts, 2002). Pfeffer and Salancik (1978) define a resource as those means that an organization needs in order to survive. In fact the core argument of their resource dependency theory states that (i) organizations will respond to demands made by external actors or organizations upon whose resources they are heavily dependent; and (ii) organizations will try to minimize that depen-

[5]Smith (2004) sees a tree as renewable natural capital while a forest is a ecosystem. For example a forest can be subject to qualitative degradation (e.g., harvesting trees, acid rain) or subject to quantitative degradation through human activity (e.g., conversion of forests into urban land)

[6]Ekins et al. (2003b) use the notion of ecological capital to define natural capital

dence when possible (Pfeffer and Salancik, 1978, Pfeffer, 1982). Cairns (2005) defines capital in a similar way: capital is any good, even an abstract one such as knowledge or environmental quality, about which decisions are made over an interval of time in order to contribute to some underlying purpose or objective. Frooman (1999) even states that the resource dependency theory defines a resource as essentially anything an actor perceives as valuable. In the language of traditional strategic analysis, firm resources are strengths that firms can use to conceive and implement their strategies to improve their efficiency and effectiveness. Firm resources include all assets, capabilities, organizational processes, information, knowledge,... (Barney, 1991). Physically speaking certain environmental aspects are (undesired) outputs rather than inputs. Because companies need to be able to emit pollutants to be able to produce value added, these environmental aspects can be seen as inputs from an economic point of view (Figge and Hahn, 2005).

To summarize, the capital approach borrows the concept of capital from economics, but broadens it in a variety of ways to incorporate more of the elements that are relevant to the sustainability of human development (Giovannini, 2004).

3.2 Weak sustainability

In 1972 Meadows et al. (1972) built a world model to investigate several major trends of global concerns. They found that if the present growth trends in world population, industrialization, pollution, food production, and resource depletion continue unchanged, the limits to growth on this planet would be reached sometime within the next one hundred years.

Responding to *Limits to Growth* Dasgupta and Heal (1974) wrote a classical paper: *The Optimal Depletion of Exhaustible Resources*. In this paper, natural resources are finite, nonrenewable, and essential to production instead of being ignored, as they had largely been until then in economic growth theory (Pezzey and Toman, 2002*a*). Dasgupta and Heal (1974) explored some of the immediate implications of the existence of exhaustible resources. They found that after perhaps an initial peak, consumption and utility approach zero in the very long run. This is the direct consequence of a positive utility discount rate, combined with the inherent scarcity of the nonrenewable resources. In fact, discounting imposes an inherently persistent tilt to consumption choices that undermines the ability of the economy to grow sustainably (Pezzey and Toman, 2005). Stiglitz (1974) stated that one way to avoid this undesirable outcome is ongoing technical progress. Adding the possibility of exogenous technical progress increases the productivity of the natural resource and thereby offsets its increasing natural scarcity. Solow (1974) found that the solution to Dasgupta and Heal's problem is a moral one. Solow's analysis is roughly a mirror of what Dasgupta and Heal did, since Solow presumed that utility should be sustained over time. In his research, Solow examined the conditions under which this is technically feasible (Pezzey and Toman, 2005). He found that such conditions exist[7]: early generations are entitled to draw optimally down the finite pool of resources so long as they add (also optimally) to the stock of reproducible capital. Hartwick (1977) formulate this as: to achieve constant consumption over time, society should invest in reproducible capital precisely the current returns from the use of flows of exhaustible resources. This Hartwick rule or savings-investment rule (Hartwick, 1978) has come to be known as a weak sustainability approach. Dixit et al. (1980) reformulate the Hartwick rule as *keep the total value of net investment under competitive pricing equal to zero* or *keep the present discounted value of total net investment under competitive pricing constant over time.* This generalized Hartwick rule is necessary and sufficient for constant utility, in other words the *Dixit-Hammond-Hoel* rule is a necessary condition for intertemporal equity along competitive paths (Buchholz et al., 2005).

Note that not only Hartwick's rule holds in the model considered by Solow (1974), the converse of Hartwick's rule holds as well. If consumption remains constant at the maximum sustainable level, then in value the accumulation of man-made capital always exactly compensates for the resource depletion (Hamilton, 1995). A general proof of the converse Hartwick's rule, namely that in an economy with stationary instantaneous preferences and a stationary technology an efficient constant utility path is characterized by the value of net investments being zero at each point in time, has been given by Withagen and Asheim (1998) and by Mitra (2002). In other words, for competitive paths which are both equitable and efficient, Hartwick's rule must hold. In contrast to this literature, the rather demanding assumption of efficiency of these paths is irrelevant in the context of the exhaustible resource model in which Hartwick first proposed his rule. Buchholz et al. (2005) show that in the context of this model, competitive paths which satisfy the *Dixit-Hammond-Hoel rule* (that the value of net investment be constant) must also satisfy *Hartwick's rule* (that the value of net investment be zero). Considering the following three conditions that a feasible path may satisfy: (i) it is competitive, (ii) is is equitable and (iii) it satisfies Hartwick's rule; Buchholz et al. (2005) show that if the path satisfies any two of these three conditions, it must also satisfy the third.

Solow (1986) showed that the Hartwick rule can be interpreted as saying that an appropriately defined stock of capital, including the initial endowment of resources, is being maintained intact, and that consumption can be interpreted as the interest on that patrimony[8]. Asheim (1986) showed that the Hartwick rule

[7]at least given the simple assumptions of the economic model of Solow (1974)

[8]This result assumes a constant interest rate and does not apply to the economies

does not apply to open economies, since the underlying stationary technology assumption is violated when gains from trade are taken into account. Therefore, Asheim (1986) developed and applied an analog to the Hartwick rule for open economies to a model of capital accumulation and resource depletion. The treatment of capital gains arising from exogenous changes in prices of extracted resources within the basic net-investment rule of an open economy is discussed by Vincent et al. (1997). Hartwick and Van Long (1999) show that even with time-dependent technology and terms of trade for constant consumption, the accumulation of one stock must exactly compensate for the aggregate decumulation, in value terms, of all other stocks provided that the rate of interest is time-invariant. In the case of a time-dependent rate of interest the Hartwick rule becomes: depending on whether the rate of interest is falling or rising, investment in a sinking fund must over-compensate or under-compensate for the aggregate decumulation of other stocks to ensure constant consumption.

Although an economy with constant utility over time must satisfy the Hartwick rule, observing that investment currently happens to be equal (or greater) to the resource rent measured at market prices does not imply that al least the current level of utility can be maintained by imposing Hartwick's rule from now onwards. This is because an economy which is depleting its natural resources too fast (for sustainability) will drive resource prices and hence resource rents too low, and thus investment at such a level does not ensure sustainability (Toman, 1994). This has been pointed out by Asheim (1994), Pezzey (1994), Vellinga and Withagen (1996), Pezzey and Withagen (1998), and Asheim et al. (2003). Asheim (1994) emphasizes that it seems impossible to develop the Hartwick rule into an indicator of sustainability, even if prices for the valuation of natural and environmental resources were readily available through a perfect intertemporal equilibrium. A correct indicator for permanent sustainability would be resource rents measured by shadow prices which reflect the sustainability constraint, which includes the constraint of the current resource stock (Toman, 1994). In other words, although the result proven by Hartwick (1977) is undoubtedly correct, it does not follow that one can draw a close link between Hartwick's result and intergenerational equity without taking notice of additional conditions: the Hartwick rule does not indicate sustainability. The Hartwick investment rule cannot serve as a prescription for sustainability but rather as a description of an efficient path with constant utility (Asheim et al., 2003). Therefore Pezzey (2004) derived two one sided tests for the unsustainability of an economy. If the value of net investment is momentarily zero or negative, or if green net national product is momentarily constant or falling, then at that moment the economy is unsustainable, meaning that its current level of utility cannot be sustained forever. Pezzey (2004) calls this test one-sided because it shows only unsustainability, not sustainability. He furthermore admits that these two theoretical sustainability tests

of Dasgupta and Heal (1974) and Solow (1974). So Solow (1986) does not address the policy conflict between a present value optimal economy and the imposition of Hartwick's rule (Pezzey and Toman, 2002*a*).

show practical difficulties, because (i) values must be estimated for all significant environmental resources, (ii) the price of produced consumption must be estimated over time, and (iii) estimations of future, exogenous changes in production possibilities are needed. Therefore, Pezzey (2004) hopes that advances in theory that are impractical when first proposed stimulate developments in measurement that make them more workable. The Hartwick rule does not require substitutability between man-made and natural capital. The question of whether man-made capital can substitute for natural capital is important for the relevance of the Hartwick rule for sustainability only to the extent that a lack of such substitutability means that eventual productivity cannot be satisfied (Asheim et al., 2003). Note that the Hartwick rule is an important starting point to measure sustainability. A literature overview of the progress of the measurement of performance and sustainability can be found in chapter 4.

Krautkraemer (1985) examined a dynamic model in which the productive value of the resource is explicitly determined within the model and in which the amenity values associated with preserved environments are taken into account. In fact, the concern about the loss of amenity values when preserved natural environments are disrupted by the extraction of productive resource inputs raised by Krutilla (1967) is considered. Krautkraemer (1985) found that the ability of the economy to maintain the effective supply of a nonrenewable resource input and to prevent the decline of production and consumption is a necessary but not sufficient condition for the optimality of permanent preservation of natural environment. In addition, the optimal level of permanent preservation will depend upon the initial endowment of capital and resource stocks. Hence, the amenity value of natural environments does lead to greater conservation of the resource. In other words the recreational, aesthetic and scientific amenity services provided by preserved natural environments increase the opportunity costs of extracting resources from the environment (Krautkraemer, 1985).

Weak sustainability can be seen as a different name for Hartwick-Solow's rule expressed in the form of maintaining total capital stock. Hence, the concept of weak sustainability can be seen as a by-product of growth theory with exhaustible resources if[9] (i) the definition of sustainability is restricted to non-declining consumption per capita and (ii) the environment-economy relationship is restricted to introducing an aggregate input called *natural capital* into the production function, with no special treatment for such input except for its existence in limited quantity (Cabeza Gutés, 1996).

Three factors which allow an economy to overcome the scarcity of an essential, nonrenewable resource can be identified: (i) the substitution of other factors of production for the resource input, particularly a reproducible capital stock,

[9] Note that the concept of sustainability arose from a much broader concern about the conflicts between economic activity and the environment, with special emphasis on inter- and intragenerational equity (Cabeza Gutés, 1996)

(ii) technological progress in the manufacture of commodities, and (iii) increasing returns to scale (Krautkraemer, 1985). Hence the first assumption in deriving the concept of weak sustainability is that of a high degree of substitution between natural and man-made capital. Substitutability refers to the capacity to alter production and consumption activities in the event of increasing scarcity of some resource in order to maintain a desired overall flow of services (Norton and Toman, 1997). Proponents of the weak sustainability view assume unlimited substitutability (i) of one type of productive input for another[10], (ii) of one source of instantaneous utility for another and (iii) of utility at one time for utility at another time (Pezzey and Toman, 2002*b*). The substitution issue goes beyond substituting technological progress (human and knowledge capital) or investment (built capital) for depletion of mineral and energy resources. Substitution also involves the ability to offset a diminished capacity of the natural environment to provide waste absorption, ecological system maintenance, and aesthetic services.

Apart from the assumption about the degree of substitutability, a second important offsetting force to the limits to growth, introduced by the presence of exhaustible resources, is that of technological change (Cabeza Gutés, 1996). Technological change can result in an increase of efficiency and can either reduce or replace the inputs necessary to produce goods and services. In this way, technology makes it possible to exceed the material limits of natural resources by substituting inputs if resources are depleted or if productivity limits are reached (O'Hara, 1998).

Old growth theory (e.g., Stiglitz (1974)) focused on exogenous technological change, while new growth theory explicitly allows for the endogeneity of technological change (Vollebergh and Kemfert, 2005). Endogenous technical progress results from people's and firms' economic decisions to invest in accumulating human capital and knowledge or improved product quality (Pezzey and Toman, 2002*b*). Sustainable balanced growth[11] is feasible and optimal if substitution effects[12] offset exactly the income effects due to the growth in productivity (Bovenberg and Smulders, 1995). Vollebergh and Kemfert (2005) observe several theoretical papers that succeed in showing how the fundamental mechanisms behind what is called directed technological change may at least postpone absolute scarcity issues, with small effects on economic growth under some reasonable assumptions. Furthermore, directed technological change conveys a positive message: shifting away from polluting towards non- or less-polluting technologies seems both possible and manageable through environmental pol-

[10] Intimately related to the subject of substitution between natural and man-made capital is that of input aggregation (Cabeza Gutés, 1996). Natural resources play radically different functions within the economy.

[11] Note that in this view growth incorporates also qualitative aspects and not only quantitative aspects as defined in section 2.2

[12] Substitution away from environmental services toward consumption and the input of man-made factors of production

icy. In fact, sustainable economic growth is theoretically feasible if it is of a qualitative and not quantitative nature. It requires that no upper limit to knowledge accumulation and no lower limit to resource intensities exist and that the market mechanism provides the necessary incentives to overcome natural resource scarcities. (Pittel, 2002). Nevertheless, Pezzey and Toman (2002*b*) state that the conclusions from endogenous growth economics are essentially unchanged from the classic results and so far, it sheds no new light on the ultimate limits to neoclassical substitutability assumptions.

To summarize the weak sustainability approach, we can say that with respect to natural capital as an input into the production of consumption goods, proponents of weak sustainability hold that (Neumayer, 2003): (i) natural resources are super-abundant, (ii) either the elasticity for substituting man-made capital for resources in the production function is equal to or greater than unity, even in the limit of extremely high output-resource ratios; (iii) either technical progress can overcome any resource constraint. Weak sustainability is a paradigm of resource optimism. Therefore, Neumayer (2003) calls proponents of weak sustainability environmental optimists. Those neoclassical economists assume that every technology can be improved upon and every barrier can be surmounted or broken through (Ayres, 2007). This has been called *the age of substitutability* (Goeller and Weinberg, 1978).

The neoclassical approach to sustainability assumes that the goal of policy intervention is generalized present value maximization, subject to a sustainability constraint[13] or modified by a public sustainability concern. Therefore, Pezzey and Toman (2002*b*) state that the basis of the neoclassical sustainability economics is distinct from classical utilitarianism, neoclassical utilitarianism and rights-based view: it rejects classical utilitarianism which prohibits any discounting; it rejects neoclassical utilitarianism which sees maximizing present value as a complete prescription for intertemporal equity; and it rejects the purely right-based view that it is the future generations' resource opportunities, not utility outcomes that matter. Hence, Beckerman (1994) is wrong in saying that weak sustainability offers nothing beyond traditional welfare maximization, because the weak sustainability approach maximizes present value under a sustainability constraint (Neumayer, 2003). Pezzey (2004) emphasizes that it remains a paradox why sustainability should be of interest in a present-value-maximizing economy. The goal of policy intervention within the neoclassical sustainability approach is present value maximization subject to a sustainability constraint. But, individuals must in fact believe there will be no policy intervention in favour of sustainability, or else they would modify their plans for the future, causing prices today not to be present value optimal (Pezzey, 2004). Possible solutions to this paradox can be found in alternative criteria (e.g., Asheim et al. (2001)) explained in section 3.5. Another

[13] An example of such a constraint is that the capacity to provide non-declining utility must be maintained at any point in time (Neumayer, 2003)

solution is suggested by Pezzey (2004): individuals choose their actions to maximize some form of present value, but vote for a government which applies a sustainability concern. This solution does not require an explicit concern for equity but makes a split between private and public concerns about the future as suggested by Marglin (1963).

Common and Perrings (1992) see the Solow/Hartwick interpretation as the economic notion of sustainability. In contrast, the ecological notion involves resilience, conceived as stability of the parameters defining an ecological-economic system. Common and Perrings (1992) argue that while it is not necessary to sacrifice the intertemporal efficiency required by a Solow/Hartwick interpretation of economic sustainability in order to assure ecological sustainability, intertemporal price efficiency is not a necessary condition for ecological sustainability.

3.3 Strong sustainability

Daly (1990) formulates some operational principles of sustainable development for the management of renewable resources: (i) harvest rates should be equal to regeneration rates (sustained yield), (ii) waste emission rates should equal the natural assimilative capacities of the ecosystems into which the wastes are emitted, and (iii) the rate of depletion of renewable energy sources should be limited to the rate of creation of substitutes for these renewable resources. Daly (1990) states that manmade and natural capital are basically complementary and only very marginally substitutable. Therefore, Daly can be seen as an important architect of the *strong sustainability* view that capital-resource substitutability is very limited, because he stresses the importance of the sustenance of specific resource sectors (Pezzey and Toman, 2002*a*). The ideas of Daly (and others) are derived from the work by Nicholas Georgescu-Roegen[14]. In his groundbreaking work, *The Entropy Law and the Economic Process*, Georgescu-Roegen (1971) elaborates on the implications of the entropy law for economic processes and how economic theory could be grounded in biophysical reality. Waste does not just disappear out of the system as conventional economics assumes; it has to accommodate somewhere. This first law of thermodynamics (*the conservation law*) implies the mass-balance principle (Ayres and Kneese, 1969). As in Boulding (1966), the implications of Georgescu-Roegen (1971) are to use low-entropy energy stocks as efficiently as possible if they are in short supply. Furthermore, if we are ultimately to run out of stock of low-entropy materials, we should in the meantime prepare by adapting economic systems to use the fixed flows of solar energy that will remain available (Edwards-Jones et al., 2000).

[14] An overview of the contribution of Nicholas Georgescu-Roegen to ecological economics can be found in Cleveland and Ruth (1997)

In other words, this second law of thermodynamics (*the efficiency law*) implies that a minimum quantity of energy is required to carry out the transformation of matter (Stern, 1997). The first law is widely accepted but the application of the second law of thermodynamics to economics results in much more controversy and confusion (Pezzey and Toman, 2002*b*). Nicolas Georgescu-Roegen and Herman Daly state that this law implies that our civilization is totally dependent on a finite stock of high quality (low entropy) resources stored in the Earth's crust. Because recycling materials requires low entropy, and materials can never be recycled with 100% efficiency, our economic system is doomed to run down as the low entropy materials are used and become unavailable. Robert Ayres shares the concern that we are currently using up our resources much faster than they were originally produced, but he rejects the notion that entropic dissipation of materials is an inherent limit to growth. He emphasizes that the Earth is not a closed system because it receives solar energy, and as long as there is an adequate flux of available energy perpetual motion machines are not impossible. Waste will accumulate over time in a storehouse or *wastebasket* and given the availability of energy, there is no barrier to treating this wastebasket as an ore pile and recovering material from it. Due to the second law of thermodynamics, there will always be waste from the recovery process itself but this waste goes back into the wastebasket and as the waste pile is big enough it is possible to compensate for the losses (Ayres, 1996, 1997, 1998, 1999, 2007). Nevertheless Ayres (2007, p. 127) advocates strong sustainability:

> I have to reiterate that, while there is plenty of room for substitution and some possibility of major breakthroughs; the pessimists, those who espouse the notion of *strong sustainability*, appear to be closer to the truth than the optimists who believe in more or less unlimited substitution possibilities.

Krysiak (2006) tried to provide a rigorous and general proof of the relevance of physical constraints for economic analysis. He showed that in a static setting for economies containing irreversible processes, a non-zero resource input as well as non-zero emissions are necessary to sustain a positive level of consumption. In a dynamic setting, these physical constraints imply that more of a good with non-vanishing marginal entropy production always necessitates more resource use (Krysiak, 2006). Mark that these results indicate only that limits to growth for the production of most physical goods are likely to exist, they do not quantify these limits. Therefore it is not clear if these limits will be met in the (near) future.

Baumgärtner (2003) sees the entropy concept as one of the cornerstones of ecological economics because is has the potential to establish relations between the natural world and purposeful human action. In other words, physical laws are seen as limiting the extent to which other resources can be substituted for scarce natural resources or ecological degradation. In addition, because matter is conserved, waste is an inherent part of any economic activity, and natural limits may constrain the capacity of the environment to process these wastes (Ayres and Kneese, 1969, Toman, 1994). Proponents of the strong sustainability approach derive arguments from natural science that substitutability is an impossible concept. They see capital and natural resources rather as complements than as substitutes, or subject to strict and fairly imminent limits (Pezzey and Toman, 2005).

Another argument concerning the limits to substitution is that some forms of natural capital are not replaceable by produced capital, at least beyond certain minimum stock sizes. This is because these stocks may provide life-support services to the economy or represent pools of irreplaceable genetic information or biodiversity (Stern, 1997). Therefore, basic life support systems[15] are almost certainly impossible to substitute (Dietz and Neumayer, 2007). Human beings cannot live in highly degraded natural environments, even with a degree of material wealth. This wealth cannot undo the direct threats to well-being through illness or injury, the disruptions of natural systems and processes (O'Connor, 1993, Norton and Toman, 1997, Pezzey and Toman, 2002*b*). Besides this, the loss of some natural capital may be irreversible (Dietz and Neumayer, 2007).

Yet another argument for the existence of substitution limits is that the actual process of production offers fewer changes for altering inputs than is assumed in neoclassical constructs (Faber et al., 1999, Pezzey and Toman, 2002*b*), because for a given choice of technology, input proportions are largely fixed (in the short term).

A next reason to pursue strong sustainability is that there remains considerable risk, uncertainty and ignorance attached to the way in which natural systems such as the global carbon and biogeochemical cycles work. This means we cannot be sure what effect damaged natural systems will have (Dietz and Neumayer, 2007).

An argument that has been noted only rarely is the finite amount of human information processing capacity (Pezzey, 1992*b*, Pezzey and Toman, 2002*b*). It is not clear whether the human brain can capture the knowledge needed to stay abreast of materials and energy dissipation by adapting new technologies.

Although these last propositions are not as fundamental as those based on thermodynamics and are largely empirical questions, they may nevertheless be just as important as thermodynamics in constraining actual production (Stern, 1997). Furthermore, there is also an ethical argument for non-substitutability, which posits that increased future consumption is not an appropriate substitute for natural capital losses (Dietz and Neumayer, 2007).

[15] An example of a basic life support system is the ozone layer shielding ultraviolet radiation (Ekins et al., 2003b)

Advocates of strong sustainability argue that traditional neoclassical models overestimate the possibilities of substitution between natural and manufactured capital including related problems of complementary, irreversibility, pure uncertainty and discontinuous change (Daly, 1995, Gowdy, 2004, 2005). According to the neoclassical theory, a market economy is an atomistic isolated entity which is self-regulating and self-sustaining. This takes no account of (i) thermodynamic sources and their depletion, (ii) the interdependence of materials, energy, and environmental support structures, (iii) the limits of the environmental systems (iv) nor of the contributions and limits of social systems (Christensen, 1991). Gowdy (2005) argues that the weak sustainability solution to the intergenerational welfare problem, separating capital stock from the output it produces, does not work unless an assumption is made that either unlimited substitution among different kinds of capital is possible or that money is a universal substitute for anything. But the value of capital, natural or otherwise, cannot be defined independently of output: it is not true that with a given stock of capital, anything at all can be produced (Gowdy, 2005).

Note that no advocate of strong sustainability argues for preserving all stocks. This would lead to the absurd implication that no depletable resource should ever be touched (Pezzey and Toman, 2005). Rather, proponents of strong sustainability argue for maintaining the key functions of natural resource stocks (Common and Perrings, 1992).

Howarth (1997) argues that substitution between natural resources and human-made capital are only defensible if they benefit both present and future generations. However, identifying such *Pareto improvements* is feasible only if the relevant costs and benefits can be quantified with sufficient accuracy. In other words, the unique properties of real world commodities, assets and resources mean that they are imperfect substitutes in consumption and production, though the nature of these properties range from intrinsic spiritual values to thermodynamic properties (Stern, 1997). Limits on material outputs need not limit the value of economic activity. Hence, it is possible to have economic growth in the sense of providing better and more valuable services to ultimate consumers, without necessarily consuming more physical resources (Ayres, 1996).

Besides substitution possibilities, a second important aspect in the sustainability discussion is technical or technological change. Even if substitution possibilities were limited, constant utility could be maintained forever if technical change improves natural capital (Ayres et al., 1996), or in other words, if technical change would sufficiently increase the productivity of natural capital. Proponents of strong sustainability argue that even continuous technological change will not change their pessimistic outcome (i.e. future consumption will finally fall to zero). Several reasons justify warnings against an overly optimistic view (Vollebergh and Kemfert, 2005). First, not all our environmental resources can be preserved equally effectively by technological change[16]. Second, sometimes more radical changes are needed, and these fundamental changes require a transition of a whole system. Third, learning-by-doing is not entirely free, diffusion of knowledge can be costly and our understanding behind this process is rather limited (Vollebergh and Kemfert, 2005). Finally, with respect to the role of technological change in discussing sustainability several questions arise (Ayres et al., 1996): (i) the problem of empirically distinguishing between the substitution among resources within a given technology and technological change; (ii) will technological change follow the *right* direction?[17]; and (iii) the fact that the positive effect that technological change might have in offsetting the limits to growth by exhaustible resources cannot by analyzed without considering the negative feedback that the new technologies might have on the environment (Cabeza Gutés, 1996). These arguments temper the technological optimism of adherents of the strong sustainability view.

The question of physical scale is central in the debate on substitution possibilities: if substitutability is relatively easy, then the total scale of human activity relative to the natural environment is of limited significance relative to the efficient use of resources and, depending on one's ethical perspective, the adequacy of society's total savings for the future (Toman, 1994). Microeconomic analysis indicates that substitution can allow economic activity to continue, but these analyses ignore the macroeconomic and global effects of substitution (Ayres et al., 1996). Furthermore, a larger overall scale of throughput should imply more possibilities, because the range of possible activity types increases. However, a larger throughput for a particular process may make substitution more difficult by further damaging fixed ecological factors (Pezzey and Toman, 2002*b*). Daly (1991*b*) emphasizes the importance of the proper scale as a third independent policy goal after the goals of efficient allocation and just distribution. Optimal allocation of a given scale of resource flow within the economy is one thing, while optimal scale of the whole economy relative to the ecosystem is an entirely different problem: *Optimally loaded boats will sink under too much weight, even though they make sink optimally!* (Daly, 1991*b*) (see also section 2.1). Lawn (2001) states that the policy instrument of the present price system is solely concerned with the goal of allocation. Hence the market is very effective at revealing relative scarcities but sustainability is a question of absolute scarcity. To achieve an optimal level of resource throughput in the economy, Aubauer (2006) suggests to use the distribution of certificates. All three goals can be met if certificates are handed out to consumers and this will solve the resource distribution conflict between generations and between persons of the same generations (Aubauer, 2006). Therefore, proponents of the strong sustainability approach make a

[16]For example our consumption of nature and its associated environmental good biodiversity

[17]If prices are far from reflecting true scarcities (for example due to market failures), then nothing will ensure that economic resources will be invested in developing technologies that are biased into saving natural capital (Cabeza Gutés, 1996)

more clear distinction between local and global impacts. Local resource depletion and ecological degradation, while often having serious consequences, may be more easily compensated for by economic diversification, trade, and migration than regional or global adversities (Toman, 1994). When the probability of distant but potentially catastrophic environmental damage is admitted[18], the traditional existing economic approaches are the least capable of addressing environmental problems. In other words, the wider and more durable the environmental activities, the less is the scope for a market solution involving the property rights. In this class of problems (high fundamental uncertainty and potential costs) the precautionary principle is advocated (Perrings, 1991). This precautionary principle can be defined as the commitment of resources now to safeguard against the potentially catastrophic future effects of current activity (Perrings, 1991, Howarth, 1997). Closely related to the precautionary principle is the use of safe minimum standards. Under safe minimum standards, unique natural assets such as endangered species, and undisturbed communities and ecosystems are preserved intact, unless the costs of doing so are intolerably high (Randall and Farmer, 1995). The framework of the safe minimum standard, based on Ciriacy-Wantrup (1968) and Bishop (1978), has been criticized as poorly grounded and inconsistent with the notion of Pareto efficiency, but these critics make abstraction from the considerations of rights and fairness that are essential to political economy (Howarth, 1997). This will be further discussed in section 3.5.

3.4 Weak versus strong sustainability

The answer of Solow (1997) on the question *How much of a drag on the sustainability of current production, might be exercised by the limited availability of natural resources* depends on (i) the importance of natural resources as inputs into production, (ii) the ease or difficulty with which capital and renewable resources can substitute for nonrenewable resources; and (iii) the technological progress in the future (Solow, 1997). Proponents of both the weak and the strong sustainability paradigm have clearly a different view about (i) the substitution possibilities between resources and (ii) the possibilities of technological progress. Hence, proponents of weak sustainability belief that any natural resource can be substituted by another resource, or by man-made capital, or by technical progress, or by some combination thereof (Neumayer, 2003).

Whilst weak sustainability could be interpreted as an extension to neoclassical economics, strong sustainability calls for a paradigmatic shift away from neoclassical environmental and resource economics towards ecological economics (Neumayer, 2003). Advocates of the strong sustainability view note that the perfect substitution argument violates the law of the conservation of

[18] Remark that the probability of damage is often low but existing

matter. Proponents of the weak sustainability view counter that the class of growth models that include resources can account for these thermodynamics constraints with the *essentiality condition* (Ayres et al., 1996). In this way output is only zero when the resource input is zero, and strictly positive otherwise. Neoclassical models including the essentiality condition still suggest that a constant level of economic output can be maintained if the degree of substitution between resources and capital is sufficiently high (Solow, 1974, Ayres et al., 1996). This view is rejected by proponents of the strong sustainability view because it does not recognize the limits to the degree to which we can substitute for depleted resources and a degraded environment, and because it does not account for all the services provided by the environment (Ayres, 1996). Complementarity limits but does not exclude substitution. There still are many opportunities for mitigating resource depletion and environmental degradation through the substitution of human-made capital. The point of adherents of the strong sustainability view is that such opportunities are more limited than many people assume, and that the appropriate starting point is recognition of complementarity rather than the dubious assumption of near perfect substitution (Ayres et al., 1996). Proponents of strong sustainability can therefore be called pessimists while weak sustainability is rather a paradigm of resource optimism (Neumayer, 2003).

An important discussion point between proponents of weak and strong sustainability is the scale issue. Neoclassical economists (weak sustainability) favor marginal forms of analysis in practice and tend to pay less attention to the concept of the scale of an economy in relation to its resource base, while adherents of the strong sustainability paradigm believe that there are important thresholds of scale (Norton and Toman, 1997). Ayres et al. (1996) find that the substitution issue has been clouded by inattention to system boundaries. For example, elasticities of substitution between manufactured and natural capital calculated for firms or industries may reflect substitution possibilities at those scales. However, they may not reflect possibilities at another scale (e.g., national scale) because they do not account for the indirect natural capital costs of producing and maintaining manufactured capital.

van den Bergh (1999) finds the present distinction between processes and technological change too crude to deal with specific relationships between the various categories of production inputs, including materials. He argues for a disaggregate view on capital, substitutability and complementarity. Therefore he makes the distinction (i) between direct and indirect substitution, (ii) between stocks, flows, funds, and services; and (iii) between more concrete types of substitution and complementarity related categories of production inputs such as materials, energy, throughput (energy and materials), agents (economic funds), and capital (natural and economic funds). *Direct substitution or replacement* refers to changes within a category of production factors that fulfill the same or a similar function in the production process. *Indirect substitution or saving* refers to a process involving multiple categories of production factors, which fulfill different, and often complementary functions in the production process. Therefore, indirect substitution is immediately related to an increase in the efficiency and productivity of the production process. Note that the addition of technical change does not alter this disaggregate substitution framework, but simply widens the choice spectrum of direct and indirect substitution, based on process or product innovation. In other words, casting environmental problems in terms of the substitution between natural and economic capital at the most aggregate level seems to neglect the essential differences between these factors of production (van den Bergh, 1999).

Spangenberg (2005) emphasizes that the substitution between two capital stocks makes no sense. It needs to be replaced by an approach where the impacts of all dimensions (economic, social and environmental) and their characteristics are considered to an explicit set of multi-dimensional criteria. Hence, instead of substitution decisions following a single criterion, a systematic or intuitive multi-criteria assessment of the trade-offs would be performed.

Neumayer (2003) concludes that both paradigms are non-falsifiable under scientific standards. Both rest on certain assumptions, hypotheses and claims about the future that are non-refutable. Therefore, Ekins et al. (2003b) finds that the choice between weak and strong sustainability should be an empirical rather than a theoretical or ideological matter. Similarly, Pezzey and Toman (2002*b*, p. 308-309) state that:

> Up till now, the weak sustainability view of substantial substitution and innovation possibilities seems to have been borne out by history. Whether it will be in the far future remains an open empirical question, that will require big advances in data and methodology to answer.

Hence, the pessimistic predictions might have failed because the concern has forced people to react in time and develop better technologies and social institutions. Furthermore, Neumayer (2003) states that to conclude that there is no reason to worry, because the pessimists have been wrong in the past, is tantamount to committing the same mistake the pessimists are often guilty of, that is the mistake of extrapolating past trends. In the future, problems (e.g., environmental problems) may take completely new and surprising forms. To solve these problems (towards more sustainability), not only advances in data and methodology but also an open-minded cooperation between different sciences such as ecology and economics will be required (Tahvonen, 2000).

3.5 Intergenerational equity and fairness

In this section, we address some contributions towards the sustainability literature concerning intergenerational equity or distribution. Criticizing the more utilitarian perspective on intergenerational distribution (proposed by the adherents of the weak sustainability paradigm), rights-based theories of intergenerational justice that result in stewardship obligations for the current generation to preserve options over time are emphasized within this literature. Pezzey and Toman (2002*b*) categorize proponents of this view within the strong sustainability view. Nevertheless we addressed the question about the assumptions of substitution and technical advance that undergird the neoclassical approach in the previous section 3.3, while the concerns about intergenerational equity are addressed in this section. These concepts put more emphasis on concepts of community, as opposed to the more individualistic concepts underlying utilitarian analysis (Pezzey and Toman, 2002*b*). Similar but stronger concerns are communicated by *environmental justice* advocates. They go beyond *fair share* principles, which maintain that every community should have an equal share of environmental goods and bads, regardless of the race or class of its population (*distributional justice*). They argue that environmental bads should be eliminated at the source (*procedural or process justice*)[19].

Criteria for intergenerational equity can be justified by choosing a set of basic axioms that the agent's utility path over time must satisfy (Pezzey and Toman, 2005). Many economists defend the criterion of maximizing the present value of utility as an acceptable reflection of intergenerational equity, based on an axiomatic approach in Koopman (1960). However, others criticize in varying degrees of constant discounting as very inequitable to the far future (Pezzey and Toman, 2005). A well-known example is the constant utility path with utility as its maximum sustainable level first explored by Solow (1974), after Rawls (1971) *maximin* criteria. Dasgupta (1974) showed that Rawlsian maximin solution may be inadequate for problems of intergenerational justice, because under some specific utility functions exhibiting altruism towards future generations, plans will not be continued by future generations. These plans however, are optimal from the standpoint of a given generation. Also, the analysis of Calvo (1978) has left no doubt that time inconsistency is a definite possibility when individuals are altruistic towards future generations. Both papers show that the validity of the maximin principle is dependent on both utility and technological functions.

Besides the ulitarian and maximin criteria, a number of alternative criteria, and the axioms supporting them can be found in the work of Howarth (e.g., Howarth (1995)), Pezzey (e.g., Pezzey (1997)), Chichilnisky (e.g., Chichilnisky (1996)), Asheim (e.g., Asheim (1996*a*) and Asheim et al. (2001)) and many others.

Chichilnisky (1996) and Chichilnisky (1997) proposes two axioms that capture the idea of sustainable development and characterize the sustainable preferences that they imply. These axioms require that neither the present nor the

[19]More information about bringing together sustainability, environmental justice and equity can be found in Agyeman et al. (2003)

future should play a dictatorial role. Other criteria used in literature (e.g., discounted utility, Rawlsian and basic needs criteria) did not satisfy the suggested axioms and are not adequate to analyse sustainability (Chichilnisky, 1996). On the other hand, Asheim (1996*a*) evaluates different criteria of intergenerational justice in the model of capital accumulation and resource depletion as in Dasgupta and Heal (1974) and in Solow (1974). Asheim (1996*a*) found that the Chichilnisky criterion fails by not yielding existence. The Ramsey (classical utilitarianism)[20], the Solow (Rawlsian maximin)[21], the Dalton (modified principle of transfers)[22], and the Calvo criteria[23] lead to non-decreasing consumption. However the Ramsey criterion is unfair to the least fortunate first generation. Furthermore, the Solow criterium tends to perpetuate poverty and do not respect the altruistic concern that parents may have for their children. Therefore, of the considered criteria, only the Dalton and Calvo criteria remain in the presence of resource constraints (Asheim, 1996*a*).

Since Koopman (1960) the view prevails that equity might be difficult to apply in the intergenerational context if there is an infinite number of generations. For example Diamond (1965) shows the impossibility of treating all time the same. An important conclusion of this literature is that the ordinary procedure for establishing effectiveness is blocked when efficiency and equity are postulated in the context of an infinite number of generations (Asheim et al., 2001). Pezzey (1997) challenges the validity of Koopmans' stationarity axiom, and Asheim et al. (2001) state that the impression suggested by this literature that generations cannot be treated equally, is exaggerated. They find that under certain assumptions equity combined with efficiency is compatible with effectiveness and can be used to justify sustainability in the following sense: only sustainable paths are ethically acceptable whenever efficiency and equity are endorsed as ethical axioms (Asheim et al., 2001). Asheim et al. (2001) provided a new and solid normative foundation for the consideration of policies intended to achieve sustainability (Woodward and Bishop, 2003).

Bromley (1989) and Howarth and Norgaard (1990) turned the discussion about sustainability in a different way by focusing on the intergenerational allocation of resource rights. Bromley (1989) emphasizes that the interests of the future generations are only protected by an entitlement structure that gives present generations *a duty* to consider the interest of the future. Future generations thus obtain a correlated right.

Howarth and Norgaard (1990) showed that efficiency in itself is inadequate to

[20]The Ramsey criterion is characterized by the requirement that any feasible utility transfer with negative cost should be undertaken

[21]The Solow criterion is characterized by the requirement that any utility transfer from a richer to poorer generation should be undertaken, no matter the cost of the transfer

[22]The Dalton modified principle of transfers can be stated as follows: any utility transfer with zero cost from a richer to a poorer generation should be undertaken

[23]Calvo (1978) applies the Rawlsian maximum on the subjective welfare, which amounts to ethical preferences that are in accordance with welfarism

ensure a socially desirable intertemporal equilibrium and that the initial distribution of property rights is of fundamental importance. It is important not to overlook potential improvements in social welfare achievable through the reassignment of property rights across generations. Efficiency is a necessary but not a sufficient condition for a socially optimal intertemporal allocation of natural resources. The problem of achieving optimality involves both choosing the appropriate intergenerational distribution of resource rights and fulfilling the conditions of perfect competition (Howarth and Norgaard, 1990). Hence, if development is not sustainable, it is because the institutions through which the present provides for the future have not evolved according to changes in social and economic structures, technology, and population pressure (Howarth and Norgaard, 1992). Using a general-equilibrium framework, Howarth and Norgaard (1992) showed that the valuation of environmental services and how society cares for the future are interdependent. So, incorporating environmental values per se in decision-making will not bring sustainability unless each generation is committed to transferring to the next sufficient natural resources and capital assets to make development sustainable. This makes that valuation techniques rooted in partial-equilibrium reasoning are still appropriate for small, local issues but the relationships between social goals and valuation indicates serious conceptual inadequacies in the analyses to date of global issues such as climate change. The linkages between the issues of intertemporal efficiency and intergenerational distribution in the analysis of climate change strategies are considered by Woodward and Bishop (1995), Howarth (1996) and Howarth (1998). Similarly, Burton (1993) argues that standard discounting practices confuse two issues: (i) intertemporal discount rates of members of the society, and (ii) intergenerational equity considerations. The distinction is of particular importance to natural resource and environmental management decisions since the benefits and costs of such decisions may occur extended periods of time (Burton, 1993). Concerns in policy debates about excessive long-term discounting can be reinterpreted as concerns about intergenerational resource allocation, without necessarily suggesting the need for massive intervention in capital markets to directly alter the discount rate (Pezzey and Toman, 2002*a*).

Toman (1994) developed a conceptual framework to consider how individualistic resource trade-offs might be balanced against social imperatives for safeguarding against large scale, irreversible degradation of natural capital. His framework is an extension of the logic of the safe minimum standard based on Ciriacy-Wantrup (1968) and Bishop (1978). The framework is a two-tier system in which standard economic trade-offs guide resource assessment and management when the potential consequences are small and reversible. However, these trade-offs are increasingly complemented or even superseded by socially determined limits for ecological preservation as the potential consequences become larger and more irreversible (Toman, 1994). Howarth (1995) states that a deontological approach to intergenerational fairness suggests that sustainability criteria should be imposed as prior constraints on the maximization of social preferences concerning the distribution of welfare between present and future generations. In fact, moral obligations to future generations are distinct from altruistic individualistic preferences for the well-being of future generations. This view implies a strong bias against actions that generate present benefits but impose the risk of irreversible future losses when scientific research would permit the effective resolution of uncertainty over generational time (Howarth, 1995). Solow (1993) defends more conventional reasoning on sustainability (Pezzey and Toman, 2002*a*). The intergenerational trade-off should be managed well and equitably, meaning that (i) the production is carried on efficiently and resource market inefficiency and environmental externalities are internalized, and (ii) each generation is allowed to favor itself over the future, but not too much (the discount rate should not be too large) (Solow, 1993). Howarth (1997) argues that attempts as described in Solow (1993) are compromised by the uncertainty of future technology and preferences and by the paradoxes that surround interpersonal welfare comparisons. Therefore Howarth (1997) suggest also to use a two-tier approach to policy analysis. In this setting, the property rights of future generations are defined in terms of a structured bequest package (as in Norton (1995)) that includes undiminished stocks of natural resources and environmental quality, just as principles of distributional fairness are employed in defining the initial endowments. Subject to these endowments, cost-benefit criteria may be employed to identify Pareto-improving resource allocations where the depletion of natural capital is appropriately compensated by investments in other assets (Howarth, 1997).

Krautkraemer and Batina (1999) illustrate the potential conflict between efficiency and equity. The social rate of time preference can be high enough that future generations are considerably worse off tha the current generation even if allocative efficiency is achieved. Different social welfare criteria have different implications for sustainability under different technological scenarios (Krautkraemer and Batina, 1999).

Page (1997) compares two approaches to the problem of achieving the social goals of sustainability and intergenerational efficiency. In the separated approach, policy actions with long-run environmental consequences are evaluated by benefit-cost analysis using discounting, besides the equity considerations are assessed. Finally, the two (and perhaps additional factors) are brought together in the overall decision. Page (1997) calls the alternative approach (a two tier one) the integrated approach. In this case, equity and efficiency analysis are done in an integrative way right from the beginning, and the equity analysis shapes the decision environment in which the efficiency analysis takes place. In fact, depending on how important the equity considerations are, the decision is addressed in one of two tiers of decision rules (Page, 1997). Page (1997) states that many social decisions are made trough legal and political institutions and not in markets or by marketlike benefit-cost criteria. Hence, as a society, we can choose to elevate sustainability to a more *constitutional* principle. However, the dividing line between social decisions to be made using individualistic economic criteria and those using more collective mechanisms is not clearly marked (Pezzey and Toman, 2002*a*). One more traditional approach to resolve the tension between economic criteria (benefit-cost analysis or present value maximization) and intertemporal concern is suggested by Pezzey (1997). He defends the possible use of different variants of sustainability as a prior constraint on present value optimality. In contrast to Beckerman (1994) and Dasgupta and Mäler (1995) who claims that present value optimality is the only right way to reflect society's intertemporal concern, Pezzey (1997) emphasizes the use of sustainability constraints (inequality constraints) within the present value maximization. On the other hand, Pezzey (1997) argues that is is neither easy, nor particularly desirable in practice, to define sustainability authoritatively as a constraint.

To conclude this short overview of intergenerational equity, we follow the appeal of Pezzey (1997, p. 460,464) towards more empiricism:

> To whom is unjustness manifest? To whose moral institutions is a consequence acceptable? Who decides what is a *primitive* axiom, and why is it so important?
> In every case the answer appears to be an economic philosopher from almost any letter of the alphabet. All of them choose different, though often overlapping, sets of axioms. (...) None of them refer to data. I do see great clarifying value in philosophical debate, but without empiricism I do not see how it can ever hope to be conclusive.(...)
> There will also remain an important role for philosophical inquiry into intertemporal concern, albeit a clarifying rather than a decisive role. No scientific data analysis can tell us what the concern *ought* to be. Even if economist succeed in measuring which axioms we do respect when we make intertemporal choices, philosophers will still be needed to put axioms into our heads in the first place.

3.6 Lessons learned

A way of describing different notions of sustainable development is discussing the substitutability between resources. Resources can be defined as all aspects that are relevant to the sustainability of human development. The weak sustainability view states that substitutability of human-made resources for natural resources is unlimited while the strong sustainability view state that substitutability is impossible or subject to strict limits. Proponents of weak sustainability belief that any natural resource can be substituted by another resource (e.g., natural or economic resource) or by technical progress. Adherents of the strong sustainability view are much more pessimistic about technical progress and substitution possibilities. The choice between strong and weak sustainability should be rather an empirical than a theoretical matter, because both views rest on certain assumptions and claims that are not immediately refutable.

Note that the present distinction between strong and weak sustainability is illustrative but not complete. In fact, a more disaggregated view on resources, technological progress and substitutability and complementarity is advisable. Furthermore, equity concerns (inter and intragenerational) are not considered in the discussion about resource substitutability but they are also essential for a sustainable development.

Nevertheless, the described conceptual framework (weak versus strong sustainability) can serve as a guideline to discuss existing methods to assess sustainability. Therefore, several existing empirical applications (and methodologies) will be discussed in the following chapter using this framework.

Chapter 4

Measuring performance

Don't measure yourself by what you have accomplished, but what you should have accomplished with your ability
—John Wooden

4.1 Introduction

In this chapter, we will first explain the most important economic measures to assess performance. However, these measures, by themselves or together, do not assure long term sustainability. There is a need to consider a fuller range of factors in assessing performance.

Performance can be seen as the way in which someone or something functions, operates or behaves. Mostly performance is seen as an objective phenomenon. However, the interpretations and measures of performance arise out of an interactive process among individuals and institutions (Thomas, 2004).

There are a lot of performance measurement systems, because there are different objectives (e.g., performance of public government, performance of sport athletes,...) and different methods.

Sustainability performance can be defined by the integrated achievement of social, environmental and economic performance. Other possibilities are defining sustainability performance in an intergenerational and/or intragenerational context. Schaltegger and Wagner (2006) remark that sustainability performance is often understood as performance concerning non-market issues such as environmental and social issues and thus excluding economic performance. These approaches are misleading and therefore we will start in this section with the traditional economic performance measurement before discussing several existing methods to assess sustainability performance on national and on firm level. In other words, we start with a short overview of methods to measure economic performance (section 4.2) before giving an overview of methods to measure (sustainability) performance (section 4.3).

4.2 Measuring economic performance

The most common framework to measure economic performance can be described as the efficiency - effectiveness framework. In plain words, efficiency is about doing things right and effectiveness is about achieving the right things. An effective organization realizes the stated objectives.

We will focus on the traditional economic performance measurement in terms of efficiency and productivity. But several other indicators are used to measure performance, such as profitability (e.g., Davidova et al. (2003)[1]), production costs, income, turnover, growth of turnover, growth of income,...

To assess economic performance three notions are essential: value added, productivity and efficiency. This because they give a good indication of the appropriate use of economic resources.

4.2.1 Value added

Value added is a very important indicator to measure the economic performance of a country, a region, a sector or a company. Macroeconomic activity is typically measured by Gross Domestic Product (GDP), the GDP of a country is defined as the market value of all final goods and services produced within a country in a given period of time. The first rigorous assessments of national income and wealth was done by Petty (1664). However the creation of modern national income accounting was prompted by (i) the economic crisis of the Great Depression in the 1930s, (ii) the political and military conflict of World War II and (iii) the emergence of Keynesian macroeconomic theory (Carson, 1975, England, 1995). Remind that Gross National Product (GNP) is the same as GDP except that it measures the monetary value of the goods and services annually produced by domestically owned rather than domestically located factors of production. Note that the value added indicator is often used on macro economic level but it can be used at all levels (macro, meso and micro).

[1]Several ratios can be used to measure profitability, as in Davidova et al. (2003): for example the private-cost-benefit ratio which is the ratio of the sum of paid and unpaid costs to total revenue

The core System of National Accounts (SNA) is a unifying framework for the economic statistics (United Nations, 1993). A well-known indicator derived from this system is the volume increase of gross domestic product. The 1993 System of National Accounts is a conceptual framework that sets the international statistical standard for the measurement of the market economy. It is published by the United Nations, the European Commission, the International Monetary Fund, the OECD[2] and the World Bank. It is broadly accepted, credible, internally consistent, and has a long established theoretical structure. The non-priced use of the environment in production and consumption processes remains largely uncovered within the SNA framework (de Haan, 2004) (see section 4.3).

Value added measures to which extent a company (or a sector) receives higher prices for their output than the prices that have been paid for the input used in the production process to produce that output (goods or services) (Ooghe and Van Wymeersh, 2003). Hence, value added can be defined as the value of total production minus the costs of intermediate use. Value added is then used to remunerate all production factors. In traditional economic analysis the production factors are labor, human-made capital (e.g., buildings) and land[3].

4.2.2 Productivity and efficiency

More important than the absolute value of the value added is the proportion of the value added to the use of the production factors (called *productivity*). Productivity can be defined as the ability displayed by production factors to produce (Thiry and Tulkens, 1989). In general, productivity is measured as the ratio between output (e.g. value added, total revenues,. . .) and the production factors that realized it. If only one production factor is considered, then productivity is called partial. Hence, the output of a company divided by its labor use indicates labor productivity. If all production factors are simultaneously assessed, then total factor productivity can be measured. Often, it is impossible to incorporate all production factors. Furthermore, to measure total factor productivity, value terms are necessary.

There exist several possible notions of productivity and efficiency. To avoid possible confusion and misunderstanding, we will explain and define the efficiency and productivity notions, as commonly accepted in production economics.

Farrell (1957) defines efficiency as the actual productivity of a company compared to maximum attainable productivity, measured by dividing the output

[2] OECD stands for the Organization for Economic Co-operation and Development

[3] Remind that as explained in section 3.1 that to assess sustainability a broader definition of the capital forms is needed than this three production factors

by the input. The efficiency and productivity concepts can be easily understood using a graphical illustration (figure 4.1). Within production economics, functions can be used to describe the production technology. In fact, the production function (or production frontier) describes the technical relationship between the inputs and outputs of a production process. A production function defines the maximum output(s) attainable from a given vector of inputs (Coelli et al., 1998). Figure 4.1 shows a production frontier (PM) with only one output and one input, which is clearly a simplification of a real production process. The productivity of the observations 'b', 'c' and 'd' is calculated by dividing the output by the input, indicated by the slope of the lines 'B', 'C' and 'D'. Observations 'b', 'c' and 'd' are situated on the production frontier and are technically efficient, while observation 'a' is technically inefficient. Using the same amount of input (Q_i), observation 'a' produces less output (Q_0a) then observation 'b' which produces more output (Q_0b). The (output-orientated) technical efficiency of observation a equals $\frac{Q_0a}{Q_0b}$. In fact, the technical efficiency of observation 'a' is the actual productivity of 'a' divided by the maximum attainable productivity (in this case the productivity of observation 'b').

Although observations 'b','c' and 'd' are all technically efficient, the productivity level differs, in this example observation 'c' reaches the highest productivity, observation 'c' is the point of optimal scale. Observation 'b' can increase its productivity by using more inputs and observation 'd' by using fewer inputs (improving the exploitation of scale economies). However, changing the scale of a firm can be difficult to achieve quickly. Technical efficiency and productivity can be given short-run and long-run interpretations (Coelli et al., 1998). Note that technical efficiency can be measured in terms of equiproportionate contraction of all inputs (input-orientated technical efficiency) and in terms of equiproportionate expansion of all outputs (input-orientated technical efficiency) (Kumbhakar and Lovell, 2000). The output- and input-orientated measures are equivalent measures of technical efficiency only when constant returns to scale exist ($i.e. f(\alpha x_1, \alpha x_2) = \alpha f(x_1, x_2)$).

Unit isoquants (e.g., 'K' in figure 4.2) can be used to illustrate the relation between two inputs for a given level of output. Figure 4.2 depicts all combinations between input X_1 and input X_2 for a fixed amount of output Y. Observations on the isoquant are technically efficient (TE), while observations above the frontier are technically inefficient. Observation 'a' uses too much inputs X_1 and X_2 to produce the output amount Y. In other words the technical efficiency (input-orientated) of observation 'a' equals 0b/0a. If information about prices is available and some kind of behavioral assumption (e.g., cost minimizing) is appropriate, then the allocative efficiency (AE) can be calculated. The allocative efficiency in input selection involves selecting the mix of inputs that produces a given quantity of output at minimum cost. In figure 4.2, all observations on the isocost line 'L' are allocatively efficient. Observation 'b' is technically efficient but it has an allocative efficiency lower than 1 (AE = 0c/0b). The combination of allocative and technical efficiency results in a

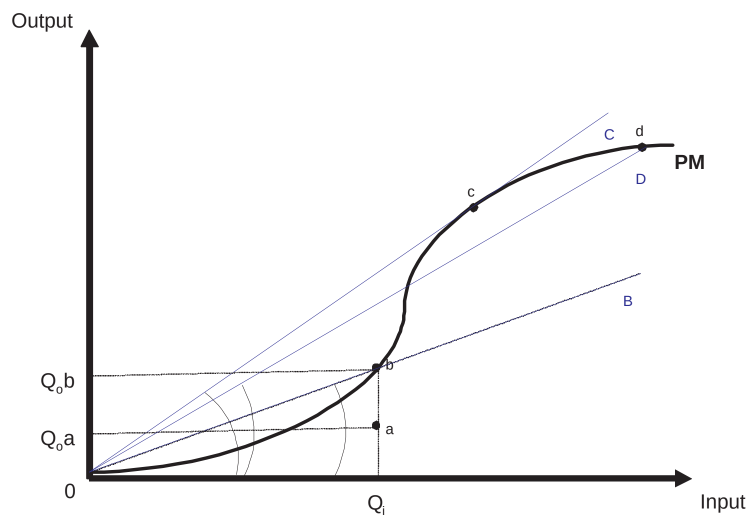

Figure 4.1: Production frontier, technical efficiency and productivity

measure of overall economic efficiency (EE). Only observation 'd' is economic efficient (figure 4.2). The economic efficiency of observation 'a' equals 0c/0a.

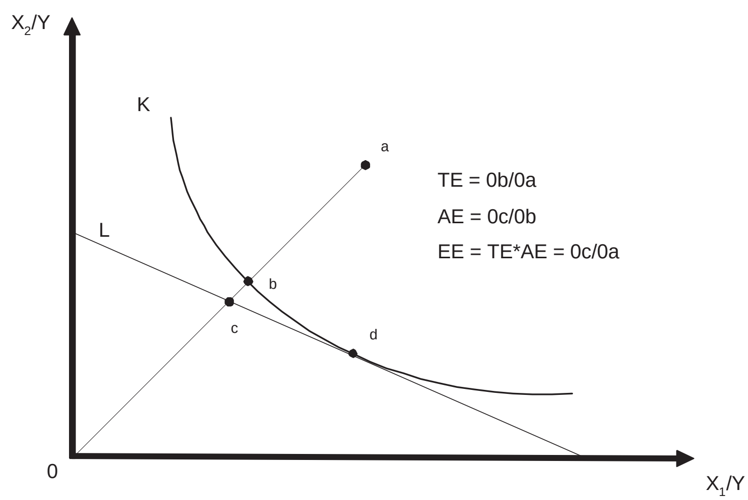

Figure 4.2: Unit isoquant (K) and isocost line (L) for the inputs X_1 and X_2 for a given level of output Y (TE= technical efficiency; AE= allocative efficiency; EE= economic efficiency)

As indicated in figure 4.2, technical efficiency has been measured along a ray from the origin to the observed production point; these measures hold the relative proportions of inputs constant. One advantage of these radial efficiency measure is that it is units invariant; in other words changing the units of measurement does not change the value of the efficiency measure (Coelli et al., 1998).

Note that besides productivity and efficiency also income, profit and risk can be seen as important aspects to assess economic performance (Steunpunt Duurzame Landbouw, 2006, Dessers et al., 2006). Efficiency and productivity give probably the best indication of the economic success of a firm (or sector). But value added is needed to remunerate all production factors (labor, capital and land). This means that a high productivity does not guarantee that the shareholders can count on a large income. After the payment of the production inputs and the depreciation of investment goods, employees and the government (taxes) receive their share. Risk can be defined as the probability that occasions arise with a possible impact on the achievement of (company) objectives. The impact can be negative (e.g., a decrease of output prices) or positive (e.g., a decrease of the interest rate).

To summarize, the proper measurement and analysis of productivity and efficiency is critical in understanding the economic performance of sectors and agents. Hence, understanding productivity and efficiency and its key determinants is most important in assessing competitiveness and long-run economic performance of any sector (Veeman, 1995).

4.3 Measuring sustainability performance

4.3.1 Introduction

Sustainable development is a major policy concern for many. It seems unlikely that a competitive economy will achieve sustainability (Pezzey and Withagen, 1998). Intervention towards sustainability (e.g., on inter- and intragenerational grounds) is therefore necessary, but in order to monitor how well we are doing, indicators are needed (Hanley, 2000). In other words, to make sustainability a reality, we must measure where we are now and how far we need to go (Wackernagel et al., 1999). Moreover, indicators of sustainable development need to be developed to provide a solid basis for decision making at all levels (local, regional, national, and supranational levels) (Hueting and Reijnders, 2004). It is necessary to move from trying to define and describe sustainability towards developing concrete tools for promoting and measuring achievements (Veleva and Ellenbecker, 2001). Sustainable development is a complex idea that can neither be unequivocally described nor simply applied (Martens, 2006). Moreover, there exists no universal definition of sustainability that may be applied at all times and all places (Köhn et al., 1999, Patterson, 2006). Although we are faced with a large amount of uncertainty, only through testing our theories will we find ways to truly make progress towards more sustainability (Kane, 1999).

To close the gap between theory and practice, von Wirén-Lehr (2001) suggests a four step strategy: (i) goal definition, (ii) indicator/indicator set development, (iii) strategy evaluation, and (iv) management advice. Bosshard (2000) distinguishes nine different steps in an assessment procedure[4]: (i) vision development[5], (ii) development of criteria reflecting the vision, (iii) definition of goals for each criterion, (iv) determination of system delimitations, (v) development of indicator definitions, (vi) methodology development of measuring, (vii) defining the unit of measurement, (viii) assigning standards, (ix) value synthesis. The value synthesis includes the weighting of each criterion according to a given situation or project.

From concept to measurement means capturing the complexity of sustainability in time and space, and its tradeoffs among different components and aggregation levels, and this in a system context (Becker, 1997). As already explained, there are various interpretations of sustainable development, hence sustainable development is described in broad terms. Nevertheless, sustainability has to be defined in considerably narrower terms in order to establish operational rules of thumb to serve as the basis for the development of sustainability indicators (Lawn, 2006*c*).

Indicators of sustainability must be realistic in what they seek to accomplish, and what they can say about the paths we are on (van Kooten and Bulte, 2000). We also need to consider which trajectories are equitable, economically and ecologically desirable and achievable (Moffatt, 2000). Hence the measurement of sustainability is a daunting task. In fact, the search for reliable indicators has gone on and will go on for decades (Opschoor, 2000). Ideally an indicator is a means devised to reduce a large quantity of data to its simplest form, retaining essential meaning for the questions that are being asked of the data (Lenz et al., 2000). The value of a sustainability indicator is its potential to improve decision making, and so it is best thought of as a source of information (Pannell and Glenn, 2000). Hence, indicators describe (complex) phenomena in a quantitative way by simplifying them in such a way that communication is possible with specific target groups (Lenz et al., 2000). Indicators are a logical device to use in sustainability assessment (Bell and Morse, 1999). Nevertheless Bell and Morse (2003) see a number of key questions related to indicator development and application: (i) What indicators should one select? (ii) Who selects them? (iii) Why are they selected? (iv) What are they meant to help achieve? (v) What about the balance between the various

[4] In fact, these nine steps can be seen as a breakdown of step 1 (goal definition) and step 2 (indicator/indicator set development) of von Wirén-Lehr (2001)

[5] As explained in section 2.3.3

dimensions of sustainability? (vi) How are the indicators to be measured? (vii) How are the indicators to be interpreted, and by whom? (viii) How are the results to be communicated, to whom and for what purpose? and (ix) How are the indicators to be used? Indicators need to be relatively few in number, clear, concise and analytically robust (Hass et al., 2002, Patterson, 2006). A lot of quality demands for indicators exist, for example relevance, accuracy, comparability, coherence, accessibility and clarity. The calculation procedure should be objective and scientifically sound. Furthermore, indicators should relate to clear policy objectives. These criteria are applicable to all kind of indicators. A particular issue with respect to sustainability indicators is that they must be capable of integrating a large variety of factors (Hamilton et al., 2004). They also should cover the functioning of a system as a whole (Bossel, 1999, van den Bergh, 1999). On the other hand, in practice indicators are often a compromise between scientific accuracy and the information available at reasonable cost (Saisana et al., 2005). Sustainability indicators are still a simplified classic reductionist set of tools based on quantification (Bell and Morse, 1999). Furthermore, Shields et al. (2002) argue that indicators of sustainability will only be effective if they support social learning by providing users with information they need in a form they can understand and relate to.

Sustainability indicators serve as performance indicators in the sense of saying to us that things are getting better or that things are getting worse (Patterson, 2006). This implies that a reference point or benchmark system is necessary. To give guidance towards sustainability, reference values are needed for each indicator. Bell and Morse (2003) see two broad approaches: (i) a defined target or (ii) a defined direction. Reference points or benchmarks can include policy targets, best available technologies, comparisons with other countries or firms,. . . . Peet (2006) explains that indicators help us understand the world around us and control responses to it, and this is necessary for everyday life. Familiar examples are the temperature of one's skin or the level of one's bank balance. The more complex the system in which we live, and within which we need to control even simple decisions, the more we rely on indicators. As already explained, one has to avoid information overload. Therefore, Peet (2006) argues that we must pay the most attention to *red light* indicators, that indicate the need for urgent action. Patterson (2006) calls this *headline* indicators. These indicators try to reduce the complexity to a manageable and understandable level and to capture the communication power of a single number. Patterson (2006) emphasizes that when it comes to sustainable development, a headline indicator has to encapsulate the essential characteristics of social, economic and environmental progress. A headline indicator can be a composite index, often made up of a hierarchical structure of sub-indexes and variables. This is because one single variable is unlikely to be capable of capturing all sustainability dimensions. In the following sections we will discuss several potential headline sustainability indicators.

Often neglected, but van der Werf and Petit (2002) emphasize the need of validation. Indicators should be validated by (i) evaluating the appropriateness of its set of objectives relative to its purpose, and (ii) confronting indicator values and real-world data or submitting the design of the indicators to a panel of experts (van der Werf and Petit, 2002). Another way to validate an indicator to increase its reliability is to make the calculations behind an indicator transparant and subject to expert judgements and peer review (Reus et al., 2002). Bockstaller and Girardin (2003) present a methodological framework to evaluate indicators. They distinguish (i) a *design validation* to evaluate if the indicators are scientifically founded, (ii) an *output validation* to assess the soundness of the indicator outputs, and (iii) an *end use validation* to be sure the indicator is useful and used as decision aid tool.

An other often neglected issue in the development of sustainability indicators is the data analysis. Hardi and DeSouza-Hulety (2000) examine the relevant lessons of several empirical projects measuring sustainability. They recommend that data assessments should be done before final selection of indicators. Based on this assessment, recommendations can be made to fill data gaps. Furthermore, they suggest that data analysis should be based on statistical and econometric techniques.

Gahin et al. (2003) try to answer the question: *are indicators helping to make a difference in the community, or do they just become another report to gather dust on the shelf?* They conclude that indicators are a worthwhile effort and can yield many intangible benefits that provide a foundation for change. Indicators are important to capture our progress towards sustainability in a simplified and readily understandable way (Al Waer and Sibley, 2005). But Gahin et al. (2003) remind us that indicators are not a substitute for action. They can lead to progress, albeit sometimes slowly and incrementally, toward community sustainability and well-being.

There has been an explosion of activity to develop sustainable development indicators, in order to determine whether sustainable development was actually being achieved. Two major approaches can be distinguished as indicator system: (i) a set of indicators, and (ii) a single, composite index or a limited amount of aggregated indicators. In other words, one could keep the indicators entirely separate, but listed or presented together within a single table or diagram (*visual integration*), or one could combine the indicators to yield a single index of sustainability (*numerical integration*). The set of indicators or visual integration is often called the framework approach, while the numerical integration methods can be described as the aggregation approach (Ekins et al., 2003b).

4.3.2 Indicator sets: visual integration

Diaz-Balteiro and Romero (2004) call the indicator approach a pragmatic approach used by many researchers to characterize sustainability. Several sustainability indicator sets exist. Well-known examples are developed by the UN (United Nations, 2001), OECD (OECD, 2001*b*, 2006) and the EU (European Commission, 2005). These indicator sets have a common aim to draw up lists of indicators able to inform policy makers and public opinion about changes in historical paths of economic, social and environmental phenomena, without trying to summarize this complexity in a single composite measure of sustainability (Giovannini, 2004). Indicator sets to measure sustainability mostly translate the three-pillar approach in indicators of each pillar. In general, the emphasis of indicator sets lies in the multidimensional aspect of sustainability. The sustainability indicator sets are often placed in tabular formats and sometimes in more diagrammatic formats. Examples of the latter are the AMOEBA (Gilbert, 1996, Bell and Morse, 1999, Wefering et al., 2000, Giampietro and Pastore, 2001) and the Flemish adapted radar graph (Steunpunt Duurzame Landbouw, 2006, Meul et al., 2007c). In this way, the need to combine a broad set of indicators into a visual device suitable for decision makers are considered (Bell and Morse, 1999). Note that a difficulty in assessing the sustainability using the multidimensional approach is the fact that the units of measurement and the appropriate scales for measurement differ both within and across the dimensions of sustainability (Rigby and Cáceres, 2001).

Often a large number of indicator sets giving information on developments in the economic, social and environmental areas are selected. The problem is how many and which indicators to use? (Bell and Morse, 1999). The tendency to include a large number of sustainability indicators is the consequence of the need to cover the breadth of sustainability (Bell and Morse, 1999). Examples of atomistic lists of indicators are Azar et al. (1996) and Gustavson et al. (1999). Hass et al. (2002) provide an overview of sets of sustainable development indicators used by national and international institutes. For example the European Union developed a set of indicators based on the United Nation list of sustainability indicators. An overview of the measuring progress towards a more sustainable Europe can be found in European Commission (2005). Potential problems with the indicator approach are hidden non-linearities, interaction between indicators, dynamic aspects. Hence, most of these indicators have not been linked together in a common system. de Haan (2004) gets the impression of a fairly incoherent shopping list of numbers without underlying structure.

Also Farrell and Hart (1998) argue that in many cases, the sustainability indicators are simply combined lists of traditional economic, environmental, and social indicators with the word sustainable added to the title. Moreover, these lists are often long and thus impractical in use (López-Ridaura et al., 2002). Frameworks for measuring sustainable development should integrate the economic, environmental and social dimensions of sustainable development. Although the fact that unconnected indicators encourage the fragmented view, combining of several indicators can be seen as a significant first step (Farrell and Hart, 1998). The next important step is to analyze the links between social, environmental and economic aspects.

In theory, each indicator framework should provide an overview for considering sustainability problems and the associated interconnections between them (Lenz et al., 2000). von Wirén-Lehr (2001) distinguishes two types of frameworks: (i) system-based frameworks and (ii) content-based frameworks. The purpose of system analysis is the better understanding of a given system (Lenz et al., 2000). There is a need for a system approach (Hardakar, 1997) because it tries to show the interrelationships between all relevant aspects of a system. The system-based approach provides indicators describing key attributes of systems as a whole. An example of a system-based framework can be found in Bossel (2001). A content-based framework provides specific indicators that characterize single parts of the system of concern. Content-based frameworks facilitate the translations of functions into specific objectives and quantitative parameters, but the lack of a holistic approach in most frameworks does not allow for the evaluation of the system as a whole. An example of a holistic content-based framework is described by Van Cauwenbergh et al. (2007).

The organizations that are developing sustainability indicators range from the international to the very local, from corporations to national and municipal governments (Farrell and Hart, 1998). Several guides on how to develop sustainability indicators are developed because the process is considered as very important. A well-known example has been offered by the *Bellagio principles*[6], developed by an international group of researchers and practitioners. These principles deal with four aspects of assessing progress toward sustainable development. First, the assessment of progress toward sustainable development should be guided by a clear vision of sustainable development and goals that define that vision. Second, the indicators should reflect a holistic view of the linkages between the social, environmental and economic aspects, they should consider the essential elements (equity, economic development, ecological conditions on which life depends,...), and they should have the adequate scope while still having practical application. Third, the process of developing indicators should be open and inclusive with an effective communication and a broad participation. Fourth, the developers need to conduct ongoing assessment of the quality of the indicators (Hardi and Zdan, 1997, Farrell and Hart, 1998).

Furthermore, several indicator frameworks and indices are devel-

[6]The Bellagio principles are developed during a meeting held in November, 1996, at Bellagio, Italy

oped specific for certain activities, for example for agricultural activities[7]. Examples of such indicator frameworks for agriculture can be found in Smith and McDonald (1998), Pannell and Glenn (2000), Sands and Podmore (2000), Schultink (2000), von Wirén-Lehr (2001), Meul et al. (2007c) and Van Cauwenbergh et al. (2007). Note that for the sustainability evaluation of production systems, a variety of assessment tools exist, including *life cycle assessment*, *risk mapping*, *(environmental) impact assessment*, *multi-agent systems*, *cost-benefit analysis*, *linear programming* and *indicator lists* (Payraudeau and van der Werf, 2005, Van Cauwenbergh et al., 2007). Reviews of assessment methods of agricultural systems can be found in Payraudeau and van der Werf (2005) and in van der Werf et al. (2007).

4.3.3 Composite indicators: numerical integration

As mentioned, sets of sustainability indicators are often long including both qualitative and quantitative factors. Furthermore, these sets contain sometimes trade-offs among issues that cannot be resolved simultaneously (Cornelissen et al., 2001). Therefore, the indicator sets are often integrated in one single or a limited amount of composite indicators. Composite indicators can be defined as based on sub-indicators that have no common meaningful unit of measurement and there is no obvious way of weighting these sub-indicators. Developing an aggregated indicator implies selecting a number of different components and combining them into a single unit[8]. Aggregated sustainability indicators in a compact form are in particular useful to compare policy options (Farrell and Hart, 1998, Jollands et al., 2003), because they summarize complex or multi-dimensional issues and they provide the big picture (Saisana et al., 2005), without the danger of information overload (Jollands et al., 2003). The need to integrate sustainability indicators is directly related to the need of interpretation (Sauvenier et al., 2005). Furthermore, aggregated indices can help to convey simple messages and to reach new audiences, but also run the risk of being misinterpreted. The lack of transparency by highly aggregated indicators can be a serious problem (Bell and Morse, 2003). Therefore, it is essential that these indices satisfy several quality criteria and are interpreted in their proper context (OECD, 2001*a*). Jollands et al. (2004) conclude that aggregate indices do have a role in assisting decision makers, as long as they are not used in isolation from more detailed information. Costanza (2000) note that detailed information of aggregated indicators is not lost, usually it is possible to look at the details of how any aggregate indicator has been constructed, but decision makers are

[7] Agricultural sustainability is then seen as sustainability in reference to agricultural production systems (Cornelissen et al., 2001)

[8] A review of aggregation methods can be found in Saisana et al. (2005) and in OECD (2001*a*)

too busy to deal with these details. Sauvenier et al. (2005) argue that the integration of indicators is a net advantage, since indicators are a prerequisite to integration, the most detailed level of information stays always available. In practice, indicators and indices are the result of a compromise between scientific accuracy, concise informativeness and usefulness for strategic decision making (Lenz et al., 2000).

4.3.4 Assessing sustainability: a challenging task

Measuring sustainable development is necessary for addressing the long-term future of our societies (Kee and de Haan, 2003). Without integrated information on sustainability problems, public awareness of these issues will be limited and the formulation and monitoring of policy responses will be difficult (OECD, 2001*b*). There is little doubt that integrated approaches are required to support sustainable development (Martens, 2006). In making the concept of sustainable development concrete, one has to take into account a number of practical elements and obstacles. It is important to note that the measurement of sustainability is value-laden. Hence, it is important that researchers clearly specify on which paradigm (e.g., weak or strong sustainability) their sustainability measurement is based[9]. Many sustainability indicators comprise implicit valuations, weighting schemes and policy objectives, which are insufficiently recognized as such (van den Bergh, 1999). In the area of sustainability, a number of different indicators have been developed (Farrell and Hart, 1998) and these indicators depend on the underlying view of sustainability they embody.

Furthermore, sustainability assessment is inevitably based on strong simplifications both of the theoretical paradigm and the characterization of systems of concern (von Wirén-Lehr, 2001).

Also, no matter which approach is used, reliable, good quality data are needed to quantify indicators (Hass et al., 2002). The usefulness of a specific indicator or index is always limited by underlying data availability, quality and validity (Lenz et al., 2000).

Further, the measurement of sustainability is also subject to change. Activities we previously regarded as being sustainable may become regarded as unsustainable, because of better information, changing social values or increased uncertainty either in perception (e.g., precautionary principle) or fact (e.g., climate change) (Pearson, 2003).

There have been many efforts to develop indicators of sustainable development. As mentioned earlier there exist several concepts of sustainability and sev-

[9] Note that besides paradigm choice, it is also important that other measurement aspects are indicated for example scale, dimensions (economic, social, environmental), time, intergenerational and intragenerational issues.

eral approaches to develop sustainability indicators (Hezri and Dovers, 2006) at several levels (e.g., national and firm level). Examples of approaches are extended national accounts, biophysical accounts and indices, monetary indices, eco-efficiency and indicator sets (Hamilton, 2004).

The use of money as a common unit to integrate sustainability indicators seems logic, but it implies that all assets and processes involved can be valued, treating the environment as a commodity. On the other hand, bio-physical indicators are more difficult to integrate because of their different units of measurement (Bartelmus, 2000). In straightforward terms one can say that purely monetary assessments are useful for documenting core aspects of weak sustainability, while (bio)physical ones are useful for tracking strong sustainability (Ekins et al., 2003b, Hamilton et al., 2004). Used in tandem, these two measures help tracking the progress toward sustainable development, and highlighting critical issues and identifying policy responses (Hamilton et al., 2004).

Remark that besides the use of indicators, there exist also other approaches to assess sustainability. Growing recognition of the complex socio-political context of sustainability issues, has brought calls for more participatory and ideologically open approaches to sustainable development assessment (Meppem and Bourke, 1999). This requires a move away from technocratic decision-making and towards more dialogic approaches to decision-modeling and analysis (Bebbington et al., 2007). Brown (1998) and Walter (2002) underlines the social aspects by their plea in favor of a sustainability economics that will emphasize stewardship of ecological and human communities. A stewardship perspective implies that a community-oriented sustainability economics needs to be concerned with a broad range of issues. Individuals are seen as the product of their culture and as imbedded in a process of social learning and adaption, an thus as members of a human and an ecological community (Barrett, 1996, Walter, 2002). Within these transition based methods (as explained in section 2.3.3), the process of working with organizations and stakeholders to assess sustainability may prove more useful than the sustainability assessment itself. These methods will not be taken into account in our overview about measuring sustainability. Applications of participatory approaches to support sustainable development initiatives can be found in Nevens et al. (2007) and Bebbington et al. (2007).

Any meaningful analysis of sustainability needs to pay attention which concepts and assumptions are used in the measurement methodology. Therefore, our main focus in this review of sustainability measures will be on the underlying assumptions of the indicators. We make the distinction between measuring weak (section 4.3.5) and strong (section 4.3.6) sustainability on macroeconomic level. In a last section, we will discuss sustainability measures on firm level (microeconomic level) (section 4.3.7). We will not discuss all existing approaches but focus on the most important ones. Besides, we will not further discuss the numerous existing indicator sets (visual integration). Furthermore, because we will focus on measuring weak versus strong sustainability, we will pay too little attention to equity and intragenerational concerns. We will also shed insufficient light on the link between consumption and sustainability, we will more target on the production aspects (the *supply* side).

4.3.5 Measuring weak sustainability

As explained in section 3.2, weak sustainability can be seen as an extension of neoclassical economics. In a simplistic way, one can say that weak sustainability measures attempt to put a monetary value on the loss or impairment of environmental services, towards internalizing the externalities into the budgets of households, enterprizes and nations (Bartelmus, 2000).

Monetary asset accounts analyze to what extent the wealth of a nation is increasing or decreasing per citizen, thereby informing about progress on *weak sustainability*. In fact, these measures answer the question of whether countries are depleting their natural assets faster than they are building up produced assets (Hamilton et al., 2004).

We will describe several approaches to measure weak sustainability. Following Hanley (2000) we divide these measures into those based on flows and those based on stocks. Flow based measures (section 4.3.5.1) are essentially attempts to adjust Net National Product to transform it into an indicator of sustainability, for example the Index of Sustainable Economic Welfare (ISEW). Stock-based measures (section 4.3.5.2) revolve around the concept of natural and man-made capital stocks, for example the Genuine Savings (GS) approach.

Note that all weak sustainability measures (which are all related green accounting measures) are instantaneous measures, because they cannot conclusively tell us whether the economy is on a weak sustainability path (Asheim, 1994, Dietz and Neumayer, 2007).

4.3.5.1 Flow-based measures

Introduction Several different opinions exist about the possibilities to account for sustainability. Furthermore, there exist several, often different, defined notions such as sustainability accounting, green accounting, environmental accounting,... Green or environmental accounting describes the effort to incorporate environmental issues[10] into economic decision making. Sustainability accounting can be defined in a similar way: more specific the incorporation of environmental and social issues into the economic accounts. Schaltegger et al.

[10]These environmental aspects can be expressed in monetary terms (benefits and costs) or only in physical terms

(2006) see sustainability accounting as accounting for ecosystems and for communities, and consideration of eco-justice, as well as more conventional issues of effectiveness and efficiency. On the other hand, Cairns (2005) emphasizes that green accounting does not provide a method of accounting for sustainability and cannot be massaged, manipulated or extended to do so. Note that the distinction between measuring environmental and economic values and measuring sustainable development is not always very clear. A pragmatic restriction which stipulates that indicators should cover all the three pillars or components can be made as for example in Hass et al. (2002). But in fact this distinction makes no sense because sustainability can be seen as covering the three pillars but also as assuring inter- and intragenerational equity.

The research area of environmental accounting was initiated long before that of indicators of sustainability but is related to it (Proops et al., 1999). A pragmatic definition of sustainability accounting can be made: sustainability accounting is defined as accounting with a clear sustainability rule attached to it. Because sustainability is a very diverse concept, there exist a countless numbers of rules, indicating for example the triple-bottom line, inter- or intragenerational equity,... In general, El Serafy (2006) sees green accounting as a powerful economic tool, needed for understanding resource dependant economies and for guiding them along a sustainable development path. Lintott (2006) adds that not only does monetary environmental accounting require weak sustainability. Most weak sustainability methods apply monetary environmental accounting, if it is to be implemented.

Net National Product The Gross domestic product (GDP) indicator is very useful to measure economic performance but it is not a good measure of welfare. The core system of national accounts contributes to the understanding of the development of welfare, but it does not provide a complete picture (de Haan and Keuning, 1996). Hence, the GDP is a snapshot of today's economy and does not account for sustainability (Castaneda, 1999). Hueting (1991) formulates this as *society is sailing by the wrong compass, at the expense of the environment*. It does not attempt to correct for changes in environmental quality or the depletion of natural resources (Hrubovcak et al., 2000, Matthews and Lave, 2000).

The Net National Product (NNP) is calculated by subtracting depreciation from the Gross National Product (GNP) or in other words consumption plus the sum of investment minus depreciation for all asset types. A standard welfare interpretation of NNP is that it is the largest permanently maintainable value of consumption (Weitzman, 1976). The crucial result proved by Weitzman (1976) was that this conventional measure of income is precisely the level of consumption that, if obtained permanently, would have a present value equal to the economy's wealth. In other words, NNP can serve as an indicator of welfare. Note that NNP does not measure the maximum amount of constant feasible consumption which means that NNP does not equal the sustainable level of consumption (Pemberton and Ulph, 2001). Moreover, Asheim (1996*b*) showed that it would be impossible to develop the concept of NNP into an indicator of sustainability, because even in an economy without market imperfections, the competitive prices at a given time cannot provide information on whether consumption exceeds the sustainable level since, in an intertemporal competitive equilibrium, the relative prices of the multiple capital goods will depend on the entire equilibrium path. This is a general problem: without sustainability prices, we cannot know whether the economy is currently sustainable, but without knowing whether the economy is sustainable, currently observed prices tell us nothing definite about sustainability (Pezzey and Toman, 2002*a*). Neglecting this problem, one can say that under certain conditions, green NNP[11] exactly equals (weak) sustainability. Sustainability is then defined as the annualized equivalent of the present discounted value of consumption that the economy is capable of achieving (Weitzman, 1997). Note that a restrictive assumption is the absence of technological change, some crude calculations of Weitzman (1997) suggest that our best present estimates of sustainability may be largely driven by predictions of future technological progress. Reviewing these result Pezzey and Toman (2002*a*) state that although the theoretical link between national accounting measures and sustainability is problematic, Weitzman's provocative empirical results form an important challenge to those concerned about sustainability, one that cries out for further exploitation and discussion.

Cost-Benefit Analysis In practice, NNP[12] excludes many non-priced goods that contribute to welfare. Therefore, the aim of green accounting is to make NNP more comprehensive by including as many non-priced goods as possible (Cairns, 2005). This can be done through cost-benefit analysis.

Valuation and its application in cost-benefit based management have a long history (Farrow et al., 2000). The rule in cost-benefit analysis is: do action A if its benefits exceed its costs. In fact, the rule is: Do A if the benefits exceed those of the next best alternative course of action. The benefits of the next best alternative to A are indicated as the costs of A. The problem arise over the measurement of benefits and costs, because (i) for market items, market prices may not represent the true social value of the items and (ii) for non-market items including public goods and the external effects of market items, alternative valuation methods should be used (Layard and Glaister, 1994).

Cost-benefit analysis attempts to put all relevant costs and benefits on a common temporal footing. Estimating the monetary value of en-

[11] Green NNP is defined as the net aggregate that substracts from GNP not just depreciation of capital but also the value of depleted natural resources evaluated at competitive market prices (Weitzman, 1997)

[12] Remind: NNP = GDP minus deprecation of capital

vironmental damages is complicated, controversial, and generally uncertain (Matthews and Lave, 2000). Economists decompose the total economic value conferred by resources into three main components(Tietenberg, 2003): (i) use value, (ii) option value and (iii) nonuse value. Use value reflects the direct use of the environmental resource (e.g., fish harvested from the sea). The option value reflects the willingness to preserve an option to use the environment in the future even if one is not currently using it. Nonuse value reflects the common observation that people are more than willing to pay for improving or preserving resources that they will never use (Tietenberg, 2003). As explained in section 2.4.3 several techniques exist to value the benefits from environmental improvement or, conversely to value the damage done by environmental degradation (Tietenberg, 2003).

The market price represents the economy's best valuation of an additional unit of a good or service. Environmental valuation becomes most complex precisely when the standard framework fails to yield prices in well-functioning markets. The valuation of eco-systems poses daunting challenges, but it is an essential step for making management decisions (Arrow et al., 2000). It becomes in particular complex when dealing with problems of the commons (shared or unassigned resources). The shortcomings include uncompensated and indirect impacts on third parties[13] and a limited or very costly ability to process the available information. This failure in turn leads to problems in standard methods of evaluation for environmental goods and services (Farrow et al., 2000).

For goods and services not valued in the market, a number of approaches to help estimate their social values are developed. There are two major groups of methods to value environmental goods and services using (i) observed behavior or (ii) hypothetical behavior (table 4.1). An example of using observed behavior is the travel cost method. Travel cost models examine how much longer people are willing to travel to enjoy a higher quality of environmental amenity. In a similar way, environmental valuation can be made using observed behavior with lost wages, medical expenditures, jury awards for wrongful injury (Matthews and Lave, 2000). Two other observable methods are known as the hedonic property vale and hedonic wage approaches. Hedonic property value studies attempt to decompose the various attributes of value in property into their component parts (e.g., property values in polluted neighborhoods versus clean neighborhoods). Hedonic wage approaches attempt to isolate the component of wages which serves to compensate workers in risky occupations for taking on the risk (e.g., exposure to a toxic substance) (Tietenberg, 2003).

Hypothetical methods to measure environmental and resource values are the contingent valuation and ranking approach. Contingent valuation and similar methods ask respondents to state their preferences between hypothetical payments and increases or decreases in environmental quality or risk (Farrow et al.,

[13] called externalities

2000). In this way a surrogate market is constructed by asking people what they would be willing to pay (WTP) or willing to accept (WTA) for these improvements (Matthews and Lave, 2000). The major concern with the use of the contingent valuation method is the potential for survey respondents to give biased answers (Tietenberg, 2003). Furthermore, not everyone favors the contingent valuation approach. Sagoff (2000), for example, argues that one should rely on deliberative, democratic processes, including stakeholder negations in stead of using the WTP approach. This is because, in his view, WTP fails to correlate with any independently defined conception of value. In contingent ranking methods, respondents are asked to rank hypothetical situations in terms of their desirability.

Hence, economic analysis provides a number of decision tools that can be of great use to allocate resources and to make decisions on the environment. Azqueta and Delacámara (2006) reflect on some of the ethical constraints to the ability of conventional economic valuation techniques. They explain that the same environmental asset will be demanded as a resource at lower stages of development (both individual and socially) and as a part of the common heritage at a later stage. In the former, the use of conventional methods to value environmental goods and services will be warranted, whereas this would not be the case in the latter. Azqueta and Delacámara (2006) argue that the logic of valuation is not valid for historical, cultural or natural heritage because they are not commodities and they have superior values.

Table 4.1 gives an overview of the most common economic methods for measuring environmental and resource values. Other methods are choice modeling and production-function-based techniques. Choice modeling is a recent innovation in stated preference methods with the idea that a commodity is most usefully treated as the embodiment of a bundle of attributes or characteristics, which are the things of real interest to consumers (Lancaster, 1996, Perman et al., 2003). The production-function-based techniques values the environmental aspects as an input of a production function (Ellis and Fisher, 1987, Barbier, 2000). The state of the art on non market valuation can be found in Champ et al. (2003).

Table 4.1: Economic methods for measuring environmental values

Methods	Observed behavior	Hypothetical behavior
direct	market price	contingent valuation
	simulated markets	
indirect	travel cost	contingent ranking
	hedonic property values	
	hedonic wage values	
	avoidance expenditures	

Source: Tietenberg (2003)

Early applications Following Hicks (1946), the level of sustainable income is defined as that level of sustainable income which can be achieved without an overall decline in the value of total capital stock. But the exact nature of the required modifications of the gross domestic (or national) product is under dispute (Proops et al., 1999). An early estimate of green NNP[14] has been performed by Repetto (1989) where the depreciation of Indonesia's forest, petroleum reserves and soil assets was estimated. While this study called many to action, it also operated as a brake, leading many economists and statisticians to warn against a focus on green NNP, because it tells policy makers nothing about the causes or solutions for environmental problems (Hecht, 1999).

A widely cited article is published by Costanza et al. (1997). In this research the economic value of 17 ecosystems services is estimated. They found a value in the range of 16-54 trillion US dollars per year for the entire biosphere, while global gross national product is around 18 trillion US dollars per year. In other words, ecosystem services provide an important portion of the total contribution to human welfare. This implies that if ecosystems services were actually paid for, in terms of their value contribution to the global economy, the global price system would be very different from what it is today (Costanza et al., 1997). Reviewing the valuation methods used by Costanza et al. (1997), Pearce (1998) states that their article is deeply flawed. However, its intention to show how valuable the natural world is, was correct. On the other hand, Bockstael et al. (2000) argue that it makes little sense to talk about the economic value of ecosystems as if the choice were between having them as they are or not having them at all, because economic valuation is about trade-offs and as such it requires defining the alternatives clearly. Evaluating well-defined changes to ecosystems is the only reasonable question to ask and the most relevant question for most policy analysis (Bockstael et al., 2000). However, Costanza et al. (1997) argue that although valuation is certainly difficult and fraught with uncertainties, one choice we do not have is whether or not to do it. In other words, there is a need for ecological pricing of environmental assets to the overall economy (Hannon, 2001).

Sustainability accounting For the purpose of measuring sustainable development, the conventional system of national accounts is inadequate. For example it does not deal with the priceless environmental and social externalities (Kee and de Haan, 2003). To develop information systems linking different types of information within the system of national accounts (SNA), social accounting matrices (SAM) are developed (United Nations, 1993). These matrices are called hybrid accounts (United Nations, 2003) or NAMEA (National Accounting Matrix including Environmental Accounts) in many European countries (de Haan and Keuning, 1996). The system of environmen-

[14]Remind that green NNP is calculated as GNP - Dp - Dn where Dp is depreciation of produced capital and Dn is depreciation of natural capital

tal and economic accounting (SEEA) comprises four categories: (i) flow accounts for pollution, energy and materials, (ii) environmental protection and resource management expenditure accounts, (iii) natural resource asset accounts and (iv) valuation of non-market flow and environmentally adjusted aggregates (United Nations, 2003). The national accounting matrix including environmental accounts (NAMEA) shows environmental burdens that are consistent with the economic figures in the national accounts (de Haan and Keuning, 1996). The NAMEA tries to record the origin or destination of physical flows in connection to economic transactions as presented in the national accounts. In other words, the NAMEA extends the SNA with environmental accounts in physical units. In this way, a coherent linkage of monetary and physical data guarantees a consistent comparison of environmental burdens to economic benefits, or environmental benefits to economic costs (de Haan, 2004, de Haan and Kee, 2004). In most European union countries, NAMEA-type tables have been compiled (Keuning and Steenge, 1999). In addition, also a Japanese NAMEA has been developed (Ike, 1999).

The most important difference between the present version of the SEEA and the NAMEA, is that the NAMEA-system does not contain a Green National Income or Eco-Domestic product. Proponents of the NAMEA-system find just subtracting (hypothetical) environmental costs form actual national income yields an incoherent and meaningless figure, a correct Green National Income can only be the result of an explicit modeling exercise (Keuning and Steenge, 1999, de Haan and Keuning, 1996). Note that the NAMEA approach is in fact a hybrid approach and can therefore called a strong sustainability approach. Similar examples are the sustainability gaps (Ekins and Simon, 2001), sustainable national income (Gerlagh et al., 2002) and the GREENSTAMP. These approaches will be explained in section 4.3.6. On the other hand, Neumayer (2004*a*) finds that the main objective of NAMEA is rather environmental monitoring without a clear sustainability rule attached to it.

Another example of green accounting was implemented in the GARP projects (I & II) with the main focus on the impacts of air pollution. In a first step the level of emissions and their dispersion is calculated. Then the impact on economic activity is evaluated and converted into a cost figure. The two main elements are damage calculation and damage attribution. Damage attribution allows emissions to be allocated by economic sectors (Markandya et al., 2000a). The GARP methodology uses the NNP as a measure of an economy's welfare[15] and the correction of NNP for environmental effects (Markandya et al., 2000b). The GREENSENSE project tried to develop a framework based on the national accounting framework developed within the GARP projects and based on the sustainability approach developed within the GREENSTAMP project (see section 4.3.6). In this way, environmental impacts are analyzed under several sustainability targets (weak sustainability, intermediate sustainability, strong

[15] As already mentioned, this theory is due to Weitzman (1976)

sustainability). Empirical applications for policy purposes using this framework are hampered by the lack of data availability (Hunt et al., 2005).

Sustainability accounting in agriculture Furthermore, there are also several partial accounting studies, for example measuring the sustainability of the agricultural sector of a region or country. An early attempt of environmental costing for US agriculture has been done by Smith (1992). More recently, Tegtmeier and Duffy (2004) assessed the total external costs for the agricultural sector of the United States. Hrubovcak et al. (2000) incorporated environmental effects of agricultural production and the depletion of natural capital caused by agricultural production into existing income accounts. They found adjustments to income attributed to agriculture falling in the range of 6 to 8 percent. Using economic and environmental valuation techniques, Pretty et al. (2000) tried to assess the total external cost of UK agriculture. Hartridge and Pearce (2001) tried to estimate sustainability of the UK agricultural sector, assessing both the depreciation of natural capital (negative externalities) and appreciation of natural capital (positive externalities). They found that negative externalities amount to at least £ 1 billion, while positive externalities offsets approximately one half of these negative effects. More recently, Eftec (2004) calculated the environmental effects of UK agriculture. They find negative externalities of nearly £ 1.5 billion but also positive externalities of more than £ 1.2 billion. Reviewing several green accounting studies for UK agriculture, Buckwell (2005) finds the Eftec (2004) conceptually the most fully developed and empirically the most comprehensive[16]. Somewhat similar, Pretty et al. (2005) calculate the environmental costs of the UK food basket. They found that farm externalities, domestic road transport to retail outlets, domestic shopping transport and subsidies are the main contributors to the estimated hidden costs of £ 2.91 per person per week (11.8% more than the price paid).

Le Goffe (2000) used the hedonic price method to assess some external effects of agricultural activities, finding that intensive livestock farming caused the renting prices of rural cottages to decrease, while permanent grassland increased the price.

Pretty et al. (2001) present data on annual external costs in Germany, in the UK and in the USA. They see three categories of policy options available for encouraging changes in farmers' behavior and practices: (i) advisory and institutional measures, (ii) regulatory and legal measures and (iii) economic instruments. In practice, a mix of all three approaches, integration across sectors and full participation of all key stakeholders in the policy development and implementation process itself is needed (Pretty et al., 2001).

[16] This study is also the most recent study reviewed by Buckwell (2005)

Index of Sustainable Economic Welfare (ISEW) The *Index of Sustainable Economic Welfare (ISEW)* is also known as the *Genuine Progress Indicator (GPI)* or as the *Sustainable Net Benefit Index (SNBI)*. Because ISEW, GPI and SNBI basically differ in name only, we will use in this section only ISEW, also meaning GPI and SNBI. The guiding idea of ISEW is to correct the Gross Domestic Product (GDP) indicator with income inequalities, household labor and damage to natural capital. Computation of an ISEW usually starts from the value of personal consumption expenditures which is a sub-component of GNP. This consumption expenditures are then weighted with an index of distributional equity (e.g., GINI-coefficient). Finally, items regarded as contributing to either welfare or sustainability are added, and items that reduce either welfare or sustainability are subtracted. The recommendation is clear: ensure that the ISEW is not decreasing. In fact, Neumayer (2003) explains that the ISEW can be interpreted as an extended or greened Net National Product, which is defined as a comprehensive consumption minus genuine savings, where comprehensive consumption means that all relevant items are included in the consumption vector and not only consumption of material goods. This means that in comparison with the Genuine Savings (GS) (see section 4.3.5.2) more information about utility-relevant factors are included and thus more data is needed.

The first to propose and to develop an ISEW were Daly et al. (1989) and Cobb and Cobb (1994). Both groups of authors developed an ISEW for the USA. Following these two ISEW studies, an ISEW has been constructed for more than ten countries, for example Austria (Stockhammer et al., 1997), Chile (Castaneda, 1999), Thailand (Clarke and Islam, 2005, Clarke, 2006) and Australia (Hamilton, 1999, Lawn and Sanders, 1999). An application of the ISEW at local level (the Province of Sienna in Italy) has been worked out by Pulselli et al. (2006). Costanza et al. (2004) report on a multi-scale application at the city, county and state levels in Vermont, USA.

Besides the calculation of the ISEW in different countries, the ISEW can also be used in studies in stead of the traditional GNP. For example Talberth and Bohara (2006) found the use of ISEW in econometric modeling useful. In their study the link between economic openness and the ISEW is studied.

In almost all ISEW studies we see a similar trend: starting from around the 1970s or early 1980s, the ISEW no longer rises very much or even falls, whereas GNP continues to rise (Neumayer, 2000). Although based on a totally different methodology, Max-Neef (1995) found the results of the ISEW studies a fine illustration of the *Threshold Hypothesis*. The threshold hypothesis states that for every society there seems to be a period in which economic growth brings about an improvement in the quality of life, but only up to a point, the threshold point, beyond which, if there is more economic growth, quality of life may begin to deteriorate (Max-Neef, 1995). Following methodological improvements for the valuation of resource depletion and long-term environmental damage, Neumayer (2000) state that the threshold hypotheses fails to materialize.

The methodology has several serious flaws (England, 1995). Firstly, a drawback is the fact that the methodology employed differs from country to country depending on data availability and preferences of the authors (Neumayer, 2004*a*). Secondly, the ISEW is not derived from a theoretical model[17] which implies that the specific adjustments undertaken and their justification are often somewhat ad hoc (Neumayer, 2004*a*). This means that the growing gap between GNP and ISEW, found in the empirical applications, could be an artifact of the ISEW methodology (England, 1995, Neumayer, 2000). Neumayer (2000) and Dietz and Neumayer (2006*c*) make several recommendations about the methods used to measure the depletion of nonrenewable resources, about the long-term environmental damage and about adjustments for inequality. Lawn (2003) is surprised about the little effort that has been devoted to the establishment of a theoretical foundation to support ISEW. He argues that contrary to the opinion of Neumayer (2004*a*) the ISEW is soundly based on a concept of income and capital first advanced by Fisher (1906). Remind that Hicks (1946) pointed out that the purpose of calculating income is to indicate the maximum amount people can produce and consume without undermining their capacity to produce and consume the same amount in the future. In contrast, Fisher (1906) stated that the national dividend consists not of the goods produced in a particular year, but of the services enjoyed by the ultimate consumers of all human-made goods. Lawn (2003) finds the Fisherian view of income superior because it takes the view that economic welfare depends on the psychic enjoyment of life, instead of the rate of production and consumption. Mates (2004) shows that both Hicksian and Fisherian income can play important roles in quantifying the requirements for a sustainable macro-economy. In contrast, Lawn (2004) find that Fisherian income, unlike Hicksian income, can provide benchmarks to determine the appropriate rate of growth during a nation's development process and, from a policy-guiding perspective, when to make the transition from one investment strategy to another. Lawn (2006*a*) emphasizes that the most urgently needed refinement concerns the establishment of a consistent set of valuation methods. To counter the problem of too many different approaches, resulting in a inconsistency problem, he suggest to move towards standardization of items and valuation methods. A last remark on the ISEW method is the fact that it is not clear what specific policies are to be undertaken if the ISEW is falling.

To summarize, the ISEW methodology has successfully synthesized many of the shortcomings of traditional income accounting within a single welfare orientated framework. Many estimates are still based on preliminary estimates and upon highly speculative assumptions but Herman Daly emphasize that *ISEW is like*

[17] In contrast with the Genuine Savings which is derived from the Hartwick rule (see section 4.3.5.2)

putting a filter on a cigarette. It's better than nothing. Hence, ISEW can be seen as a springboard for future research on national accounting and not as a complete framework filled with accurate data (England, 1995).

4.3.5.2 Stock-based measures

The most famous rule to measure weak sustainability is the Hartwick rule, which requires that total net capital investment, or in other words the rate of change of total net capital wealth, will not be allowed to be persistently negative (Hamilton, 1994). Total net capital investment includes gross investment in all forms of capital that can be feasibly measured, minus depreciation or capital consumption (Dietz and Neumayer, 2007).

Hamilton (1994) introduced the term genuine to distinguish genuine savings, which refers to changes in all capital forms including natural capital, human capital and social capital, from traditional net savings, which only refers to produced capital. The following assumptions are made (Neumayer, 2004*a*): (i) population is constant, (ii) the social welfare function is a discounted utilitarian function with a constant rate of discount, (iii) the dynamic welfare is maximized such that the competitive economy develops along the intertemporally efficient path with all externalities optimally internalized, (iv) the productivity of the economy is full captured by all capital stocks, indicating stationary technology (Asheim et al., 2003) and (v) other forms of capital can substitute for the depletion of natural capital without limit or natural capital is superabundant or technical progress can always overcome any apparent resource constraint (Neumayer, 2003) (weak sustainability assumptions). With these assumptions, the genuine savings can be used as a one-sided test of sustainability. As explained in section 3.2, the Hartwick rule (and thus genuine savings) can be called a one-sided test, this because it shows only unsustainability, not sustainability (Pezzey, 2004). Hence, one can formulate the following policy recommendation: invest into all forms of capital at least as much as there is depreciation of all forms of capital (Neumayer, 2004*a*).

The first major empirical paper was by Pearce and Atkinson (1993). They tested the sustainability of 18 nations. All European countries in the sample were determined as sustainable economies while all the African countries were determined unsustainable. Besides this early crude calculations of the genuine savings, the World Bank regularly publishes estimates of genuine savings (also called *adjusted net income*) since 1999 (Hamilton, 2005). Genuine savings are calculated as

Genuine savings	=	+	investment in produced capital
		-	net foreign borrowing
		+	net official transfers
		-	depreciation of produced capital
		-	net depreciation of natural capital
		+	current education expenditures

In this way, human capital formation is estimated using current education expenditures. This is rather crude, but estimating investment for so many countries over a long time horizon in a more complete way is impossible given the data constraints (Neumayer, 2004*a*). Furthermore, social capital is not included at all due to measurement difficulties.

A very critical and discussed point is the measurement of the depreciation of natural capital. Theoretically, natural capital depreciation is equal to total rent as defined by Hotelling (1931): [(resource price (P) - marginal cost (MC)) * resource extraction (R)]. Data on marginal cost are frequently unavailable (Neumayer, 2003, 2004*a*). Because data on average cost are better available, the World Bank replaces the marginal cost with the available average cost to calculate the depreciation of natural capital[18]. Two popular alternative methods to calculate the deprecation are known as the El Serafy method[19] (El Serafy, 1981, 1991, 2002) and the Repetto method[20] (Repetto, 1989). Neumayer (2003) found on theoretical grounds more good reasons in favour of the World Bank's method comparing with the Repetto method. On the other hand, the El Serafy method can be a better approximation to depreciation of natural capital than the World Bank method (Neumayer, 2003).

Furthermore, the assumption of intertemporal efficiency is hard to defend. In reality markets fail, especially markets for natural assets, if they even exists (Neumayer, 2004*a*, Dietz and Neumayer, 2006*a*). As already mentioned, we come to the insolvable problem: without sustainability prices, it is impossible to know whether an economy is sustainable, but without knowing the sustainability of an economy, observed prices cannot be used with certainty as sustainability prices.

Another drawback is the assumption of stationary technology. The genuine savings model is vulnerable to shocks from outside the system such as exogenous technological progress, terms-of-trade effects and a non-constant discount rate (Dietz and Neumayer, 2006*a*). This assumption breaks down if a shock is partly exogenous in the sense that is not fully captured by total capital (Weitzman, 1997, Neumayer, 2004*a*). After exogenous shocks prices are no longer optimal and do not reflect economic scarcities (Neumayer, 2003).

[18]Hence, the rule becomes (P - AC)*R with AC = average cost

[19]$(P - AC) * R * [\frac{1}{(1+r)^{n+1}}]$ with r = discount rate; n = number of remaining life-time years of the resource stock

[20](P - AC)*(R-D) with D = resource discoveries

Looking forward from the base year into the future, there is no guarantee that genuine savings are giving the correct sustainability signals. In fact, after each shock, prices should be re-estimated which is impracticable (Dietz and Neumayer, 2006*a*). A pragmatic (and partial) solution is given by Pezzey et al. (2006)[21]. In their calculation of the GS for Scotland for the period 1992-1999, they include exogenous technical progress and changing terms of trade in oil using projections up to 2020.

The genuine savings rule becomes more complex if the assumption of constant population is abandoned (Arrow et al., 2003). It is not surprising that many developing countries with strong population growth appear to be even less weakly sustainable once population growth is taken into account (Neumayer, 2004*a*).

Remark that an economy may extracting large amounts of natural resources, which may indicate unsustainability by the genuine savings rule. However, the responsibility for this resource extraction may be in another country if the natural assets are exported, either directly or embodied in manufactured goods. Pillarisetti (2005) finds the methodology of the construction of the genuine savings flawed because it ignores these global negative externalities created by the advanced countries. Therefore, Proops et al. (1999) developed a weak sustainability criterion for an open economy as an important extension of the weak sustainability rule for a closed economy as in Pearce and Atkinson (1993). Assuming that countries can import and export sustainability, Proops et al. (1999) found significant differences between the open and closed economy measures for several regions. The closed economy measure understates the sustainability of the Middle East, while it overstates the sustainability of Western Europe and the USA.

Another conceptual problem is the fact that what affects current well-being need not affect sustainability at all or not in the same way, and vice versa. That is why Neumayer (2001) and Neumayer (2004*b*) suggest to combine the HDI (Human Development Index) with a sustainability rule (the genuine saving rule).

To summarize, if one is concerned with weak sustainability, then genuine savings results can be interesting. The genuine saving rule is a meaningful counterweight to gross product in the measurement of social welfare and as an indicator with a direct (one-sided) sustainability criterion. On the other hand, the genuine saving method is a very rough measure of sustainability with several assumptions (Dietz and Neumayer, 2006*a*). Finally, it is not entirely clear what specific policies should be undertaken following the detection of negative genuine saving rates (Neumayer, 2004*a*). In fact, the only useful suggestion

[21] Pezzey et al. (2006) construct not only a augmented GS measure but also an augmented Green NNP measure

is that countries with negative genuine savings should invest more of the proceeds of natural capital into the formation of other forms of capital than they currently do (Dietz and Neumayer, 2006*a*).

4.3.5.3 About measuring weak sustainability

It should be clear that the quest for an alternative to GDP is far from over (England, 1995). In fact, no currently-available single measure of sustainability is likely to be adequate and satisfactory on its own (Hanley, 2000). Conceptual and empirical problems still exist with both stock-based and flow-based measures. Pezzey et al. (2006) compared the change in green NNP and the interest on GS for Scotland during 1992-1999. They found that the former greatly exceeds the latter, indicating a mismatch which poses an unresolved problem with the theory.

GS estimates have to be treated with great care and with much caution in interpretation. First, GS cannot be an indicator of sustainability, but only a negative indicator of unsustainability. In other words, negative GS rates indicate unsustainability while a positive GS rate does not indicate sustainability. Second, methodological improvements are possible for example using the *El Serafy method* for resource accounting. Third, the quality of the data is often very poor (Neumayer, 2003). Fourth, GS studies give no policy support towards sustainability because they give no answers on the question how to reach sustainability.

Besides the GS measure, the ISEW is the most known indicator of weak sustainability. Unfortunately, it shares many of the (methodological) problems. Moreover, it suffers serious shortcomings specific to the methodology and it depend on a number of problematic assumptions (Neumayer, 2003). In contrast with the GS, based on the Hartwick rule, there is no theoretical framework to underpin the ISEW, although Lawn (2003) disagrees. On the other hand, the ISEW is one of the exceptions that do measure the problem of equitable distribution. The aspect of equity is often ignored in sustainability assessment.

Finally, weak sustainability measures (e.g., genuine savings) are based on a model of an inter-temporally efficient economy. In reality this assumption will not hold because markets fail, especially because markets for environmental assets often do not exist. Therefore, knowing that the economy is intertemporally inefficient might suggest a preference for those indicators of strong sustainability that set exogenously defined standards for environmental assets (Dietz and Neumayer, 2006*a*). Beyond the valid fundamental points of strong sustainability indicators, doubts remain with respect to the validity and usefulness of these indicators (Neumayer, 2003), as we will explained in section 4.3.6. Before describing several measurement methods for strong sustainability, it can be useful to quote El Serafy (2002):

> Weak sustainability is better that no sustainability at all, and the national accounting framework is a good medium for attaining it. Weak sustainability could be viewed as a first step that must be taken along the road leading to a stronger sustainability.

4.3.6 Measuring strong sustainability

Biophysical natural capital accounts measure sustainability by evaluating to what extent humanity's demand on the biosphere, in terms of renewable and nonrenewable resource consumption and waste production, exceeds nature's capacity to renew itself. Such biophysical accounts provide a measure of strong sustainability (Hamilton et al., 2004).

Strong sustainability is a more diffuse paradigm than weak sustainability (Neumayer, 2003), this implies that many rules have been suggested that seek to operationalize it (Dietz and Neumayer, 2007). Van der Hamsvoort (2006) emphasizes that the theoretical concept is hard to put in practice because the setting of correct sustainability constraints is hampered by substantial uncertainties and lack of knowledge. Furthermore, it appears unlikely that the human society will be prepared to pay the bill for reverting to a path of sustainable development.

Neumayer (2003) identifies two different main schools of thought. One requires that the value of natural capital be preserved. This means that for example in the case of nonrenewable resources, extraction must be compensated by an investment in renewable resources such as wind energy (Dietz and Neumayer, 2007). This conception of strong sustainability assumes unlimited substitutability between forms of natural capital and no substitution between natural resources and other resources (e.g. economic resources). An example are the *ecological footprints*. The second strand of strong sustainability requires a subset of total natural capital be preserved in physical terms so that its functions remain intact. This is called critical natural capital (Dietz and Neumayer, 2007), which can be defined as that part of the natural environment that performs important and irreplaceable functions (Ekins et al., 2003a). In other words, only some parts of the natural capital stocks are critical: only those in which replacement is impossible or unlikely (Van der Hamsvoort, 2006). The determination of the degree of criticality of natural capital is difficult. De Groot et al. (2003) argue that criticality is basically determined by two main aspects: importance and the degree of threat, whereby importance consists of a large number of criteria, including ecological and socio-economic aspects. Presenting four case studies, De Groot et al. (2006) conclude that determining the criticality of natural capital is not an easy task. Criticality is the result of many factors affecting both the threats and the importance of natural capital. Furthermore criticality is also based on different value perspectives all of which interact in different ways (De Groot et al., 2006). The following strong sustainability measures will be discussed: (i) ecological footprints (section 4.3.6.1), (ii) material flows (section 4.3.6.2) and (iii) hybrid indicators (sections 4.3.6.3).

4.3.6.1 Ecological footprints

The *ecological footprint* is an accounting tool that can aggregate ecological consumption measuring the relevant physical stocks and flows. Wackernagel and Rees (1997) state that using money values to measure natural capital constancy, is misleading from an ecological perspective because a constant monetary value of a resource can result from the physical depletion of the stock. Furthermore, prices say nothing at all about non market, but ecologically essential, stocks, processes and ecosystem functions. The ecological footprint is a method that represents critical natural capital requirements of a defined economy or population in terms of the corresponding productive areas (Wackernagel and Rees, 1997). The ecological footprint or the appropriated carrying capacity can be seen as the aggregate area of land and water in several ecological categories to produce all the resources they consume, and to absorb all their wastes they generate. Hence, the ecological footprint depends on the population size, material living standards, technology and ecological productivity (Wackernagel and Rees, 1997). The ecological footprint can be calculated for the whole world, countries, regions, cities, persons or activities. In a first step, consumption is determined in a particular spatial domain for all relevant categories (e.g., food, transport). In a second step, the land area appropriated by each consumption category is estimated for several land categories (e.g., crop land, forest, built-up area). Finally, the area figures are summed to give an estimate of the ecological footprint of that particular spatial domain. A detailed overview of the methodology and the calculation of the ecological footprint can be found in Wackernagel et al. (1999).

The ecological footprint can be seen as a useful yardstick for sustainability, because the difference between the size of a region (adjusted by its ecological productivity) and the footprints of this region's population must be covered by imports of ecological surpluses or the depletion of natural capital stocks. This difference explains why the current human economy lives in part on natural depletion rather than on sustainable flows (Wackernagel and Rees, 1997).

Ecological limits can be exceeded for a period of time because nature reacts with inertia. In this way, natural capital can be harvested faster than it regenerates, thereby depleting the capital stock, this is called *overshoot*. The purpose of ecological footprints is to illustrate the possibility of *over-*

shoot[22] (Wackernagel and Silverstein, 2000). Wackernagel et al. (2006) state that from the perspective of resource management, overshoot may be the most central sustainability concern. Furthermore, they argue that the good news is that it can be measured, the bad news is that it is no longer merely a possibility: in many regions and even for the planet as a whole, we are already in ecological overshoot.

Ecological footprints can be used to build bridges between people who may otherwise not agree (Wackernagel and Silverstein, 2000). The ecological footprint has been praised as an effective heuristic and pedagogic device for presenting current total human resource use in a way that communicates easily to almost everyone (Costanza, 2000, Rees, 2000).

Wackernagel and Rees (1997) find that estimates of the ecological footprint and appropriated carrying capacity provide clear direction for action. It can be seen as a tool for crafting sustainability strategies (Wackernagel and Silverstein, 2000). Wackernagel et al. (1999) suggest three complementary strategies to reduce footprints while not compromising our quality of life: (i) increasing nature's productivity per unit of land, (ii) doing the same with less through the better use of the harvested resources, and (iii) consume less by being fewer people and consuming less per capita. Ayres (2000) believes that the ecological footprint is too aggregated to be an adequate guide for policy purposes at the national level. In his view, it is just another way of saying things we already knew. van Kooten and Bulte (2000) share this opinion and they observe that the ecological footprint is not about measurement, but about raising public awareness and pursuit of a political agenda. In other words, the ecological footprint does not provide policy recommendations and contains no policy prescription and is more an *attention grabbing device* (Ayres, 2000, van Kooten and Bulte, 2000, Moffatt, 2000, Neumayer, 2003). On the other hand, others disagree and find the ecological footprint useful in charting progress toward, or away from, sustainability (Templet, 2000, Rees, 2000). Wackernagel and Yount (2000) state that by making the method more complete (they suggest nine methodological improvements) this tool could evolve from being largely of pedagogical use to become a strategic tool for policy analysis.

A major advantage of the ecological footprint is the fact that all human exploitation of resources and environment is reduced to a single dimension (i.e. land)(van den Bergh and Verbruggen, 1999). *Land* can be seen as a more familiar, acceptable and motivating concept to most people, than energy, CO_2, or biodiversity (Herendeen, 2000). The obvious and substantial benefit of an aggregate indicator is its production of a single number, which makes using it for decision-making relatively straightforward (Costanza, 2000, Moffatt, 2000).

[22] Wackernagel and Silverstein (2000) illustrate this with the following quote: *the daredevil jumping from the 50th floor declares while passing the 15th floor that he is perfectly safe since nobody has gotten hurt yet.*

Herendeen (2000) finds that the ecological footprint is a vivid indicator of dependence on imports and exports and an excellent tool to illustrate the larger picture.

However, a single aggregate indicator does not allow for trade-offs among the several dimensions of sustainability (van den Bergh and Verbruggen, 1999, van Kooten and Bulte, 2000). Opschoor (2000) finds composite indicators not transparant because they do not always disclose the underlying dynamics. Moreover, modeling rather than an accounting approach should be followed to realize economically feasible outcomes. Furthermore, a disadvantage is the use of physical-land conversion factors, because these factors are used as implicit weights in the conversion as well as in the aggregation (van den Bergh and Verbruggen, 1999). This weight may correspond to ecological principles and thermodynamics laws (Wackernagel and Silverstein, 2000) but they do not correspond to current social weights or technological potential (van den Bergh and Verbruggen, 1999, Ayres, 2000).

The ecological footprint methods builds on the critical importance of natural capital to economic development (Rees, 2000). However, the ecological footprint depends on assumptions about the substitution between various forms of nature. This means that despite the strong sustainability stance of its proponents, the ecological footprints require implicit judgements about the substitutability within natural capital forms and between natural capital and other forms of capital (van Kooten and Bulte, 2000). In other words, the ecological footprints method does not constrain substitutability within natural capital. Therefore, Neumayer (2003) argues that it is doubtful whether ecological footprints really represent an indicator of strong sustainability because if you define strong sustainability as maintaining critical functions of natural capital, then you have to constrain substitutability within natural capital forms. Dietz and Neumayer (2007) believe it is inappropriate to assume natural capital cannot, on the one hand, be substituted by produced capital but can, on the other hand, be substituted by another form of natural capital.

Further, van den Bergh and Verbruggen (1999) finds that the ecological footprint is dominated by energy use: Wackernagel and Rees (1997) and Wackernagel et al. (1999) assumes only the forestation strategy to reduce CO_2 build-up in the atmosphere, meaning that other strategies to absorb CO_2 and other ways to generate useful energy without producing CO_2 are ignored (Ayres, 2000). Furthermore, the ecological footprints focus mainly on productive land, and thus it omits any role for the oceans which cover most of the Earth's surface (Ayres, 2000, Moffatt, 2000).

Another objection against the ecological footprint is the anti-trade bias. van den Bergh (1999) suggests that trade can in principle spatially distribute the environmental burden amongst the least sensitive natural systems. However, it is physically impossible for every country to be a net importer of biocapacity (Wackernagel and Silverstein, 2000). In fact, Wackernagel and Silverstein (2000) emphasize that the issue is not trade as such, but its composition, an ecological-friendly trade strategy will result in smaller footprints.

Another point is the use of current technologies to calculate the ecological footprint. In fact, the role of technological change is ignored (Moffatt, 2000). Therefore, Costanza (2000) explains that the ecological footprint must be seen in terms of a technologically skeptical indicator, one that assumes that technology will not save us.

van den Bergh and Verbruggen (1999) conclude that the ecological footprint is not the comprehensive and transparent method as is often assumed.

4.3.6.2 Material flows

The principle concept underlying the material flows approach is a simple model of the interrelation between the economy and the environment, in which the economy is an embedded subsystem of the environment and dependent on a constant throughput of materials and energy (Giljum, 2006). The concept of material flows is inspired by the work of Ayres and Kneese (1969). Natural resource use must be made more efficient and the economic growth decoupled from physical growth. Industrial societies dependence on natural resources must be reduced and our economies *dematerialized* to some degree. Proponents stress the need for physical accounts, analogous to national economic accounts, this because our knowledge of resource use and waste outputs is limited (Matthews et al., 2000). These physical accounts should show the origin, use and deposition of all materials associated with economies. From the perspective of material flows, the focus needs to shift from the *sink side* of the economy to the *source side*. The material flows can help policy makers to understand and deal with the origins of specific environmental problems (Wernick and Irwin, 2005). Proponents of the material flow approach believe that to understand the links between economic activity and environmental degradation and to integrate economic and environmental planning and policy-making a more detailed understanding of the material basis is required (Adriaanse et al., 1997). Moreover, Hinterberger et al. (1997) argue that the concept of material flows uses material input as a proxy for environmental impact and therefore for sustainable economic behavior.

The basic idea of dematerialization is decoupling environmental impact from economic growth. In contrast, the Environmental-Kuznets-Curve (EKC) suggests that such a decoupling would be an automatic feature of growth, hence no further action is required (Bartelmus and Vesper, 2000). The Environmental

Kuznets-Curve[23] analyzes the effects of economic growth and development process on environmental quality, suggesting the existence of an inverted U-curve (a proxy of development is placed on the horizontal axis and a proxy of environmental deterioration is place on the vertical axis). Borghesi and Vercelli (2003) conclude that the causal relationship between globalization and global environmental degradation is quite complex and ambiguous. Moreover, EKC-analysis has significant deficiencies (Tisdell, 2001). An overview of the theoretical and econometrical problems is given by Müller-Fürstenberger and Wagner (2007). To summarize, the idea of dematerialization cannot be rejected by the findings of the EKC empirical studies.

The concept of material input counts the material input of the economic systems in kilograms or tons (Hinterberger et al., 1997). The emphasis on scale rather than on efficiency[24] partly explains why weight is used as the unit of accounting rather than money (Neumayer, 2003). A material flow analysis describes the flow of one or more specific materials in a geographic area during a certain period of time. The inflow of materials into the domestic economic system originates from both the extraction of new materials and the imports. The throughput of materials through the economic system is made up of all flows of material between the different economic processes. The outflow of materials from the domestic economic system consists of exports and the disposal of materials (Dellink and Kandelaars, 2000). The total physical requirements of a national economy are called the total material requirement (TMR)[25]. This is the sum of the total material input and the hidden or indirect flows. The hidden material flow is the portion that never enters the economy. To analyze the decoupling of the economic activity from natural resource use, the ratio of TMR to GDP can be used (Adriaanse et al., 1997). This indicator quantifies the eco-efficiency of an economic system by calculating economic output generated per material input. Although eco-efficiency can be used on national and on firm level we will discuss eco-efficiency as measure for sustainability performance in section 4.3.7.

Indicators of material flows are created by summing the weights of many different materials. This means that the material flow indicators are implicitly weighted since they do not differentiate between quality of materials (Hinterberger et al., 1997). The most important criticism of the concept of

[23]By contrast, the effects of natural resources endowment on development are mainly analyzed trough the so called Resource Curse Hypothesis (Constantini and Monni, 2006). Atkinson and Hamilton (2003) explore the link between the resource curse hypothesis and sustainability, finding that the countries where growth has lagged are those where the combination of natural resource, macroeconomic and public expenditure policies have led to a low rate of genuine savings.

[24]following Daly (1991*b*)

[25]Giljum (2006) distinguish the following economy-wide material flow-based indicators: (i) direct material input, (ii) total material requirement, (iii) domestic processed output, (iv) domestic material consumption, (v) total material consumption and (vi) physical trade balance.

material flows is that it adds up apples and oranges. Different forms of material throughput with different environmental impacts cannot be meaningfully added together just because one can express both in weight terms (Neumayer, 2003, Giljum, 2006). Therefore, the weighting by weight has been criticized as *tonne ideology*. Matthews et al. (2000) stress that summing up different materials is an attempt to create value neutral physical accounts that include all materials regardless of their economic importance or environmental impact. On the other hand, they admit that physical accounts do not in themselves provide information on environmental impacts. But they provide the means if the accounts can be weighted appropriately. There are several possibilities; for example if weighted by price, the financial attributes of material flows of national accounts could be calculated, or weighted by relative toxicity results in a possible assessment of the relative toxicity of different sectoral activities. Hence, a crucial step in refining physical accounting systems is the development of such weighting systems to show that specific material cycles can be linked to specific environmental impacts (Matthews et al., 2000). Furthermore, indicators of total material flows through the economy need to be supplemented by indicators for individual materials or materials classes (Wernick and Irwin, 2005).

An important aim is the reduction of flows by a factor 4. Factor 4 means that resource productivity can and should grow fourfold, the amount of wealth extracted from one unit of natural resources can quadruple. Thus we can live twice as well, using half as much (von Weizsäcker et al., 1997). Moreover, Hinterberger et al. (1997) explains that many scientist even agree that an increase in the resource productivity of Western economies by a factor of 10 is necessary. This claim should be understood as an average goal to achieve within the next 40-50 years. In this sense, dematerialization by a factor 10 servers as a management rule for sustainability (Hinterberger et al., 1997). This call for general reductions in material flow is not guaranteed to be ecologically effective, but is guaranteed to be highly economically inefficient with respect to whatever reduction in environmental damage might be achieved, this because the material flow analysis fails to appreciate the importance of valuing benefits and opportunity costs (Neumayer, 2004*a*). However, Neumayer (2003), and Dietz and Neumayer (2007) see much more potential in the concept of material flows once the idea of total flow reductions is abandoned and accounting is limited to flows with sufficiently similar environmental impacts, this because environmental policy is indeed too much focused on the sink-side of the economy.

Examples of empirical applications of the material flow analysis can be found in Adriaanse et al. (1997), Matthews et al. (2000), Chen and Qiao (2001), Scasny et al. (2003), Luengo and Chico (2004), Giljum (2004) and Wernick and Irwin (2005).

4.3.6.3 Hybrid indicators

Following Neumayer (2003), hybrid approaches can be defined as the combination of physical indicators with monetary valuation. Essential is that no monetary values are put upon items of natural capital, but the monetary costs of achieving the standards are estimated. This approach is based on the work of Roefie Hueting. Applications of his ideas can be found in several approaches, for example the *sustainability gaps*, the *GREENSTAMP* and the *sustainable national income*.

As described in section 4.3.5.1 traditional economic accounts use GDP as *the* indicator for economic success, but environmental losses are neglected. Therefore, a correction of national income for environmental losses seems highly recommendable (Hueting, 1991). During the last decades, several approaches to measure the monetary value of the environmental aspects are suggested.

As mentioned earlier, there are several methodological problems[26], which makes that a theoretically sound correction of national income is mostly not possible (Hueting, 1991). All hybrid approaches follow the conviction of Hueting that it is practically impossible to value environmental functions with the help of shadow prices (Neumayer, 2003).

Because there is a need for a practical indicator, Hueting (1991) proposes a workable second-best alternative. His starting point is the suggestion that human impact has reached a level that threatens the integrity of environmental functions (Hueting, 1980). Hueting and Reijnders (2004) argue in favor of using as sustainability measure a production level that does not threaten the living conditions of future generations. The concept of environmental functions is used to define the environment in a manageable way with the link between the environment and the economy. An environmental function is a possible use of the environment. When the use of an environmental functions conflicts with the use of another (or the same environmental function), loss of function occurs[27]. Hueting (1991) proposes the following procedure: (i) define a physical standard for environmental functions, (ii) based on their sustainable use, formulate the measures necessary to meet these standards and (iii) estimate the amounts of money involved in putting the measures into practice.

Sustainability gaps The sustainability gaps approach is developed by Paul Ekins and Sandrine Simon. The sustainability gap is defined as the difference between the current level of environmental impact from a particular

[26] Without sustainability prices, it is impossible to know whether an economy is sustainable, but without knowing the sustainability of an economy, observed prices cannot be used with certainty as sustainability prices.

[27] For example the loss of topsoil resulting from deforestation

source, and the sustainable level of impact according to the sustainability standard (Ekins and Simon, 1999). Besides sustainability targets, there are also policy targets. A policy target may be equal to the sustainability target but if the sustainability standard is considered too expensive or demanding politically, then a less demanding target can be set or the timescale for achieving the sustainability standard can be lengthened. Ekins and Simon (2001) find this a preferable way rather than considerations leading to the adjustment of the sustainability standards themselves, because the trade-off between achieving environmental sustainability and other political objectives is then apparent. The sustainability gaps can be combined with current trends to show how long it would take, on continuation of the trends, for the sustainability standard to be attained. This indicator is called *years to sustainability* (Ekins and Simon, 2001).

In Ekins and Simon (2001) sustainability gaps are computed for the UK for CO_2, SO_2, and other air pollutants; and years to sustainability are calculated for the Netherlands.

Furthermore, Ekins and Simon (2001) remark that physical sustainability gaps indicators give no idea of the economic implications of sustainability gap or of attempts to reduce it. Therefore, they suggest the use of marginal cost of abatement, avoidance or environmental restoration to construct monetary sustainability gaps. But they stress that such an overall monetary sustainability gap does not represent the amount of money that would have to be spent to achieve sustainability. It rather represents at one moment in time the aggregation of expenditures that would be needed to reduce all the dimensions of the physical sustainability gap to zero (Ekins and Simon, 2001).

GREENSTAMP The GREENSTAMP methodology, following Hueting (1991), is based on a requirement for strong sustainability, rejecting the monetization of the benefits of environmental goods in order to apply cost benefit analysis to identify the most efficient allocation of resources (Markandya et al., 2000b). Instead, the GREENSTAMP tries to quantify economic opportunity costs associated with meeting specified environmental performance standards, using multi-sector national economic models (Brouwer et al., 1999). The difference with the original Hueting (1991) approach is the fact that costs of achieving the environmental standards are deducted from actual national income, resulting in a sustainable income (Neumayer, 2003). This methodology is applied for the energy sector of the French economy by O'Connor and Ryan (1999) using a dynamic simulation model.

Sustainable national income In the view of Hueting (1991), Gerlagh et al. (2002) tried to calculate a sustainable national income indicator. The goal of publishing the SNI is to put into perspective the significance of gross national product in political opinion forming and policy making (de Boer and Hueting, 2004). In a first step, the sustainable resource use is defined and compared with the actual resource use. In a second step, changes (direct and indirect) in income caused by the required changes in resource use are calculated using an applied general equilibrium model. Because the magnitude of changes in allocation are too substantial for a statistic approach, a dynamic model is used to create a hypothetical sustainable economy with a hypothetical income. Gerlagh et al. (2002) see the gap between *Net National Income (NNI)* and the *sustainable national income (SNI)* as an important measure indicating the dependence of the economy on that part of its natural resource use that exceeds the sustainable exploitation levels. In terms of costs involved to meet the sustainability standards, Gerlagh et al. (2002) found that climate change is in the Netherlands the most pressing environmental issue. Further, they find that for the Netherlands, about half of its income could be attributed to resource use that exceeds a sustainable level.

About hybrid indicators Because no measures can be formulated for irreversible losses, it is very difficult to assign a value to these losses. The valuation of the depreciation of nonrenewable resources can be done by estimating the costs involved in the development and practical introduction of alternatives like solar energy, substitutes for minerals and recycling methods (Hueting, 1991). Drawbacks of the hybrid indicators are that (i) the results do not represent individual valuations, (ii) the approach is strictly static and that (iii) the approach does not indicate the state of the environment and the method is laborious (Hueting, 1991). Both GREENSTAMP and SNI tried to make the methodology less static using dynamic modeling. In this way, implausible partial equilibrium assumptions are avoided, but the hypothetical character of the estimated feasible economic output as the result of a modeling exercise represents also the greatest weakness of these indicators. This because the results and the modeling exercise are difficult to understand and several model assumptions are needed (Neumayer, 2003).

On the other hand, Hueting (1991) sees the following advantages (i) the method is the only way to confront the national income figures with the losses of environmental functions in monetary terms and (ii) the method compels us to explicitly define sustainability, and thus operationalize sustainability which makes policy measures possible.

4.3.6.4 About measuring strong sustainability

Strong sustainability is a more diffuse paradigm than weak sustainability, resulting in many approaches to operationalize it. Nevertheless, two main schools of thought can be distinguished. The first requires that the total value of natural capital should be preserved but assumes unlimited substitutability between forms of natural capital. A well-known example is the ecological footprint. The ecological footprint is an accounting tool that can aggregate ecological consumption, measuring the relevant physical stocks and flows in terms of the corresponding productive areas. The advantage of the ecological footprint is that it is an effective heuristic and pedagogic device for presenting current human resource use in a way that communicates easily to almost everyone. However, it is inappropriate to assume natural capital cannot, on the one hand, be substituted by produced capital but can, on the other hand, be substituted by another form of natural capital. Another observation is that the ecological footprint is dominated by energy use. Hence, the ecological footprint is a good attention grabbing device, but it is not that comprehensive and transparent method as is often assumed. Nevertheless, several already suggested improvements can change the ecological footprint from being mainly a pedagogical tool to become a strategic tool for policy analysis.

A second strand of strong sustainability requires a subset of total natural capital to be preserved, this is called the critical natural capital. Examples are the material flows and the sustainability gaps. The material flows are physical accounts that show the origin, use and deposition of all materials associated with economics. The advantage of the material flows is the focus on the source-side of the economy instead on the sink-side. However, different forms of material throughput with different environmental impacts cannot be meaningfully added together just because one can express both in weight terms. One can see much more potential in the concept of material flows once the idea of total flow reductions is abandoned and accounting is limited to flows with sufficiently similar environmental impact. Hybrid approaches can be defined as the combination of physical with monetary valuation. An example of an hybrid approach is the sustainability gap, which is defined as the difference between the current level of environmental impact from a particular source and the sustainable level of impact according to the sustainability standard. Several drawbacks of the hybrid indicators can be formulated for example that the results do not represent individual valuations, but it is an interesting way to confront national income figures with the losses of environmental functions in monetary terms.

Our overview of methods to assess strong sustainability is limited because there exists so many interesting methods to assess strong sustainability with both advantages and disadvantages. Furthermore, these methods operationalize sustainability in very different ways, and they all have two things in common: they are recently developed and several improvements are advisable and desirable. Nevertheless, given the increasing amount of papers, we belief that we are on the right track to assess strong sustainability.

4.3.7 Measuring firm sustainability

Schaltegger et al. (2006) emphasize that given the broad goal of sustainable development in general, corporate (or firm) sustainability is a challenging concept which needs to be put into practice. Sustainability is a global concept and a firm is only a small subsystem that interacts in various ways with surrounding systems. Nevertheless, companies are essential actors in socio-economic life and as such they contribute to the realization of sustainable development (Tyteca, 1998). Corporations are the organizations with the resources, the technology, the global reach, and ultimately, the motivation to achieve sustainability (Hart, 1997). Corporate sustainability is necessary for long-term sustainable development of the economy and society (Schaltegger et al., 2006). Defining and measuring corporate sustainability is more than just an academic concern. Corporate entities are increasingly under pressure to demonstrate how they contribute to the national sustainability goals outlined by governments (Atkinson, 2000). Hence, there is no doubt that non-market issues (both environmental and social issues) can have a substantial impact on the competitiveness and economic performance of a company (Schaltegger and Wagner, 2006). Dealing with business relevant non-market issues does not mean complying with every demand made by stakeholders. Rather, it entails identifying the issues that can and should be realized in line with a company's specific business activities (Forstmoser, 2006, Schaltegger and Wagner, 2006). Sustainability measurement at firm level could supply useful information for management and can be suitable to support the policy decision making process as well.

Remark that the line of measuring sustainability on national and firm level is not always clear. Some authors argue that it is important to use a framework considering several scales. For example, Smith and McDonald (1998) emphasize that sustainability assessment methodologies require multi-dimensional levels and multi-scales (field, farm, watershed, regional and national scale).

Green NNP may appear to be an abstract, aggregated measure that is not relevant to a firm's decisions and it does not express the great diversity of decisions made in an entire economy. But it would be wrong to condemn it for being the summary that it is intended to be (Cairns, 2005). Besides, only firm level studies can take into account within firm differences in environmental, social and economic aspects.

Not only measuring national sustainability but also firm sustainability requires a narrower operational definition to assess sustainability than the general broad description of sustainable development. Many, and very different approaches exist to assess firm sustainability. The following approaches will be briefly discussed: (i) reporting and accounting (section 4.3.7.1), (ii) eco-efficiency (section 4.3.7.2), (iii) indicator approaches (section 4.3.7.3), (iv) multi-criteria analysis (section 4.3.7.4), (v) efficiency analysis (section 4.3.7.5), (vi) life cycle analysis (section 4.3.7.6) and (vii) firm level modeling (section 4.3.7.7).

4.3.7.1 Reporting and accounting firm sustainability

Schaltegger et al. (2006) defines sustainability accounting as the description of new information management and accounting methods that aim to create and provide high quality information to support a corporation in its movement towards sustainability. Sustainability reporting is then defined as the description of new formalized means of communication which proved information about corporate sustainability. The concept of sustainable development at firm level has been more and more applied in recent years. Well-known examples are The Global Reporting Initiative (GRI, 2002) and ISO 14031 (ISO, 1999). Atkinson (2000) introduces the notion of corporate sustainability, meaning that external costs or damages associated with the generation of a corporate's income, should be translated into the environmental or full-cost account of that company. Hill (2001) measures farm sustainability using integrated environmental and economic accounting approaches. He applied his indicators on five Scottish farms, finding differences in the relative sustainability of different farm types.

Pacini et al. (2003) developed a holistic, integrated economic-environmental accounting framework to evaluate farm sustainability of three farming systems, this at farm level, site level and field level. They found that organic farming systems environmentally perform better than integrated and conventional farming systems, which does not mean that they are sustainable when compared to the intrinsic carrying capacity and resilience of a given ecosystem. An overview of several applications of sustainability accounting and reporting can be found in Schaltegger et al. (2006).

4.3.7.2 Eco-efficiency

In the view of the desirability of improving the efficiency of transformation processes, a narrow definition of sustainability emerges, namely: *sustainable development is an example of increasing eco-efficiency* (Lawn, 2006*c*). Gabriel and Braune (2005) argue in favor of eco-efficiency by explaining that the steps recommended by microeconomic eco-efficiency analyses might be too small to save the world, but the potential deficit resulting from such a decision is clearly smaller than that from a decision ignoring eco-efficiency considerations. Hubbes and Ishikawa (2007) argue that eco-efficiency should be appropriate for sustainability, although the analysis is simplified by disregarding non-linearities and dynamics.

The eco-efficiency measure is a broadly accepted criterion for corporate sustainability (e.g., Schmidheiny (1992), OECD (1998), WBCSD

(2000)). Eco-efficiency, standing for a better management of the economy with less environmental pressure, presents a promising sustainability approach (Bleischwitz and Hennicke, 2004). There is a wide and diverse variety of terminology referring to eco-efficiency (Hubbes and Ishikawa, 2007). Eco-efficiency indicators have been defined as values, parameters, measures, pieces of information, signs, targets and tools (Jollands, 2006*b*, Heijungs, 2007). In fact, eco-efficiency can be seen as a *management approach* for companies and businesses to contribute to sustainable development as acknowledged at the 1992 Rio Earth Summit (United Nations, 1992, Meul et al., 2007b). On the other hand, measurement of this management approach is typically expressed as the eco-efficiency equation. A well-known definition of eco-efficiency is the ratio of created value per unit of environmental impact. In fact, this variant of eco-efficiency can be seen as environmental productivity (Huppes and Ishikawa, 2005), and is similar to the definition of productivity in economics (see section 4.2.2). As explained in table 4.2, Huppes and Ishikawa (2005) distinguish four basic types of eco-efficiency.

Table 4.2: Four basic types of eco-efficiency

	product or production primary	environmental improvement primary
economy divided by environment	production value per unit of environmental impact *environmental productivity*	cost per unit of environmental improvement *environmental improvement cost*
environment divided by economy	environmental impact per unit of production value *environmental intensity*	environmental improvement per unit of cost *environmental cost-effectiveness*

Source: based on Huppes and Ishikawa (2005)

Figge and Hahn (2004*a*) state that eco-efficiency has three major shortcomings to measure corporate contributions to sustainability. First, eco-efficiency is a relative measure giving no information on effectiveness[28]. Second, advances in environmental performance due to improved eco-efficiency can be overcompensated because better eco-efficiency may lead to growth and thus increased use of environmental resources. This is called the rebound effect (Mayumi et al., 1998, Herring and Roy, 2002). Environmental resources which are saved due to improved eco-efficiency might be employed by other companies which are less eco-efficient. Therefore, Kuosmanen (2005) argues that accounting for economic and environmental rebound effects is an important task for improving the eco-efficiency analysis. Third, eco-efficiency does not take into account all social and environmental impacts simultaneously. Moreover, eco-efficiency does not cover the social dimension of sustainability at all (Brattebo, 2005).

DeSimone and Popoff (1997) emphasize that eco-efficiency is already creating great benefit to business and society but there is more to be done if its

[28] This is only true if you define eco-efficiency as a productivity or intensity ratio and not as an improvement ratio (see table 4.2)

full potential is to be unleashed. Therefore better ways of measuring eco-efficiency needs to be developed. Several researchers propose adapted methods of eco-efficiency to improve the sustainability measurement. Kuosmanen (2005) proposes to use efficiency frontiers to resolve the incommensurability problem to aggregate several dimensions. Efficiency frontiers are based on technical possibilities instead of value weights, as suggested by Tyteca (1998) (see section 4.3.7.5). Kuosmanen and Kortelainen (2005) examine how data envelopment analysis (an efficiency frontier method) can be used to improve the measurement of eco-efficiency.

Figge and Hahn (2004*a*) and Figge and Hahn (2005) introduced the concept of sustainable value, a new approach to measure corporate contributions to sustainability, based on the assessment of the value of capital beyond economic capital. They explore the relation between value and capital, which is clearly relevant in the context of intergenerational sustainability analysis (Hubbes and Ishikawa, 2007). They developed a valuation methodology to calculate the cost of sustainable capital and the sustainable value creation of companies. Other methods to improve eco-efficiency analysis are for example the sustainability balanced scorecard (Figge et al., 2002, Möller and Schaltegger, 2005, Wagner and Schaltegger, 2006) and the maximum abatement cost method (Oka et al., 2005, 2007).

Remark that aggregate measures of eco-efficiency are also needed to complement existing measures and to help highlight important patterns in eco-efficiency data (Jollands et al., 2004). Hubbes and Ishikawa (2007) even argue that the ultimate aim of eco-efficiency analysis is to help move micro-level decision making into macro-level optimality. Note that the unit of decision making cannot just be individual activities, as their interrelations have to be taken into account (Hubbes and Ishikawa, 2007). Hence, linking the micro level directly to the macro level seems an inappropriate solution. Jollands et al. (2004) developed aggregate measures of eco-efficiency for the use in national environmental policy. They use the principal components analysis as aggregation method to reveal trends in eco-efficiency of New-Zealand. Another example of the use of eco-efficiency on a regional level can be found in Seppälä et al. (2005) for the Finnish Kymenlaakso region and in Lawn (2006*b*) for Australia. Furthermore, eco-efficiency analysis can also be applied on a sectoral level, for example for the steel and aluminium industries (Dahlström and Ekins, 2005) or for agriculture (Meul et al., 2007b). Applications of eco-efficiency analysis at a firm level can be found in DeSimone and Popoff (1997), in Bleischwitz and Hennicke (2004) or in Seiler-Hausmann et al. (2004).

4.3.7.3 Indicator approaches

In section 4.3.2, a short overview was given of indicator approaches at macro-level. These indicator approaches can be applied in a similar way at firm level (micro level). van der Werf and Petit (2002) and Halberg et al. (2005) review respectively 12-indicator based approaches with the focus on assessing the environmental impact on agriculture and 55 input-output accounting systems[29] to facilitate voluntary improvements in farm environmental performance. They both found a great diversity of analysis. An example is the AGRO*ECO method, a conventional environmental impact assessment methodology to evaluate potential impacts of arable farming systems on the environment (Girardin et al., 2000). They developed an evaluation matrix with environmental components and farming management variables. This matrix can be used for the development of agro-ecological indicators (as in Bockstaller et al. (1997)) and for the use of multi-criteria methods for sorting, selecting or classifying farming systems according to their effects on the environment (Girardin et al., 2000). Rigby et al. (2001) developed a farm level indicator of agricultural sustainability, based on patterns of input use, for a sample of 80 organic and 157 conventional producers in the UK. This indicator is derived from earlier work by Taylor et al. (1993) and Gomez et al. (1996). Although the focus lies on the environmental issues, their analysis serves to highlight some of the conceptual issues of measuring farm sustainability such as weighting, presentation and validation. Kirner and Kratochvil (2006) find that scientific literature on agricultural sustainability indicators is characterized by a focus on the environmental dimension of sustainability. Therefore, they select indicators on a pragmatic basis (e.g., data availability) of each dimension, hence also considering economic aspects (e.g., farm income) and social aspects (e.g., animal welfare). Kirner and Kratochvil (2006) conclude that boosting efficiency and increasing output reflects a trade-off with environmental and social services provided by the agricultural sector[30]. A more comprehensive selection of indicators of sustainable production is suggested by Veleva and Ellenbecker (2001). They suggest twenty-two core indicators on all dimensions for raising companies' awareness and measuring their progress toward sustainable production systems. Other examples of sound sustainability indicator approaches on firm level are Häni et al. (2003) and Meul et al. (2007c). Both place several farm indicators in an instrument with a clear visualization.

Remark that indicator systems are composed of series of individual indicators that are not connected[31] and thus independently defined and measured.

[29]Input-output based indicators typically use a set of indicators to express the degree of environmental impact on the use of external inputs in relation to the production (Halberg et al., 2005)

[30]Kirner and Kratochvil (2006) do not claim that they make a comprehensive evaluation of sustainability in dairy farming, but their study is useful as sustainability reflection of one agricultural sector

[31]There can be trade-offs and relationships between indicators but these interlinkings are not recognized by the indicator approach

4.3.7.4 Multi-criteria analysis

Andreoli and Tellarini (2000) compare two types of methodologies to assess farm sustainability. Their first methodology transforms initial information into utility values and processes these values by using sophisticated techniques such as multi-criteria analysis. Due to the multidimensional aspect of sustainability, multi-criteria analysis is an obvious and natural choice. Nevertheless, assessing farm sustainability using multi-criteria analysis is not easy and several problems (e.g., processing qualitative data) can occur (Andreoli et al., 1999, Andreoli and Tellarini, 2000). If time, information and financial resources are not lacking the multi-criteria method is to be preferred (Andreoli and Tellarini, 2000). The second methodology simply transforms initial data on the base of quartiles by summing them up without any weighting. The advantage of the latter method is the easy implementation and this methodology can be understood by decision-makers and administrators without a background in statistics (Andreoli and Tellarini, 2000).

Hence, multi-criteria analysis allows for the conversion of a multidimensional scale to a unidimensional scale. A specific multi-criteria technique is the analytic hierarchy process (Ruf and Muralidhar, 1998). Krajnc and Glavic (2005*a*) designed a model for obtaining a composite sustainable development index in order to track integrated information on economic, environmental and social performance with time. In other words, they developed an aggregate measure which can be used to compare and rank companies regarding sustainable development. Using the concept of analytic hierarchy process, the impact of individual indicators on the overall sustainability of a company can be assessed. The analytic hierarchy process is a multi-attribute decision model used to derive weights of indicators by the prioritization of their impact on overall sustainability assessment of the company. In Krajnc and Glavic (2005*b*), the effectiveness of the proposed model is illustrated with a case study. Possible drawbacks of this model are the selection of indicators and the way in which the weights of indicators are determined (Krajnc and Glavic, 2005*a*,*b*).

4.3.7.5 Efficiency analysis

As explained in section 2.3.2, economic, social and environmental efficiency can be seen as a necessary - but not sufficient - step towards sustainability (Callens and Tyteca, 1999, Templet, 2001). Sustainability is enhanced by strategies which promote resource use efficiency in economic systems (Templet, 1999). In fact, efficiency forms the keystone of policy, planning and business approaches to sustainable development but there is a wide range of potential interpretations of the efficiency concept (Jollands, 2006*a*,*b*, Heijungs, 2007). First defined in a scientific context, the term efficiency now shows up in many everyday conversations, often based on a non-quantitative meaning (Hubbes and Ishikawa, 2007). Jollands and Patterson (2004) showed that efficiency is a core focus within economics, thermodynamics and ecology with as consequence that the term represents a multiplicity of meanings (Jollands, 2006*a*). Remind that all efficiency concepts are relative and context-dependent (Stein, 2001). In section 4.2, we defined efficiency using production economic theory. But also to measure sustainability, efficiency concepts are used such as eco-efficiency (section 4.3.7.2).

Tyteca (1998) showed that the principles of productive efficiency can be used to elaborate sustainability indicators at the firm level. Callens and Tyteca (1999) worked out indicators based on both the concepts of cost-benefit analysis and the principles of productive efficiency. Reinhard et al. (1999) estimate the technical and environmental efficiency of a panel of Dutch dairy farms. The environmental efficiency is defined as the ratio of minimum feasible to observed use of multiple environmentally detrimental inputs, conditional on observed levels of output and the conventional inputs (Reinhard et al., 2000). Reinhard et al. (2000) found respectable levels of technical efficiency and somewhat smaller environmental efficiencies and they showed that Dutch dairy farms utilize energy relatively more efficient than nitrogen. In contrast, Coelli et al. (2007) argue that methods involving the inclusion of a pollution variable as an input variable or (bad) output variable into a production technology are inconsistent with the materials balance condition. They developed a new method of measuring the environmental efficiency of firms that involves the incorporation of the materials balance concept into the production model.

De Koeijer et al. (2002) also present a conceptual framework for measuring farm sustainability on the basis of efficiency theory. They quantified sustainable efficiency and technical efficiency for a sample of Dutch sugar beet growers and they found a positive correlation between sustainable and technical efficiency. Moreover, differences in efficiency among farmers were persistent within and between years.

4.3.7.6 Life cycle assessment

One of the methodologies to measure firm sustainability, is the *life cycle assessment (LCA) methodology*. Brentrup et al. (2001) shows that the LCA methodology is basically suitable to assess the environmental impact associated with agricultural production. However, application of LCA on practical firms requires in-depth research to understand underlying processes, and to predict or measure variation in emissions realized in practice (de Boer, 2003). Several applications using the LCA methodology exist, e.g., of pesticides (Margni et al., 2002), to crop production (Brentrup, Küsters, Kuhlmann and Lammel, 2004), to N fertilizer use in winter wheat production systems (Brentrup, Küsters, Barraclough and Kuhlmann, 2004). Note that the main focus of these applications lies on environmental aspects. However, there are also attempts to integrate social and economic impacts into LCA (Weidema, 2006, Hunkeler, 2006). Remark that the life cycle assessment approach can also be used on national level. An example of the assessment of the US food system using the LCA methodology can be found in Heller and Keoleian (2003).

4.3.7.7 Firm level modeling

Another approach is used for example by van Calker et al. (2004). They use farm level modeling to determine how farm management adjustments and environmental policy affect various sustainability indicators. van Calker et al. (2006) used the combination of the multi-attribute utility theory with goal programming to develop an overall sustainability function for Dutch dairy farming systems. Besides data at the attribute level, stakeholders and experts are used for the assessment of subjective and objective attributes respectively. The sustainability function showed to be a suitable method to rank and compare different dairy farming systems (van Calker et al., 2006). Pacini et al. (2004) developed a holistically designed ecological-economic model to evaluate farm and field-level environmental-economic trade-offs. Furthermore they tried to evaluate the impact of the Agenda 2000 reform on the sustainability of organic farming, finding soil erosion being the only real environmental threat. Another example of modeling to assess sustainability is explored by ten Berge et al. (2000). They use multi-objective modeling to integrate knowledge at crop and animal level, to estimate the consequences of particular choices (e.g., choice of farming system) on scientific grounds.

4.3.7.8 About measuring firm sustainability

Indicators for corporate sustainability performance can embrace the dimensions of sustainability in a more or less integrative manner (Schaltegger and Wagner, 2006). The challenge of measuring firm sustainability turns out to be the same in several methods (e.g., eco-efficiency, LCA): (i) how to set system boundaries? (ii) how to determine the temporal and spatial scale? and (iii) how to define the functional unit? (Brattebo, 2005).

Testing several methods to integrate environmental and economic aspects (e.g., LCA, eco-efficiency and multi-criteria analysis), Park et al. (2006) recommended to use several methods simultaneously because all methods have their own pros and cons. Following their advice is attractive, but not always practical and feasible due time and budget constraints. It seems therefore advisable to use the most appropriate method depending on the objective and on the possibilities. In other words, the choice of measure may vary substantially depending on the exact question, industry and company considered and on what factors are part of the respective analysis (Schaltegger and Wagner, 2006).

Analyzing corporate sustainability performance, Schaltegger and Wagner (2006) conclude that there exists no automatic link between environmental, social and economic performance, but a link between sustainability performance, competitiveness and economic success is possible if sustainability is managed considering the basic links and the value drivers of shareholder value.

4.4 Lessons learned: measuring (sustainability) performance

In the past the performance of countries, regions and companies was defined in traditional economic terms: value added, productivity, efficiency, profit,... In fact, the more return on capital, the better the country or firm performs. Recently, the view on performance has been broadened. To create value, countries or companies do not only need economic capital but also environmental and social resources. In fact, all relevant resources should be considered when assessing performance. In this broad view, high performance is similar to high sustainability. That is why the formulation *sustainability performance* should be viewed as a pleonasm.

Measuring sustainability means that all relevant dimensions (economic, social and ecological) should be measured correctly, but also the long-term problems of absolute limits in resource use and resulting limits to growth and the international aspects of the globalized economy and its dynamics (e.g., intragenerational equity). Therefore, it seems unlikely that there exists one single measure of sustainable development which is capable of capturing all that is meant by *sustainability* (Hanley et al., 1999). The large amount of measurement systems show that indeed no single measurement system or management framework for sustainability exist (Köhn et al., 2001). No tool for measuring sustainability is complete and none will satisfy everyone (Rees, 2000). On the other hand, following Neumayer (2003), this does not mean that science measuring sustainability cannot help in our aim to reach sustainability. Science can tell a society that is committed to sustainable development many things about what appears to be necessary to fulfil the sustainability constraint. Besides, several notions of sustainability exist, indicating different views on different aspects such as substitution possibilities and believe in technical progress.

Hanley et al. (1999) apply several indicators of sustainable development for Scotland over the period 1980-1993 for: (i) green NNP, (ii) genuine savings, (iii) ISEW, (iv) ecological footprint, and (v) environmental space. Their evidence favors the opinion that Scotland is unsustainable. The different indicators give different messages about the sustainability of Scotland, but this is not surprising given the multi-faceted nature of the concept of *sustainable development* (Hanley et al., 1999). On the other hand, the diversity of results and the practical problems (definition and data) indicate that there is still work to do in developing reliable and widely accepted measures of sustainability (Pezzey and Toman, 2002*a*).

As explained in this section, there exist several approaches to assess sustainability. In fact, it is not the question which is the best approach: a physical or a monetary approach. The answer is to use both physical ánd monetary approaches. Furthermore, it is important to know the (sometimes hidden) assumptions and possibilities of the sustainability indicators. Further, improvements of existing measurement frameworks are still useful and sometimes even necessary. In addition, empirical work is needed because indicators are only useful if they are applied in practice.

More important than the correct measure of sustainability, is to ensure that the direction of change towards what is considered necessary for sustainability is known, rather than the attainment of some particular number (Ekins, 2003c). Regardless of what indicators are devised, inherent deficiencies always exist diminishing the policy guiding value of the applied indicator. Most methods are focused on measuring sustainability and give not enough policy support towards sustainability. A major research challenge is to answer the question *how to reach sustainability*. Despite this, Lawn (2006*c*) states that most indicators have the potential to provide policy-makers with important information about past and present activities. The quest for appropriate sustainable development indicators is critically important, and the need to refine and improve existing indicators remains acute (Lawn, 2006*d*). Hence, the advocates of the various sustainable development indicators must never rest on their laurels (Lawn, 2006*c*).

Part II

Empirical analysis

Introduction (part II)

The scarcest resource is not oil, metals, clean air, capital, labour, or technology. It is our willingness to listen to each other and learn from each other and to seek the truth rather than seek to be right
—Donella Meadows

The previous part of the dissertation gave a conceptual and theoretical framework. Understanding the different existing notions of sustainability is essential before making sustainability operational using empirical applications. Mark that the theoretical framework is very broad considering several notions and methods on different levels, while the empirical applications have a clear focus. More specific, using an efficiency approach, the economic and sustainability performance is measured and analyzed at farm level.

In this part, empirical work about measuring farm performance and farm sustainability will be presented. This because, as shown in the previous part, there is a need for empirical analysis of (sustainability) performance. Using data of existing farms can help to study the underlying determinants of differences in farm performance. Moreover, this knowledge supports decision makers in their aim to improve farm sustainability.

Our approach to assess farm performance consists of two steps. First, the economic farm performance will be analyzed. In a second step, environmental resource use will be integrated in the economic analysis to assess farm sustainability performance. In the first two chapters of this part (chapter 5 and 6), farm performance will be studied in the traditional economic way without considering social and environmental issues. First, the determinants of farm performance measured as technical efficiency are analyzed. Empirical results show that both structural and managerial characteristics explain differences in farm performance. Insight of the impact of these determinants helps to understand the driving factors of structural change and how policy may respond to it. In the second chapter the link between structural change and farm performance is investigated. The existence and persistence of differences in efficiency among farms and agricultural subsectors is an important explanation of structural change in agriculture. In the first two chapters, farm performance is measured as technical efficiency. The next step is to measure farm performance in a more complete way by taking environmental and social aspects into account. Integrating environmental considerations into the calculation of agricultural performance is a major research challenge. Therefore, we introduce in chapter 7 the concept of sustainable value to measure contributions towards sustainability. Furthermore, differences in farm sustainability measured in terms of sustainable value creation are explained in this chapter. Once again, the empirical model shows that several managerial and structural characteristics are significant in explaining differences in farm performance. In a last empirical application (chapter 8), the methodology to measure farm sustainability (the sustainable value approach) is combined with frontier methods (efficiency analysis). Efficiency analysis methods are used to construct benchmarks. To assess sustainability, the sustainable value approach is applied using these benchmarks. In this way, the production theoretical underpinning of efficiency analysis (as explained in chapter 5 and 6) enrich the sustainable value approach (as explained in chapter 7).

Before starting with the empirical applications, we repeat the objectives of the empirical part as formulated in chapter 1. The objectives about measuring farm performance in agriculture were:

★ To measure farm efficiency as an indicator of farm performance (chapter 5);

★ To explain differences in farm efficiency (chapter 5);

★ To link farm performance with structural change (chapter 6);

★ To measure the sustainable value of farming as an indicator of farm performance (chapter 7);

★ To explain differences in farm sustainability (chapter 7);

★ To improve sustainability assessment using production frontier benchmarks (chapter 8).

Chapter 5

Factors of farm performance: an empirical analysis of structural and managerial characteristics

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It is much more difficult to measure non performance than performance
—Harold S. Geneen

Abstract

The agricultural sector faces a continuous process of structural change, which has important consequences for productivity and efficiency of farming. A consistent way of monitoring this process, and to support related policy making, is to analyze the performance of agricultural farms with productive efficiency techniques. In this chapter, the impact of managerial and structural characteristics on farm efficiency is analyzed with a stochastic frontier model. First, an overview is given of similar studies looking to relations between structural characteristics, agent factors, and efficiency. Next, an empirical productive efficiency analysis is done on an unbalanced panel of 1018 Flemish farms over a 14-years period (1989-2002). The stochastic production frontier is estimated using the random-effects model with time-invariant efficiency, and with the translog as functional form. Finally, the stochastic production function is extended with extra regressors, to understand why farms differ in their relative efficiency. Empirical results show significant effects of education, the prospect of succession, farm size, type and location, age of farmers, solvency, and dependency on subsidies. Results are discussed in terms of capacities and incentives to perform better. Insight of the impact of these determinants helps to understand the driving factors of structural change and how policy may respond to it.

5.1 Introduction

Due to technological change and changing agricultural policies, in particular in industrial countries since the second half of the 20th century, the structure of the agricultural sector is rapidly changing. Labor leaves the sector, farm enterprizes increase in size and a growing share of agricultural production is produced by a relatively small number of highly specialized farm businesses (OECD, 2002). The changing structure has important consequences for productivity and efficiency of farming, the demand for government services and infrastructures, equity within agriculture, and the well-being of local communities.

To monitor this process and to support policy making it is important to analyze the relationship with the performance of agricultural farms. A consistent and challenging way of doing this is to measure farm efficiency. Existing efficiency analysis techniques help to understand why farms differ in their relative efficiency. Differences in performance raise a lot of relevant policy questions (Poppe and van Meijl, 2004). What are the determinants of these differences? Can these differences be influenced by policy? To whom should support be targeted frontrunners or laggards? Such questions even become more important in the light of nowadays policy changes, e.g., the shift from the mere agricultural sector scope to the more integrated (rural, sustainability) approaches.

In this chapter, we focus on the impact of managerial and structural characteristics on farm performance. This research highlights the interplay between farm efficiency and farm characteristics. First, an overview of similar studies is given. Next, own empirical research is reported, based on the Farm Accountancy Data Network (FADN) from Flemish farms. This research uses stochastic frontier analysis for estimating the production frontier and for calculating firm-level technical efficiencies. In order to analyze the impact of firm-specific factors on efficiency, we enlarged the stochastic production frontier with extra regressors, indicating firm characteristics that are postulated to affect firm efficiency.

5.2 The impact of managerial and structural characteristics on performance

Several studies have attempted to understand variations in farm performance, in particular technical efficiency by differences in e.g., size, organizational type and agent factors. Note that the definition of efficiency is not the same in all studies, most reviewed studies in this section estimate the technical efficiency (Brada and King, 1993, Battese and Coelli, 1995, Hallam and Machado, 1996, O'Neill et al., 1999, Liu and Zhuang, 2000, Piesse and Thirtle, 2000, Brümmer, 2001, Mathijs and Vranken, 2001, Mathijs and Swinnen, 2001, O'Neill et al., 2001, Wilson et al., 2001, Rezitis et al., 2002, Santarossa, 2003, Thirtle and Holding, 2003, Kompas and Che, 2004, Igliori, 2005) but some studies estimate the economic efficiency (Hall and LeVeen, 1978, Herdt and Mandac, 1981, Stefanou and Saxena, 1988, Kalirajan, 1990, Parikh et al., 1995, Munroe, 2001). Remind that the definitions of technical and economic efficiency are described in section 4.2. Furthermore, these studies use different efficiency estimation techniques (SFA and DEA, see section 6.3.2) and they consider different inputs and outputs. This makes a consistent comparison difficult, hence results of particular applications could not be assumed in other applications. On the other hand, these studies give an interesting first indication of the impact of several characteristics on farm efficiency in different situations. Figure 6.1 shows a framework to classify the different characteristics in explaining efficiency. Both agent factors and structural factors have an impact on farm efficiency. Agent factors are managerial characteristics of the farm such as the education level and age of the farm manager. Structural factors are classified in on-farm factors and off-farm factors. Examples of on-farm factors are the farm size, farm type, organizational type and the farm location. Up- and downstream relations and government interventions are examples of off-farm structural factors.

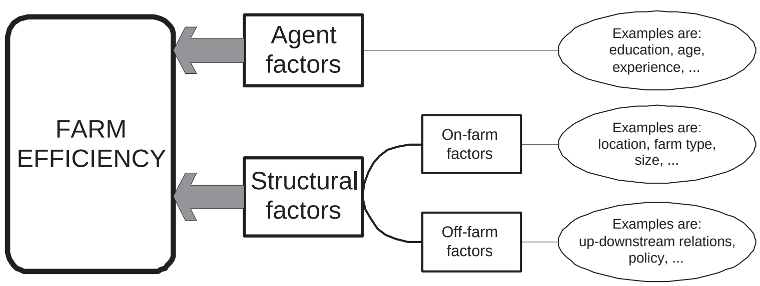

Figure 5.1: Factors affecting technical efficiency

5.2.1 Structural factors affecting efficiency

Farm size Debates concerning the *optimal farm structure* and *optimal farm size* have a long history in agricultural economics (Gorton and Davidova, 2004). Hall and LeVeen (1978) explore the relationship between economic efficiency and farm size. They found for Californian farms that the long-run average cost curve is relatively flat after initially declining rapidly. Igliori (2005) found for farms in the Brazilian Amazon that the smaller farms are less efficient. Larger farms in the UK (Thirtle and Holding, 2003), in the Greek agricultural sector (Rezitis et al., 2002), in wheat farming in Eastern England (Wilson et al., 2001), and in Portuguese dairy farms (Hallam and Machado, 1996) are operating more efficiently than smaller farms. On the other hand, Polish farms greater than 15 ha have lower efficiency than smaller farms (Munroe, 2001). Also for Irish farms (O'Neill et al., 2001) and for Philippine rice farmers (Herdt and Mandac, 1981), a negative relationship between farm size and efficiency was found. Hall and LeVeen (1978) and Seckler and Young (1978) state, however, that factors such as management, resource quality and the overall institutional structure are even more important than farm size. This may be the reason why the observed relationships between size and efficiency are not universal.

The empirical measures used to classify farm size can be criticized (Gorton and Davidova, 2004). Common measures, such as land area, may be inappropriate for capturing differences in farming systems. European size unit measures, based on standard gross margins, have rarely been used as measure of farm size in efficiency studies (Gorton and Davidova, 2004), but can be an acceptable solution (Lepoutre et al., 2004).

Finally, it's important to note that size is a relative concept so the search for a single *optimal size* is futile given the heterogeneity of farming systems and specialization of production factors (Verma and Bromley, 1987).

Organizational type Rent, share, owner-operated, and other organizational forms are different types of agricultural forms (Roumasset, 1995). A wide range of possibilities exists, going from a family farm to a large, factory-style corporation (Allen and Lueck, 1998).

The impact of organizational type on efficiency is in particular an important issue in transitional economics. Mathijs and Swinnen (2001) found with data from former East Germany that partnerships emerging from the large-scale collective and state farms had a lower technical efficiency than family farms. Thiele and Brodersen (1999) found that the differences in ownership and production types are not important in explaining the inefficiencies of East German farms, but the differences in inefficiencies are rather the result of sub-optimal input allocations. Brada and King (1993) found similar results investigating state and private farms in Poland. They conclude that the internal organization of farm units does not explain differences in efficiency. On the other hand, agricultural policies and administrative distribution of inputs are at the origin of a sub-optimal allocation of resources in former Polish agriculture (Brada and King, 1993). Using survey data on Bulgarian and Hungarian crop and dairy farms, Mathijs and Vranken (2001) found that family farms are performing better than corporate farms in the case of crop farms but not for dairy farms.

Farm type Although farms tend to evolve toward more product specialization, a lot of farms are still mixed. The fact that most farms are multi-product firms suggest that the benefits are significant in agriculture. Economies of scope are one of these possible benefits, reflecting the reduced cost associated with producing multiple outputs and the risk-reducing effects of diversification (Chavas, 2001). The efficiency in using resources can differ between farm types. Brümmer (2001) found for Slovenian private farms that the specialized cattle farms are less efficient than other farm types. Also, in the UK less specialized farms are found to be more efficient than other farms (Thirtle and Holding, 2003). However, Hallam and Machado (1996) found for Portuguese dairy farms that the efficiency is independent with the degree of specialization. Finally and contrarily, Santarossa (2003) found that specialized Scottish farms are more efficient.

Other structural factors Differences in efficiency can also be explained by environmental characteristics, such as soil quality, vegetation, altitude, climate, rivers, rain, temperature,... Igliori (2005) found that the presence of forests and rivers are negatively and significantly correlated with efficiency for farms in the Brazilian Amazon. In the case of private farms in Slovenia, Brümmer (2001) showed that farms situated more than 600 meters above sea level are less efficient.

Location is a factor in explaining differences in efficiency, which links the farm location to environmental characteristics. O'Neill et al. (2001) found that farmers in the West of Ireland are less efficient than farmers in the East of Ireland. Scottish farmers in less favored areas are also less efficient. Also for Greek agricultural farms, location is found as an important determinant in explaining differences in efficiency (Rezitis et al., 2002).

Other factors, not always easy to differentiate from organizational factors, are ownership difference, financial factors and technology. For example farms can differ in the amount of owner-occupied land. Thirtle and Holding (2003) show for UK farming that farms with a higher proportion of owner occupied land are more efficient.

Also financial determinants can have an impact on efficiency. Chavas and Aliber (1993) found that medium and long-run debt financing has a positive effect on technical efficiency. O'Neill et al. (2001) and Thirtle and Holding (2003) found that the debt ratio to assets is positively related to efficiency.

Key technology variables can also have an impact on farm efficiency. On Australian dairy farms, one of the important determinants of differences in farm efficiency is the type of dairy shed used (Kompas and Che, 2004).

Finally, also off-farm structural factors can influence firm efficiency (see figure 6.1). Contracting with upstream processors increases efficiency through facilitating the adoption of technology and access to credits. For Hungarian and Bulgarian farms Mathijs and Vranken (2001) found a strong positive effect of contracts with downstream processors on efficiency. Also support payments can influence efficiency. On Irish farms there is a negative relationship between dependency on direct payments and farm efficiency (O'Neill et al., 2001). Also on Hungarian farms inefficiencies are partly explained by subsidies, whereas Hungarian farms that had established export markets were more efficient (Piesse and Thirtle, 2000).

5.2.2 Agent factors affecting efficiency

Age The age of the farm manager may be an indication for experience. Testing a model for technical inefficiency effects on paddy farmers from an Indian village, Battese and Coelli (1995) showed, however, that older farmers are more inefficient than the younger ones. Also, in the UK young farmers are working more efficiently than older ones (Thirtle and Holding, 2003). Other studies report similar results, indicating that older farmers are unwilling or unable to adopt technical innovations (Herdt and Mandac, 1981, Parikh et al., 1995). On the other hand wheat farmers in Eastern England who have more years of managerial experience have higher levels of efficiency (Wilson et al., 2001). Wilson et al. (2001) argue that older farmers are more experienced and take profit of their knowledge to use inputs more efficiently.

In the UK, a positive relationship between age and efficiency was found up to the age of 49 years after which the relationship between age and efficiency became negative (O'Neill et al., 2001). In Chinese agriculture, farm efficiency increases with the age of the primary decision maker before he reaches the age of 40 and declines afterwards (Liu and Zhuang, 2000). Farmers become more skillful as they grew older, but the learning-by-doing effect is attenuated as the farm managers reach the middle age (Liu and Zhuang, 2000).

Education Investment in education can be seen as a strategy to improve agricultural productivity, principally through its complementarity with inputs as fertilizers, pesticides, irrigation, high-yielding varieties, and effective research and extension services (Lockheed et al., 1980). Farmers with more years of schooling tend to be more efficient. In an application on Pennsylvania dairy farms Stefanou and Saxena (1988) found that education and experience play an important role in the level of efficiency. The effect of schooling should be positive as better educated farmers are expected to have more skills to run their farm more efficiently (Kalirajan, 1990, Battese and Coelli, 1993, 1995, Parikh et al., 1995, O'Neill et al., 1999, Liu and Zhuang, 2000, Mathijs and Vranken, 2001, Igliori, 2005). The effects of education are much more likely to be positive in modern agricultural environments than in traditional ones. Hence, the effectiveness of education is enhanced in a modernizing environment (Lockheed et al., 1980).

Management characteristics Including aspects of the decision-making process in explaining differences in efficiency is an important step (Rougoor et al., 1998). Wilson et al. (2001) found that the objectives of maximizing profits and maintaining the environment are positively correlated with the technical efficiency of wheat farmers in Eastern England. Trip et al. (2002) found a statistically significant association between a high intensity of data recording and a high level of result evaluation and efficiency (Trip et al., 2002).

Other agency factors The level of education is only one indicator that determines the knowledge of the manager. Other indicators are for example following extra training, attending workshops and reading specialist publications. O'Neill et al. (1999) and O'Neill et al. (2001) measured the participation of Irish farmers to a training course and found significant positive impact on efficiency. Also farmers, who receive regular visits from advisory services, were working more efficiently (O'Neill et al., 1999, 2001). Those farmers who seek information are also associated with higher levels of technical efficiency (Wilson et al., 2001).

Having a successor for farming activities can also determine farm efficiency. Having a farm successor is significant in explaining the level of technical efficiency on Irish farms (O'Neill et al., 1999).

Human capital matters not only through age and education but also trough gender. Based on survey data on Bulgarian and Hungarian crop and dairy farms, Mathijs and Vranken (2001) found that farms with a higher proportion of woman are more efficient.

5.3 Empirical model: the stochastic frontier model

5.3.1 The basic firm efficiency model

The performance of an enterprize can be defined in different ways, for example ratio indicators, index numbers and relative efficiency scores. An important performance measure is efficiency. Farrell (1957) defined efficiency as the success in producing as large as possible an output from a given set of inputs. This definition implies an efficient production function, or production frontier, to which the actual production is referred. Hence, the standard definition of a production frontier (or function) is that it gives the maximum possible output for a given set of inputs. The production function defines a boundary or frontier. Deviations of observed outputs from this frontier are in principle one-sided and can be taken to reflect inefficiency (Cornwell and Schmidt, 1996).

To measure efficiency one has to known the form of the production function. Frontiers can be estimated with different methods. The two principal methods are data envelopment analysis (DEA) and stochastic frontiers (SFA) (Coelli et al., 1998). DEA involves the use of linear programming methods to construct a non-parametric piece-wise frontier over the data ('an envelope'). Since DEA is non-parametric, it is robust to the kind of specification error that may arise in the choice of functional form of the production frontier. Furthermore, DEA allows easier modeling of multi-product technologies. However, DEA is non-statistical, so the determinants of its inefficiency estimates cannot be determined simultaneously. Furthermore, the DEA approach cannot disentangle inefficiency from random noise. This statistical noise is due to factors outside the control of firms such as weather. This means that any deviation from the frontier is regarded as inefficiency (Cornwell and Schmidt, 1996).

In this chapter we will use stochastic frontier analysis (SFA) for estimating the production frontier. The main argument is that we want to analyze directly the determinants of the efficiency estimates. A disadvantage is the fact that we have to use an aggregated mono-product technology, this means that the efficiency scores will also contain besides the technical effect an allocative effect due to differences of mixed outputs between farms.

Aigner et al. (1977) and Meeusen and Van den Broeck (1977) introduced the stochastic frontier production:

$$y_{it} = \alpha + f(x_{it}, \beta) + v_{it} - u_i \tag{5.1}$$

With v_i as a two sided i.d.d. (independent and identically distributed) error term and with u_i as a non-negative i.d.d. error term. Both v_i and u_i are assumed to be independent of the input variables x_{ik} and of each other.

The error term v_i accounts for measurement error and random errors such as weather, strikes and luck. The error term u_i measures the technical efficiency. One of the advantages of panel data is that firm-specific technical efficiencies can be estimated without assumptions about the distribution of the errors. Another advantage is the possibility to estimate the firm specific technical efficiencies consistently (Cornwell and Schmidt, 1996).

The parameters of the stochastic frontier model can be estimated using the maximum-likelihood method. Our model will be estimated using unbalanced panel data. We tested different estimation techniques given the panel structure of the data. Instead of treating the u_i as fixed (fixed effects model) we assumed that the u_i are (Cornwell and Schmidt, 1996): (i) independent and identically distributed (i.d.d.) from a one-sided distribution ($u_i > 0$) and (ii) uncorrelated with x_{it} and v_{it} for all t. The used model is called the random effects panel data formulation with time-invariant inefficiency. It's worth mentioning that the benefits of panel data come at the expense of another strong assumption that firm efficiency does not vary over time (Cornwell and Schmidt, 1996).

The use of stochastic frontier analysis implies the choice of the functional form. The Cobb-Douglas functional form has been commonly used in the estimation of frontier models. The simplicity of this functional form is very attractive. The Cobb-Douglas production function has fixed input elasticities and returns to scale. A number of alternative functional forms have also been used in the frontier literature such as the translog functional form (Christensen et al., 1973). An important advantage is that the translog form imposes no restrictions upon returns of scale or substitution possibilities (Coelli et al., 1998).

The basic specification for the model in equation 5.1 can be written as:

$$\begin{aligned}
Ln(Yield_{it}) = \alpha &+ \beta_1 ln(Lab_{it}) && +\beta_2 ln(Cap_{it}) && +\beta_3 ln(Area_{it}) \\
&+\beta_4 ln(Med_{it}) && +\beta_5 Time_{it} && +\tfrac{1}{2}\beta_6 [ln(Lab_{it})]^2 \\
&+\tfrac{1}{2}\beta_7 [ln(Cap_{it})]^2 && +\tfrac{1}{2}\beta_8 [ln(Area_{it})]^2 && +\tfrac{1}{2}\beta_9 [ln(Med_{it})]^2 \\
&+\tfrac{1}{2}\beta_{10} [Time_{it}]^2 && +\beta_{11} ln(Lab_{it}) * ln(Cap_{it}) \\
&+\beta_{12} ln(Lab_{it}) * ln(Area_{it}) && +\beta_{13} ln(Lab_{it}) * ln(Med_{it}) \\
&+\beta_{14} ln(Lab_{it}) * Time_{it} && +\beta_{15} ln(Cap_{it}) * ln(Area_{it}) \\
&+\beta_{16} ln(Cap_{it}) * ln(Med_{it}) && +\beta_{17} ln(Cap_{it}) * Time_{it} \\
&+\beta_{18} ln(Area_{it}) * ln(Med_{it}) && +\beta_{19} ln(Area_{it}) * Time_{it} \\
&+\beta_{20} ln(Med_{it}) * Time_{it} && +v_{it} - u_i
\end{aligned} \tag{5.2}$$

The dependent variable $Yield_{it}$ is the deflated total yield (gross output) in Euros of the farm i in year t. The inputs used in the production process are: (i) the total amount of labor of the farm i in year t (Lab_{it}), (ii) the total amount of deflated farm capital[2] of the farm i in year t (Cap_{it}), (iii) the total amount of utilized agricultural area of the farm i in year t ($Area_{it}$) and (iv) the total amount of intermediate consumption of the farm i in year t (Med_{it}). $Time_{it}$ is a time trend and v_{it} and u_i are defined as above.

We have estimated both the Cobb-Douglas and the translog functional forms. Now we can test the null hypothesis that the Cobb-Douglas form is an adequate representation of the data, given the specifications of the translog model. This can be tested by using the generalized likelihood-ratio test (Coelli et al., 1998). The H_0 and H_1 are:

$H_0 : \beta_6 = \beta_7 = \beta_8 = \beta_9 = \beta_{10} = \beta_{11} = \beta_{12} = \beta_{13} = \beta_{14} = \beta_{15} = \beta_{16} = \beta_{17} = \beta_{18} = \beta_{19} = \beta_{20} = 0$
$H_1 :$ *at least one* β $(6 \rightarrow 20) \neq 0$

In our case, the translog functional form is preferred to the Cobb-Douglas functional form. The hypothesis that the coefficients of the second order term in equation 5.2 were 0, was rejected[3]. We can calculate predictions of firm-level technical efficiencies following Battese and Coelli (1988).

5.3.2 The extended firm efficiency model

Determinants of technical inefficiencies among firms can be investigated by regressing the predicted inefficiency effects, obtained from an estimated stochastic frontier upon a vector of firm-specific factors such as firm size, age and education of the managers. This is called the second-stage analysis. But, there is a problem with this two-stage approach. In the first stage the inefficiency effects are assumed to be independently and identically distributed (i.d.d.) for predicting the values of the inefficiency effects (the approach of Jondrow et al. (1982)). However, in the second stage, the predicted inefficiency effects are assumed to be a function of firm-specific factors, which implies that they are not i.d.d. So this approach can lead to biased results. As a solution the parameters of the stochastic frontier and the inefficiency model should be estimated simultaneously (Reifschneider and Stevenson, 1991, Battese and Coelli, 1995).

[2]Farm capital is calculated as total capital minus land capital, in this way overlap with the land input (Area) is avoided.

[3]The test statistic is calculated as (Coelli et al., 1998): $LR = -2[ln(L(H_0)) - ln(L(H_1))]$ where $L(H_0)$ and $L(H_1)$ are the values of the likelihood functions under the null and alternative hypotheses. If H_0 is true this test statistic is usually assumed to be asymptotically distributed as a χ^2 random variable with degrees of freedom equal to the number of restrictions involved (in this case 15). Reject H_0 in favor of H_1 if LR exceeds $\chi^2(\alpha)$. Thus the critical value for a test of size ($\alpha = 0,05$) is 25,00 (Neter et al., 1996). In this case the LR exceeds the critical value, so we have to reject H_0 and take the translog formulation as functional form.

In order to analyze the impact of firm-specific factors on efficiency, we extend the stochastic production function with extra regressors, indicating firm characteristics that are postulated to affect firm efficiency (Reifschneider and Stevenson, 1991)[4]. The inefficiency term u_i is decomposed in several systematic influences related to specific firm variables (z_{it}) and one non-negative random error term w_i capturing the residual unexplained firm technical inefficiency:

$$u_i = g(z_{it}, \beta) + w_i \tag{5.3}$$

Making the combination of the equations 5.1 and 5.3, we obtain the firm-specific efficiency model:

$$y_{it} = \alpha + f(x_{it}, \beta) + g(z_{it}, \beta) + v_{it} - w_i \tag{5.4}$$

Important to note on this approach is the fact that this analysis assumes unconditionality. For example take the size of a firm. We analyze the impact of the size on firm efficiency assuming that efficiency has no impact on size. An example of a conditional analysis between technical efficiency and farm size has been made by Alvarez and Arias (2004). They found that more efficient farmers increase the size of their operations. But on the other hand, Alvarez and Arias (2004) found the same positive relation between efficiency and size in the unconditional approach as in the conditional approach[5].

[4] An other but similar approach is proposed by Battese and Coelli (1995)

[5] The relationship between size and efficiency was the same, but stronger in the unconditional than in the conditional approach

5.4 Empirical efficiency analysis of Flemish farms

5.4.1 Data

This study uses farm accountancy data from a group of 1018 Farms in Flanders, belonging to the Belgian Farm Accountancy Data Network (FADN). The Belgian FADN data are collected and managed by the Centre for Agricultural Economics (CAE)[6]. Data of 1018 farms for a period of 14 years (1989-2002) are available. The number of observations of each farm differs from 1 to 14 (unbalanced panel data). In total the sample contains 8926 observations of 1018 different farms. Table 5.1 shows some descriptive statistics of the variables.

Table 5.1: Descriptive statistics

Variable	mean	Std. Dev.	Min.	Max.
Gross output (Euro)	186910	141368	1420	1499698
Agricultural area (ha)	31	23	0	264
Labor (in full-time equivalent units)	1.55	0.49	0.11	6.35
Farm capital (Euro)	246667	159921	248	1188975
Intermediate consumption (Euro)	105451	87629	2335	853650
Age of farm manager	43	11	19	77
Solvency[1]	0.46	0.32	0	1
Size unit[2]	19	11	1	125
Share land in property[3]	0.27	0.26	0	1
Years accounting by CAE[4]	16 years	10 years	1 year	44 years
Subsidies interest[5] (Euro)	2420	2991	0	31920
Subsidies revenues[6] (Euro)	455	1412	0	41674
Subsidies costs[7] (Euro)	0.23	5.35	0	165
Subsidies income[8] (Euro)	4290	6408	0	136763

[1] measured as own capital divided by total capital
[2] based on the standard gross margin (FADN, n.d.)
[3] the amount of land in property over the total amount of utilized farm land
[4] the total number of years that the Centre for Agricultural Economics (CAE) do/did the bookkeeping of the farm
[5] subsidies on investments (interest support)
[6] subsidies on animal products (subsidies on sale and purchase of animals are not included)
[7] subsidies on farm costs (subsidies on investments are not included)
[8] direct payments to producers: suckler cow premium, slaughter premium, set-aside premium, arable crops hectare aid,. . .

The 8926 Flemish observations belong to six different farm types (FADN-typology): (i) specialist field crops (960 observations), (ii) specialist grazing livestock (3414 observations), (iii) specialist pigs (1493 observations), (iv)

[6]The Belgian FADN-data were collected by the former CAE. Since 2006, the CAE is assimilated into the Institute for Agricultural and Fisheries Research. The Flemish FADN-data are now collected by the agricultural monitoring and study service of the Flemish Ministry for Agriculture.

mixed cropping (164 observations), (v) mixed livestock (1565 observations) and (vi) mixed crops-livestock (1330 observations). Further, the five different Flemish provinces are used to group the observations according to location: (i) Antwerpen (2048 observations), (ii) Vlaams-Brabant (1624 observations), (iii) West-Vlaanderen (2066 observations), (iv) Oost-Vlaanderen (1865 observations) and (v) Limburg (1323 observations). Next, each farmer in our dataset is each year asked about his expectancy about his succession. There are three possibilities (successor1, successor2, successor3): (i) there is a successor (1146 observations), (ii) it's not clear yet if there is a successor (3815) and (iii) there is no successor (3965 observations). Finally, we can divide all 1018 Flemish farmers into five different levels of agricultural education (diploma1 till diploma5): (i) certificate of higher agricultural education (19 farmers), (ii) certificate of higher technical agricultural education (175 farmers), (iii) certificate of lower technical agricultural education (219 farmers), (iv) certificate of technical and vocational agricultural education (55 farmers) and (v) no certificate of agricultural education (550 farmers).

5.4.2 The basis model results

The results of the estimation of the translog stochastic production frontier defined in equation 5.2 are given in table 5.2.

Table 5.2: Estimation coefficients of the translog stochastic production frontier

Variables	Coefficient	st. error	Variables	Coefficient	st. error
$Constant$	1.3208***	0.0841	$ln(Lab_{it}) * Time_{it}$	0.0047***	0.0017
$Ln(Lab_{it})$	0.4441***	0.0671	$ln(Lab_{it}) * ln(Cap_{it})$	0.0187	0.0168
$Ln(Cap_{it})$	0.3251***	0.0276	$ln(Lab_{it}) * ln(Area_{it})$	0.0016	0.0037
$Ln(Area_{it})$	0.0552***	0.0071	$ln(Lab_{it}) * ln(Med_{it})$	-0.1164***	0.0162
$Ln(Med_{it})$	0.3223***	0.0225	$ln(Cap_{it}) * ln(Area_{it})$	0.0084***	0.0015
$Time_{it}$	0.0358***	0.0034	$ln(Cap_{it}) * ln(Med_{it})$	-0.1172***	0.0046
$[ln(Lab_{it})]^2$	0.0579***	0.0121	$ln(Area_{it}) * ln(Med_{it})$	-0.0134***	0.001
$[ln(Cap_{it})]^2$	0.0219***	0.0033	$ln(Cap_{it}) * Time_{it}$	0.0049***	0.0009
$[ln(Area_{it})]^2$	0.0052***	0.0002	$ln(Area_{it}) * Time_{it}$	0.0012***	0.0002
$[ln(Med_{it})]^2$	0.1305***	0.003	$ln(Med_{it}) * Time_{it}$	-0.0076***	0.0008
$[Time_{it}]^2$	-0.0021***	0.0001			
Number of observations	8926		Variances: $\sigma^2(v)$	0.0227	
Iterations completed	31		Variances: $\sigma^2(u)$	0.0685	
Log likelihood function	3226.3		Variances: σ^2	0.0912	

* significant at 10%, **significant at 5%, ***significant at 1%

The firm efficiency is directly estimated in equation 5.2 trough the firm specific variable u_i which measures the deviation of individual firms' output from the production frontier (Jondrow et al., 1982).

We can test whether there are no technical inefficiency effects in the model. Under the null hypothesis, $H_0 = \gamma^7$, the model is equivalent to the traditional average response function, without the technical inefficiency effect u_t.

We test the following hypotheses:

$H_0 : \gamma = 0$ (*there is no inefficiency*)
$H_1 : \gamma \neq 0$ (*there is inefficiency*)

In this case the LR exceeds the critical value, thus we have to reject H_0 and hence the traditional response function is not an adequate representation of the data[8]. Hence, *there is technical inefficiency*.

Knowing that there is technical inefficiency we can calculate predictions of firm-level technical efficiencies as in Battese and Coelli (1988). The mean efficiency of all 1018 firms equals 81.6%, we observe a minimum efficiency of 47% and a maximum efficiency of 100%. Figure 5.2 shows the histogram of all predictions of the 1018 firm-level technical efficiencies. These results show a wide range in the level of technical efficiencies across all farms.

5.4.3 Characteristics affecting efficiency: empirical results

The impact of structural and managerial firm characteristics on efficiency is analyzed with the model mentioned in 5.3.2. It is assumed that the farm characteristics only affect the level of technical efficiency thereby systematically shifting the production frontier up- or downwards. Variables indicating structural aspects are: (i) location (*Antwerpen, Brabant, Oost-Vlaanderen, Limburg and West-Vlaanderen*), (ii) farm type (*specialist field crops, specialist grazing livestock, specialist pigs, mixed cropping, mixed livestock and mixed crops-livestock*), (iii) farm size (*size unit*), (iv) solvency, (v) relative amount of land in property (*share land*) and (vi) dependency on subsidies (*subsidiesinterest, subsidiesrevenues, subsidiescosts and subsidiesincome*). Variables indicating managerial aspects are: (i) age, (ii) education level (*diploma1 till diploma5*), (iii) number of years in accounting network (*yearsaccounting*) and (iv) succession (*successor1, successor2 and successor3*). The results are shown in table 5.3.

[7] γ can be calculated as in (Battese and Corra, 1977): $\gamma = \frac{\sigma^2(u)}{\sigma^2(u)+\sigma^2(v)} = \frac{\sigma^2(u)}{\sigma^2} = \frac{0.6848}{0.09116} = 0,75$. A value of γ of zero indicates that the deviations from the frontier are due entirely to noise, while a value of one would indicate that all deviations are due to technical inefficiency (Coelli et al., 1998).

[8] The one-side generalized likelihood-ratio test should be performed when ML estimation is involved because this test has the correct size (Coelli, 1995):$LR = -2[ln(L(H_0)) - ln(L(H_1))]$ where $L(H_0)$ and $L(H_1)$ are the values of the likelihood functions under the null and alternative hypotheses. H_0 is rejected in favor of H_1 when LR exceeds $\chi^2(\alpha)$.

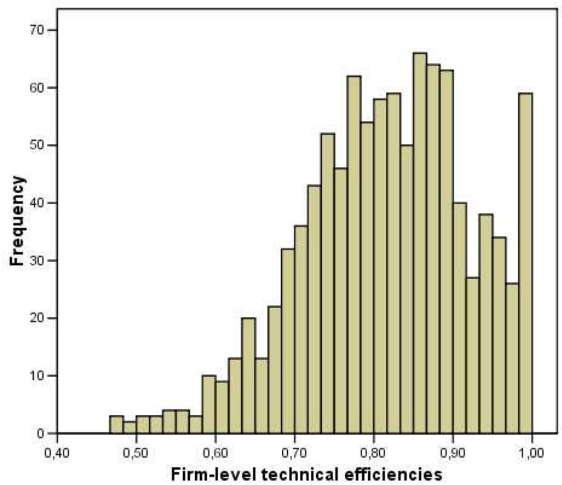

Figure 5.2: The frequency distribution of the predictions of firm-level technical efficiencies

Impact of managerial characteristics

Education level Several farm aspects have a significant impact on efficiency (table 5.3). The level of education is divided in five groups (from diploma1 till diploma5). Diploma1 indicates that the manager has a very high level of education, diploma5 indicates a very low level of education. The mean firm efficiencies are shown in table 5.4. Table 5.3 and 5.4 show that managers with the highest level of education are more efficient than managers with lower education. To test if the mean efficiency of the several education levels are significant different of each other, we execute a one-way anova (analysis of variance). We found that the F-value exceeds the critical value so this means that not all means are the same.

The level of education is only one indicator that determines the knowledge of the manager. Other indicators are for example following extra training, attending workshops and reading specialist publications. Unfortunately, only data about the level of education is available.

Age of the farmer The impact of age of the agricultural manager on efficiency is negative, indicating that older managers are less efficient. So age has

Table 5.3: Estimation coefficients of the enlarged stochastic production frontier

Variables	Coefficient	st. error	Variables	Coefficient	st. error
$Constant$	1.1861 ***	0.0920			
$Ln(Lab_{it})$	0.3953 ***	0.0615	$Oost-Vlaanderen$	0.0285 ***	0.0101
$Ln(Cap_{it})$	0.2887 ***	0.0321	$Limburg$	0.0134	0.0117
$Ln(Area_{it})$	0.0587 ***	0.0082	$Spec.\ field\ crops$	-0.0324 **	0.0135
$Ln(Med_{it})$	0.5459 ***	0.0278	$Spec.\ grazing\ livestock$	-0.0449 ***	0.0110
$Time_{it}$	0.0506 ***	0.0034	$Mixed\ cropping$	-0.0792 ***	0.0205
$[ln(Lab_{it})]^2$	0.0405 ***	0.0108	$Mixed\ livestock$	-0.0368 ***	0.0109
$[ln(Cap_{it})]^2$	0.0165 ***	0.0041	$Mixed\ crops-livestock$	-0.0473 ***	0.0127
$[ln(Area_{it})]^2$	0.0049 ***	0.0004	$Years\ accounting$	0.0013 ***	0.0003
$[ln(Med_{it})]^2$	0.0837 ***	0.0036	$Share$	0.0690 ***	0.0117
$[Time_{it}]^2$	-0.0018 ***	0.0001	$Diploma2$	-0.0485 ***	0.0148
$ln(Lab_{it}) * Time_{it}$	0.0044 ***	0.0015	$Diploma3$	-0.0423 ***	0.0144
$ln(Lab_{it}) * ln(Cap_{it})$	0.0135	0.0160	$Diploma4$	-0.0694 ***	0.0159
$ln(Lab_{it}) * ln(Area_{it})$	-0.0010	0.0035	$Diploma5$	-0.0536 ***	0.0138
$ln(Lab_{it}) * ln(Med_{it})$	-0.1017 ***	0.0146	$Successor1$	0.0349 ***	0.0073
$ln(Cap_{it}) * ln(Area_{it})$	0.0075 ***	0.0018	$Successor2$	0.0126 **	0.0055
$ln(Cap_{it}) * ln(Med_{it})$	-0.0964 ***	0.0060	Age	-0.0025 ***	0.0003
$ln(Area_{it}) * ln(Med_{it})$	-0.0141 ***	0.0010	$Solvency$	-0.0789 ***	0.0092
$ln(Cap_{it}) * Time_{it}$	0.0019 **	0.0009	$Sizeunit$	0.0079 ***	0.0003
$ln(Area_{it}) * Time_{it}$	0.0013 ***	0.0002	$Subsidies\ interest^1$	-0.0210 ***	0.0007
$ln(Med_{it}) * Time_{it}$	-0.0085 ***	0.0008	$Subsidies\ revenues^1$	-0.0229 ***	0.0012
$Antwerpen$	0.0633 ***	0.0093	$Subsidies\ costs^1$	0.1008	0.6130
$Brabant$	-0.0184 *	0.0113	$Subsidies\ income^1$	-0.0029 ***	0.0005
Number of observations	8926				
Iterations completed	55				

* significant at 10%, ** significant at 5%, *** significant at 1%

[1] the subsidies are calculated as a percentage of total revenues, indicating the dependency on payments

the omitted variable for location is *West-Vlaanderen*, for education *diploma1*, for succession *successor3* and for farm type *specialist pigs*

Table 5.4: Mean firm efficiency of different education levels

Education level	mean efficiency	Number of observations	Std. Dev.
Diploma1	0.8455	19	0.0931
Diploma2	0.8367	175	0.0937
Diploma3	0.8336	219	0.0981
Diploma4	0.8052	55	0.0899
Diploma5	0.8016	550	0.1141

F-value one-way anova = 6.38 (5% critical value = 2.37)

an inverse impact on efficiency in this sample. Does this mean that experience has no impact on farm performance? Older farmers could benefit from their experience to use inputs more efficiently. The problem is that other variables intervene. Further exploration of the data reveals that there is a positive link between age and solvency: in general older farmers have less debts than the younger ones. Also the outcome of the education variable is linked to age: in our data base, in 2002, the average age of the highest educated farmers was about 41 years, while the average age of the lowest educated farmers was 48 years. Further research is needed to unravel these interactions. The relation between age and efficiency will be analyzed in more detail in section 5.5.

Farm bookkeeping The coefficient of the variable *yearsaccounting* is significant and positive, meaning that farmers who do the bookkeeping for many years in the accounting system are operating more efficient. All farmers in our data sample do the bookkeeping in the FADN accounting system but the number of years is different. A possible explanation of the positive effect is the fact that farmers get feedback and in this way can improve their efficiency.

Farm succession In the sample three succession indicators are defined. The first indicates that the prospect of a successor for the present manager is almost certain (*successor1*), the second means that it is not yet clear whether or not there will be a successor (*successor2*), the third indicator indicates that there is no successor available (the omitted variable). Observing the results of the inefficiency model in table 5.3, we find that agricultural firms with successor are more efficient than farms without a successor. Also farms with uncertainty about succession have a significant higher efficiency.

Impact of structural characteristics

Agricultural (sub)sector The mean firm efficiencies of each agricultural sector are shown in table 5.5.

Table 5.5: Mean firm efficiency of different agricultural sectors

Agricultural sector	mean efficiency	Number of observations	Std. Dev.
Specialist field crops	0.8338	960	0.1102
Specialist grazing livestock	0.8218	3414	0.1079
Specialist pigs	0.8676	1493	0.0959
Mixed cropping	0.7718	164	0.0897
Mixed livestock	0.8228	1565	0.0846
Mixed crops-livestock	0.7857	1330	0.1032

Observing the results in table 5.5, the mean efficiency of the specialist pig farms is the highest. The sectors mixed cropping and mixed crops-livestock score the lowest efficiency. To obtain higher (technical) efficiency, farmers have to specialize, this is consistent with economic theory. Furthermore, note the smaller standard deviations of the mixed sectors (and the specialist pigs sector), indicating smaller differences in efficiency between farms within the mixed agricultural sectors. Mixed farms spread their risk.

Furthermore, we can test if the mean efficiency of the several different agricultural sectors are significant different of each other. We formulate the following hypothesis:

$H_0 : \mu_{sector1} = \mu_{sector2} = \mu_{sector3} = \mu_{sector4} = \mu_{sector5} = \mu_{sector6}$
$H_1 : at\ least\ one\ \mu_{sector}\ is\ different$

To test the H_0, we use an one-way anova. The test-value F in this case equals 104.0. The F-value exceeds the critical value (2.21) so we have to reject H_0 in favor of H_1. This means that not all means (of efficiency from each agricultural sector) are the same. We can compare the different means by executing a post hoc multiple comparison one way anova. Such a test suggest that the mean efficiency in specialist pigs sector is significant higher than the mean efficiency in the specialist field crops sector. This mean is significant higher than the mean efficiency in the specialist grazing livestock and mixed livestock sectors. Those mean efficiencies are significant different from the mean efficiencies in the mixed cropping sector and in the mixed crops-livestock sector.

With respect to the farm type, the results of the firm-specific inefficiency model in table 5.3 show that the estimated coefficients are significant. In the model, the omitted sector is the specialist pigs sector. These results show that all agricultural sectors have a significant lower efficiency then the specialist pigs sector.

Farm location The Flemish firms are all situated in one of the five Flemish provinces. The five provinces are Antwerpen, Vlaams-Brabant, Limburg, Oost-Vlaanderen and West-Vlaanderen (= the omitted province in table 5.3). The results show that the efficiency in the province Antwerp and Oost-Vlaanderen are significantly higher and the efficiency in Brabant is significantly lower than in West-Vlaanderen.

The effect of farm location on efficiency is significant, but certainly not easy to explain. Differences in soil-type, landscape, erosion, easiness of cultivation are possible explanations but all this characteristics are linked and the reality is too complex to give a clear explanation of the significant different effect on efficiency of those Flemish regions. Also local socio-economic differences (e.g. local agricultural policy, the presence of experimental farms,. . .) could explain the effect of farm location on efficiency.

Farm solvency Also financial determinants have an impact on efficiency. In our model we incorporate the financial parameter solvency. If solvency equals 1, all capital is financed with farmers money (net worth) and if solvency equals 0, all capital is financed with debts. Table 5.3 shows that solvency has a negative impact on efficiency. This means that farmers with low solvency rates are more efficient than farmers with higher solvency rates. A possible explanation is that farmers with a low solvency have higher repayment obligations. In this way those farmers are stimulated (forced) to work more efficiently. But also other aspects (e.g., policy measures) could explain this result. Note that our analysis is unconditional, it's also possible that efficient farms have higher investment rates and thus lower solvency rates (if they are using more loan capital to pay the investments).

Farm size To analyze the impact of size on efficiency, we also added a size unit in the inefficiency model. Table 5.3 shows that size has a significant positive impact on efficiency. So larger farms are working more efficiently than smaller ones.

Land property Also, the share of land in own property has a significant impact on efficiency. Farms with a higher share of owned land are more efficient. A possible explanation is that farmers will do a better job in working on the land that they own.

Dependency on support payments Finally, the impact of the dependency on support payments is found to be significant. The more a farm depends on support payments, the lower its efficiency in using its resources.

Economic theory suggest that subsidies are lowering efficiency because they are disturbing the optimal functioning of markets. In this sample we have divided the subsidies received by the farmer in four groups. In fact, we measure in our model the dependence on support payments. Table 5.3 shows clearly that subsidies have a negative effect on efficiency. The greater a farm's dependence on support payments, the lower its efficiency in using its resources. O'Neill et al. (2001) found on Irish farms also a negative relationship between dependence on direct payments[9] and farm efficiency. In our data set we have four types of grants and as explained in section 5.4.1 the two most important are interest subsidies and income subsidies. The two other categories contains only a small part of total money support to farmers. The category of income subsidies consists of many different income grants. The most common are arable area payments, slaughter premiums and suckler cow premiums. The last European agricultural reform (MidTerm Review) will bring all this direct income support premiums in one single support premium based on a reference period. To receive the support, farmers have to fulfill some conditions. Farmers have to respect environmental, food safety and animal welfare standards. More important in this analysis is the fact that producers in order to qualify for full payment, must set aside a part of their land for example 5% or 10%. There is no obligation on small producers to set aside land, but they may do so on a voluntary basis. So this means that farmers set aside land to receive income

[9] O'Neill et al. (2001) measured the dependence as a share of direct payments in gross margin

subsidies, this can partly explain the negative impact of income subsidies on efficiency. The other important subsidy category are the interest subsidies. At this moment the Flemish agricultural investment fund[10] provides farmers grants for investments and establishments. To receive this grants you have to be a farmer and use economic bookkeeping. There are in this case no restrictions on land use, intermediate consumption and labor. Nevertheless there are restrictions on the kind of investments. There are also different investment categories[11]. For example environmental investments get the highest support. So the different investment categories and changing policies may influence the impact of interest subsidies on efficiency.

Despite of this, we can say that increasing efficiency by providing subsidies to farmers is not a good policy measure. Moreover, policy makers have to take this in account when providing support to achieve other objectives (for example taking care of the countryside)[12]. The positive association between dependence on subsidies and farm technical inefficiency suggests that EU agricultural policy is encouraging a less competitive agricultural sector (O'Neill et al., 2001). Furthermore, the dependence on grants may not only increase inefficiency but also slow down the take-up of technical innovations. On the other hand, there are environmental benefits arising from the more extensive farming methods encouraged by the present policy (O'Neill et al., 2001). Quantifying the negative effects on agricultural productivity and the positive effects on the environment of the payments regime are important issues in assessing the overall benefits of EU's Common Agricultural Policy. This major research challenge is not explored in this chapter.

It's important to note that the used approach is unconditional. So we assume that subsidies have an impact on efficiency but efficiency has no impact on subsidies. To solve a part of this problem the same inefficiency model was tested but instead of using the subsidies in year t the subsidies in year t-1 were used. The results were similar: dependency on payments has a significant negative effect on efficiency[13].

5.4.4 An overview of the impacts on efficiency

The parameter estimates for the inefficiency model, presented in table 5.3, and summarized in table 5.6, only indicate the direction of the effects of these variables upon inefficiency (Wilson et al., 2001). Through differentiating each

[10] This fund is called VLIF (Vlaams LandbouwInvesteringsFonds)

[11] This categories and their interpretation are changing quite a lot. Policy makers try in this way to implement their policies.

[12] This will certainly not mean that policy makers have to stop subsidizing certain activities but they have to make correct cost-benefit analysis, incorporating the trade-off between subsidies and efficiency.

[13] The coefficients of the lagged estimation had the same sign but were smaller

Table 5.6: An overview of some determinants of efficiency

Firm characteristic	Significant impact[1]	Direction of impact[2]	quasi-elasticity
Firm size	yes	+	0.1843
Firm solvency	yes	-	-0.0445
Firm accounting	yes	+	0.0254
Dependency on interest subsidies	yes	-	-0.0362
Dependency on revenue subsidies	yes	-	-0.0067
Dependency on costs subsidies	no		
Dependency on direct income support	yes	-	-0.0080
Age of firm manager	yes	-	-0.1337
Share own land	yes	+	0.0228
Education of firm manager	yes		
Succession of firm manager	yes		
Number of years since take over	no		
Firm location	yes		
Firm sector	yes		

[1] Significant impact on efficiency (yes or no)
[2] Direction of impact positive(+) or negative (-)

of the explanatory variables in the inefficiency model with respect to each of the inefficiency effects variables (evaluated at their mean values), we can calculate the quasi-elasticities for each firm variable (z) as: $\varepsilon_z = \frac{du}{dz}\frac{\tilde{z}}{\tilde{u}}$. $\tilde{u}$ stands for the estimated mean efficiency of our sample, $\tilde{z}$ is the mean value of the firm variable in question. The quasi-elasticities are shown in table 5.6. The impact of size on efficiency has the highest elasticity. A 10% increase in size will result in a 1.8% increase in efficiency. Observing the different subsidies categories, we see that especially the dependency on interest subsidies will lower the firm efficiency. The impact of the other subsidies is much lower. Note that the size of the effects, measured by the quasi-elasticities, is only valid for small changes.

5.5 The impact of farmer's age on efficiency

In this section we analyze the impact of the farmer's age on efficiency in more detail. As indicated in section 5.4.3 the age of the firm manager has a significant impact on efficiency. We found that age has an inverse impact on efficiency in this sample. Does this mean that experience has no impact on farm performance? Is a farmer with no (or little) experience (e.g., a starter) more efficient than a farmer with 10 years of experience? Older farmers can be more experienced and can use their knowledge to use inputs more efficiently. To study the impact of age in more detail, we expand our model and we will also analyze the interplay between age and solvency and between age and education. We will use the same data and model as in the previous sections.

5.5.1 Age, solvency and education

Figure 5.3 shows the link between the age of the firm manager and the farm solvency for all farms in our set in 2002. It is clear that in general older farmers have less debts (a higher solvency rate) than younger ones. Furthermore, the education of the farm manager is linked with farmer age. In 2002 the average age of the highest educated farmers (diploma 1) was about 41 years while the average age of the lowest educated farmers (diploma 5) was 48 years in our data set.

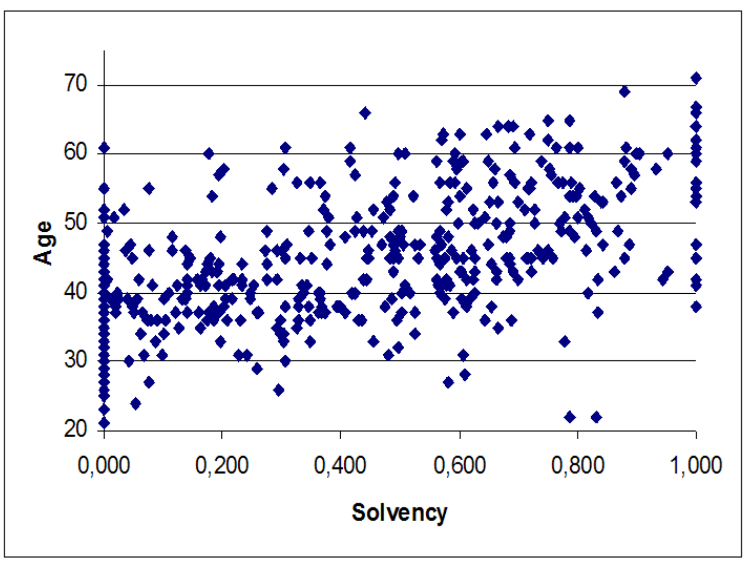

Figure 5.3: The link between farmer's age and farm solvency

5.5.2 Age and efficiency

To analyze the impact of age on efficiency, we enlarge our stochastic frontier with the variables age^2, $age \times solvency$ and $age \times diploma$. The results of the estimation of this inefficiency model can be found in table 5.7[14].

Table 5.7: Estimation coefficients of the extended stochastic production function with focus on age

Variables	Coefficient	st. error	Variables	Coefficient	st. error
Constant	1,0658 ***	0,1002	$ln(Lab_{it}) * Time_{it}$	0,0048 ***	0,0016
$Ln(Lab_{it})$	0,5569 ***	0,0648	$ln(Cap_{it}) * Time_{it}$	0,0045 ***	0,0010
$Ln(Cap_{it})$	0,2841 ***	0,0312	$ln(Area_{it}) * Time_{it}$	0,0012 ***	0,0002
$Ln(Area_{it})$	0,0435 ***	0,0079	$ln(Med_{it}) * Time_{it}$	-0,0100 ***	0,0009
$Ln(Med_{it})$	0,4828 ***	0,0269	*Spec. field crops*	-0,0430 ***	0,0141
$Time_{it}$	0,0448 ***	0,0035	*Spec. grazing livestock*	-0,0403 ***	0,0113
$[ln(Lab_{it})]^2$	0,0590 ***	0,0114	*Mixed cropping*	-0,1006 ***	0,0234
$[ln(Cap_{it})]^2$	0,0130 ***	0,0042	*Mixed livestock*	-0,0395 ***	0,0119
$[ln(Area_{it})]^2$	0,0030 ***	0,0004	$Mixed\ crops - livestock$	-0,0604 ***	0,0132
$[ln(Med_{it})]^2$	0,0909 ***	0,0036	$successor1$	0,0342 ***	0,0073
$[Time_{it}]^2$	-0,0018 ***	0,0001	$successor2$	0,0176 ***	0,0048
$ln(Lab_{it}) * ln(Cap_{it})$	0,0031	0,0165	age	0,0085 ***	0,0017
$ln(Lab_{it}) * ln(Area_{it})$	-0,0024	0,0036	$(age)^2$	-0,0001 ***	0,0000
$ln(Lab_{it}) * ln(Med_{it})$	-0,1243 ***	0,0154	$age \times solvency$	-0,0006 ***	0,0002
$ln(Cap_{it}) * ln(Area_{it})$	0,0087 ***	0,0017	$age \times diploma$	-0,0001 **	0,0000
$ln(Cap_{it}) * ln(Med_{it})$	-0,0917 ***	0,0064	*size unit*	0,0085 ***	0,0003
$ln(Area_{it}) * ln(Med_{it})$	-0,0146 ***	0,0010			
Number of observations	8926				
Iterations completed	43				

* significant at 10%, ** significant at 5%, *** significant at 1%

Table 5.7 shows clearly that age has a positive impact on efficiency. The impact of age on efficiency is first increasing but at a certain age the impact is decreasing indicated by age^2. Figure 5.4 shows the relation between efficiency and age. This figure should be interpreted as follows. The relation between efficiency and age of a farmer with a solvency of zero and the highest level of education is indicated by the black line in figure 5.4. In fact, we see that in that case a farmer of 60 years has an efficiency of 14% (0.14) but this means that the farmer (of 60 years old) has a higher efficiency (14%) compared to a farmer of the age of zero years, which is in fact meaningless. Therefore, it is more useful to read figure 5.4 by comparing two observations. For example consider the same farmer of 60 years old and a farmer with the same characteristics of 50 years old, we observe a difference in efficiency of 2% (0.02), because in this case the impact of age of a farmer of 50 years old is more or less 0.16 while the impact of age on the efficiency of a farmer of 60 years old is more or less 0.14 (see figure 5.4).

The impact of solvency is also indicated in figure 5.4. On a firm only financed with debts (solvency equals 0), the impact of age on efficiency is increasing for farmers younger than 42 years. On the other hand, the impact of age on

[14]In this extended stochastic production model, we do not use all farm characteristics because we only focus here on the impact of age and the link with other manager aspects.

efficiency is decreasing for farmers older than 42 years[15]. Higher solvency rates results in lower critical age levels.

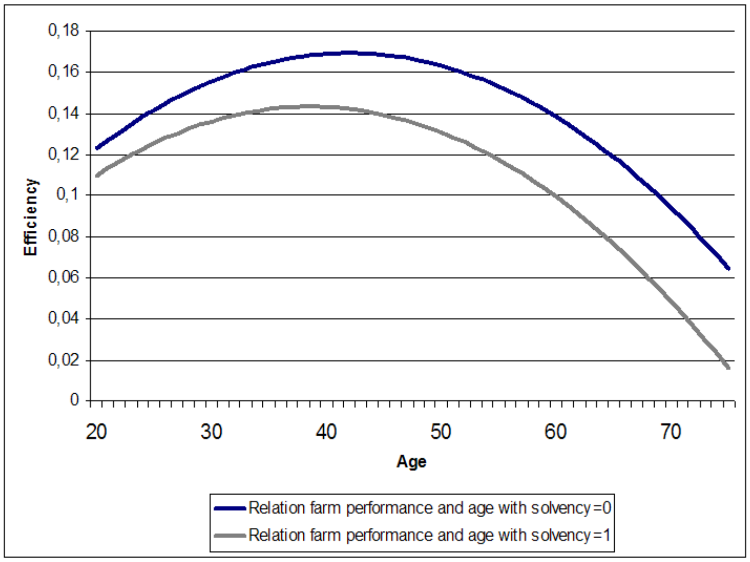

Figure 5.4: The relation between firm-efficiency and farmers age for different solvency levels

Education has a similar impact on the relation between age and efficiency as solvency. This is shown in figure 5.5. Lower education results in lower critical age levels. The critical age of farmers with the highest education (diploma1) is 42 while the critical age of farmers with the lowest education (diploma5) is 39[16].

The aim of this section was to analyze the impact of age on efficiency in more detail and to analyze the link between solvency and age and between education and age. We found that farm efficiency increases with the age of the decision maker before he/she reaches a certain critical age. This increase can be explained as a rise in experience. At a certain age, the impact of age on efficiency will decrease. Furthermore we found that a high solvency rate and a low education level will decrease this critical age level.

[15]With a education level of 1, in other words the highest level of education (diploma1)
[16]With a solvency rate of zero

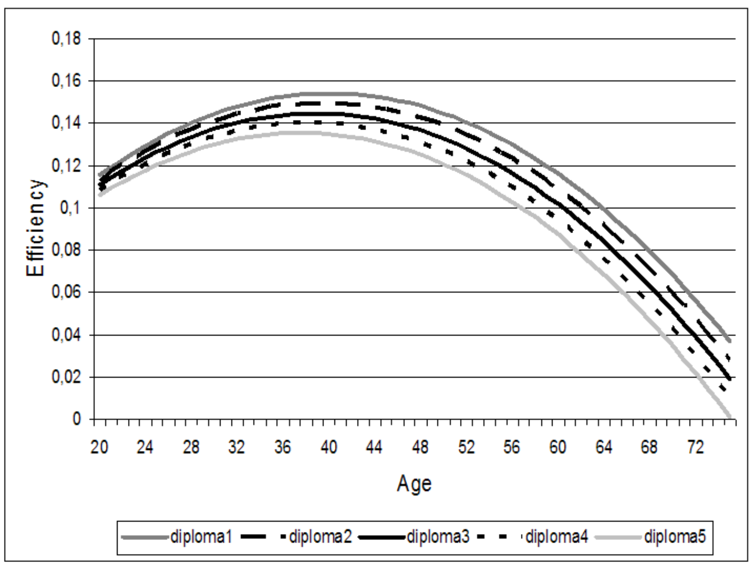

Figure 5.5: The relation between firm-efficiency and farmers age for different education levels

5.6 Conclusion and discussion

In this chapter, the continuous structural change in agriculture has been used as point of departure and motivation to measure farm efficiency. Efficiency is defined here in the sense Farrell (1957) introduced it: the relative position of the actual production with respect to the efficient, frontier production function. Measuring farm efficiency is thus seen as a consistent way of monitoring farm performance in a changing environment. Not only the performance measurement itself, but also the understanding why farms differ in their relative efficiency can bring new insight in the process of structural change and feedback to the concerned policies and government interventions.

Many managerial and structural characteristics are linked to farm performance. Stochastic frontier analysis of a representative Flemish farm data panel revealed that farm size, farm accounting and having a high share of own land have a positive effect on efficiency. On the other hand farm solvency, farmer's age and dependency on support payments are found to be negatively related to farm efficiency.

The link between managerial and structural characteristics and efficiency can be discussed in terms of capacities and incentives. Intrinsic capacities are e.g., education, size, age and the presence of a successor on the farm. The effect of schooling is clear, and empirically confirmed by the SFA on the Flemish farms. As also learning-by-doing enhances experiences, and thus capacities to use inputs better, the effect of age should be efficiency increasing. On the other hand, age of the farmer, who is not only manager, but also, and mostly, the main provider of labor, can become negatively related to efficiency when negative effects outweigh the positive. The empirical analysis showed an overall negative relationship, but a non-linear relationship would perhaps better differentiate between the predominantly positive and negative phase. This chapter showed that age, solvency and subsidies are slow-down factors of efficiency. This is appropriate as a first indicator, but in order to unravel the mechanisms through which efficiency and structural change can be steered, this is not sufficient. To study the interaction of age, education and solvency and their impact on farm performance in more detail is also an interesting topic. Analyzing the non-linear relationship we found that in general the efficiency of a farmer will increase till a certain age, afterwards the impact of age on efficiency will decrease. Lower solvency rates and higher education will increase this critical level.

Succession has a positive impact on efficiency. In many cases, the successor works together with his parents on the farm, and the combination of young labor forces, familiarity with new technologies and learning-by-doing experiences may increase capacities, but also provide extra incentives to perform better. The prospect that the farm business will continue, incite to adopt new technologies, renewed investments, or simply to get the best from the ongoing practices. The negative link between solvency and efficiency has probably also to be interpreted in this way. Low solvency can mean lot of new investments, and thus increase the capacities linked to the use of new technologies, but also it can mean an extra whip for getting better results that allow for repaying the debts. Age, combined with the prospect of succession, may also reflect the existences of incentives. Becoming older in agriculture may mean becoming satisfied with the earned income and rather wanting to slow down activities. Having a successor, however, provides new incentives to keep the farm highly performing.

What does this means for structural change? Specialized pig farms are found to be the most efficient farm type. This is not surprising, given the scarcity of space in Flanders. High productivity per unit of space becomes, however, also the main threat. High productivity also means a high level of by-products. Internalization of environmental effects means extra inputs that are needed for the same outputs. Size of the farm is another positive factor of efficiency. Here, the problem of conditionality, however, arises. A larger size guarantees more efficiency, but on the other hand farm growth may be facilitated when the farm performs better. Structural change in agriculture highly depends on the farm life cycle. The fact that a non negligible part of the farms have an older farmer without successor slows down the efficiency improvement of the sector.

Finally, what may be the role of the authorities? Policies to improve taking-over of non efficient farms by efficient farms will improve the performance of the overall agricultural sector. Based on the results of our study, it may be crucial that farms without successor stay not too long in business, before their production factors can be taken over by other, more efficient farmers. However, other outcomes of the study suggest a negative correlation of efficiency with subsidies on investments (in which first instalment is comprised). The more a farm depends on support payments, the lower its efficiency in using its resources. The reason for this result is not clear. Very plausible would be that they give wrong incentives, e.g., for doing sub-optimal investments, or getting stuck in business that is not profitable anymore. The results may suggest that agricultural policy is encouraging a less competitive agricultural sector by providing grants. On the other hand, there could be environmental and social benefits (e.g., survival) arising from those policy measures. Also the problem of conditionality can arise. It is also possible that farmers with a low efficiency level receive more support payments which can be the objective of the policy makers.

Finally, the methodology employed in this research to estimate and calculate efficiency and productivity does not take into account the environmental costs associated with the use of agricultural inputs or the environmental benefits associated with the production of agricultural goods. Integrating the environmental considerations into the calculation of agricultural performance is an important and challenging topic (see chapter 7).

Chapter 6

Linking farm efficiency with farm growth

[1]Parts of this chapter have been published as Van Passel, S., Van Huylenbroeck, G., Mathijs, E., 2006, Linking farm efficiency with farm growth, In: Causes and Impacts of Agricultural Structures edited by Mann, S., pp 23–41

There is nothing wrong with change, if it is in the right direction
—Winston Churchill

Abstract

Rapid technological change and changed consumer demands have led to the creation of an industrialized food and agricultural system. At farm level the agricultural sector continues to be subject of structural change with important consequences for productivity and efficiency of farming, equity within agriculture, the demand for government services and infrastructure, and the well-being of rural communities. The structural change of the agricultural sector can also influence policy which on its turn may influence agricultural structures. This highlights the importance of investigating determinants of agricultural structures and structural change. Theoretically, efficient farms grow and survive; inefficient ones decline and fail. This chapter investigates the link between structural change and farm performance. Using a large data sample of Flemish farms, the interplay between farm growth and farm efficiency is analyzed. We found that in general more efficient farms grow in size and less efficient farms decline in size. Analyzing determinants of farm growth, we observe heterogeneity between agricultural subsectors. The existence and persistence of differences in efficiency among farms and agricultural subsectors is an important explanation of structural change in agriculture.

6.1 Introduction

Since the beginning of the twentieth century, the spread of mechanization, increased land productivity due to technical change and rural migrations towards cities have transformed the system of agricultural production around the world (Chavas, 2001). Agricultural structures have been shaped by a variety of factors including economic, cultural, historical, political, technological and geographical conditions (Happe, 2004). The European agricultural production is undergoing major structural changes. Labor moves out of the sector and the average farm size is increasing. Slow growth in food demand and the effects of technical change on supply are likely to exert downward pressure on agricultural prices (Blandford and Hill, 2005). Generally, the agricultural sector is becoming more capital intensive and the share of external capital is low but increasing. In Europe, still a lot of different farm types remain and many farms have mixed activities, although farms are evolving towards more product specialization and are using their land more intensively (e.g., pig farming and horticulture). Structural change can also influence policy decisions which on their turn may influence agricultural structures. For example, one of the pressures for reform of the Common Agricultural Policy in the EU in 1992 was the high-input/high-output farming in the EU (Howarth, 2000).

Hence, understanding structural change of the agricultural sector is an important issue, because of the implications for the performance of farming, agricultural output and resource use. The viability of European agriculture crucially depends on its ability to adapt to structural change. More in general, the growth and relative performance of firms is crucial to the success of any economy (Konings, 1997).

Traditionally, studies on the growth rates of farms test the law of Gibrat (Gibrat, 1931). This law states that growth is determined by random factors and is independent of the initial farm size. Based on Jovanovic (1982) different papers add economic factors to this elementary stochastic model by emphasizing the importance of experience, human capital and other individual characteristics of the firm (e.g., Evans (1987*a*), Weiss (1999), Rizov and Mathijs (2003)). In general, an important explanation about structural change is the difference in performance. Theoretically, inefficient farms decline and fail and efficient farms grow and survive. In the long run firms differ in size not because of the fixity of capital, but because some are more efficient than others.

This chapter investigates the link between structural change and farm performance. Investigating structural change and the determinants of structural change is not easy to accomplish. Therefore, we restrict our research to analyze the link between farm growth and farm efficiency. Farm growth is only one aspect of structural change but a very important one. Farm efficiency is an important indicator to measure farm performance. The linkages between farm efficiency, farm growth and managerial and structural farm characteristics are complex and strongly interrelated as can be seen in figure 6.1. Using a large data set of Flemish farms, their growth and efficiency is calculated and the impact of efficiency on growth is analyzed. The evolution of market structure is a complex phenomenon and the quest for any single model that encompasses all the statistical regularities observed is not an appropriate goal (Sutton, 1997). Therefore the aim of this study is to study the interplay between efficiency and growth and not to test if Gibrat's law holds amongst Flemish farms. We will test the hypothesis that differences in farm efficiency among farmers and agricultural subsectors can explain (some) structural change in agriculture (indicated by the white arrow in figure 6.1). The impact of the structural and managerial characteristics on farm efficiency was analyzed in chapter 5 (indicated by the grey arrows in figure 6.1). This chapter is a logical continuation of the previous chapter. The economic analysis in these chapters 5 and 6 can serve as a detailed contextual analysis, which can help to assess farm sustainability by integrating environmental resource use in the economic analysis (see chapters 7 and 8).

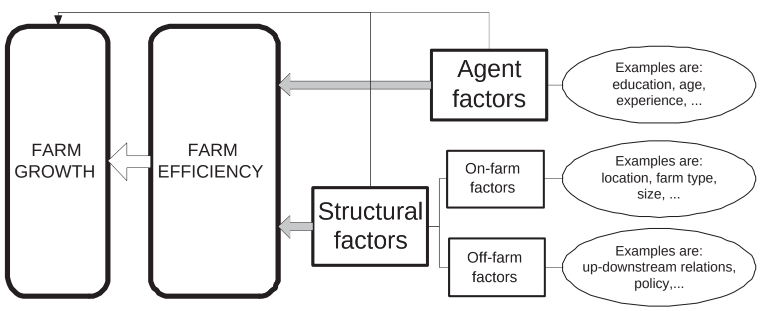

Figure 6.1: The link between farm growth, farm efficiency and structural and managerial characteristics

Starting from the passive learning model of Jovanovic (1982) the impact of several farm characteristics on farm growth can be investigated. But the hypothesis that in general more efficient farms grow in size and less efficient decline in size has not been empirically tested in previous work. Improved techniques to measure efficiency such as stochastic frontier analysis and data envelopment analysis are nowadays more easily to apply and are used more frequently. Therefore, this chapter analyzes directly the impact of farm efficiency on farm growth.

6.2 The concept of structural change and growth

Structural change can be studied in various ways. A lot of different issues can be analyzed, for example consider farm exit and entry[2] and farm growth and decline. This research only focuses on farm growth and decline. Before presenting the empirical model and results, we give a short overview of the concept of structural change. Furthermore, theories of firm growth will be described. Finally, we will also give a short review of empirical farm growth research of the agricultural sector.

Economics Glossary defines a structural change as a change in the parameters of a structure generating time series (Economic Glossary, n.d.). The vagueness of this definition reflects the different ways of looking at structural change. Structural change can be portrayed in various ways. Traditional characteristics are (i) the composition of output, (ii) input characteristics, (iii) types of farming processes and (iv) marketing channels employed. Blandford and Hill (2005) state that changes in these characteristics may have important economic, environmental and social implications but they provide limited insight into sustainability of agriculture over the longer term. Therefore the focus must be on the units in which production takes place: farms. Farms bring together labor, land and capital combined with other inputs to produce agricultural goods and services.

Several patterns of structural change are common in European agriculture (Blandford and Hill, 2005): (i) exit of agricultural labor, (ii) changes in numbers of holdings and their average size, (iii) the family nature of farming, (iv) farming combined with other activities, (v) tenure and family ownership of land and (vi) personal characteristics.

Over the past century, productivity has been a major force behind changes in agricultural output (Ahearn et al., 2002). In situations of rapid economic growth, significant labor migrations have reduced the proportion of the active labor force employed in agriculture. The migration from rural areas toward urban jobs has induced a sharp increase in farm labor productivity due in large part to mechanization. By contrast, the migration of labor can be caused by the increase of labor productivity[3]. Taking place over several decades, this process has transformed farming into a sector employing only a small proportion of the active population. In developed countries, such changes have induced a trend toward mechanization and significant increases in farm size. Further, the product mix produced by farmers has also changed. In general, farms have

[2] With farm exit and entry, we refer to farms that stop or start with farm activities. Note that farm exit does not mean that these farms go bankrupt, for example farmers can retire

[3] It's often difficult to distinguish causes and consequences in structural change processes

evolved toward more product specialization. Furthermore, agricultural sectors around the world are increasingly relying on trade and market mechanisms as a means of guiding resource allocation in agriculture. At the farm level, economic survival pushes managers toward implementing efficient production systems adapted to local conditions, toward developing marketing skills that can take advantage of market opportunities, and toward risk management strategies that can effectively deal with weather risk and changing market conditions (Chavas, 2001).

There are several views and models about the forces driving structural change in the agricultural production sector. Examples are *the technological treadmill* (Cochrane, 1958), *the overproduction trap* (Johnson and Quance, 1972), *the 'preserving the inefficient' mechanism* (Robinson, 1975) and *the life cycle hypothesis* (Lin et al., 1980). A good overview of the different theoretical views explaining structural change can be found in Harrington and Reinsel (1995). Each explanation or model can be combined into a more comprehensive synthesis. Harrington and Reinsel (1995, p. 12) put this as follows:

> The different explanations are like the blind men who examined the elephant and perceived very different animals. The different models must be combined to present a picture of that elephant.

Finally, it is important to note that agricultural structures are not static. Structural change can be characterized as an evolutionary process of constant adjustment to changes in demand, supply, and technological progress (Happe, 2004).

Reviewing the existing theories of firm growth[4] is a useful first step towards understanding the survival and growth of farm enterprises. Analyzing firm growth and survival, neoclassical economics suggests that one's attention must focus on the factors that have an impact on supply and demand for the product produced by the enterprizes (Rizov and Mathijs, 2003). Stochastic models extend this static framework by making it more dynamic. Such models use *the law of Gibrat*. This law states that growth is a stochastic process determined by random factors and in particular that growth is independent of initial farm size. Therefore, this law is called the Gibrat's law of Proportional Effects (Gibrat, 1931, Sutton, 1997). Many early empirical studies of the mobility of firms were shaped by this purely statistical model (Caves, 1998). For example, Simon and Bonini (1958) and Ijiri and Simon (1964) analyzed the size distribution of firms using a stochastic model of firm growth.

In the 1980s, starting from Gibrat's law a new direction in the literature appeared. The probabilities of survival of a firm, conditional on its age, size,

[4] A interesting review of the wide and extensive literature investigating the growth of firms can be found in Sutton (1997)

and other characteristics were analyzed. Also the firm's growth rate, conditional on survival and its dependency on age, size and other characteristics were studied. A theoretical model of firm growth to explain these deviations from the proportional growth law is the *passive learning model* of Jovanovic (1982)[5]. In this model, a sequence of firms enters the market. Each firm has some level of *efficiency* but it does not know its relative efficiency prior to entering. Firms learn about their efficiency as they operate in the industry. The efficient grow and survive; the inefficient decline and fall (Jovanovic, 1982). Jovanovic's model stimulated both theoretical and applied economic research (Lundvall and Battese, 2000). A disadvantage of Jovanovic's model is the immutability of the efficiency parameter (Rizov and Mathijs, 2003). Managers are born with an efficiency level and while they learn what that level is over time, they cannot alter it. In essence, this restriction assumes away one of the most dynamic processes taking place within industries, namely *learning-by-doing* (Malerba, 1992, Lundvall and Battese, 2000).

Starting from Jovanovic (1982), different empirical studies add economic factors to the elementary stochastic model by emphasizing the importance of experience, human capital and other individual characteristics of the firm. Firm growth is found to decrease with firm age and firm size (Evans, 1987*a*,*b*). Dunne et al. (1988) have investigated the relative importance of different types of entrants, the correlation of entry and exit patterns across industries over time, and the entrant's post-entry size and exit patterns. After controlling for life-cycle, size and product market effects, Konings (1997) found that de novo private firms outperform privatized and state-owned ones in transition countries. Lotti et al. (2003) investigates new small farms in the early stage of their life cycle. They found that Gibrat's law fails to hold in the years immediately following the start-up, when small farmers have to rush in order to achieve a size large enough to enhance their likelihood of survival. Gibrat's law should not be considered as a representation of overall industrial dynamics, but rather as a way to describe the growth behavior of mature, large and well-established firms (Lotti et al., 2004). Other examples of empirical studies for manufacturing industries which analyses the contribution of firm characteristics to firm growth are Variyam and Kraybill (1994), Nafziger and Terrell (1996) and Goedhuys and Sleuwaegen (2000). For the insurance industry Hardwick and Adams (2002) constructed a growth model with several firm specific factors such as input costs, profitability, output mix, company type, organizational form and location.

A large number of empirical studies have analyzed firm growth for manufacturing industries, but less has been done for the case of agriculture (Rizov and Mathijs, 2003). Empirical work in the agricultural sector that goes beyond testing Gibrat's law by focusing on the farmer and his character-

[5] There are several models that describe firm dynamics. An example of another model is the model of active exploration developed by Ericson and Pakes (1995)

istics as a key determinant of farm growth is rare (Weiss, 1999).

Several studies tested the law of Gibrat on agricultural firms (e.g., Shapiro et al. (1987), Hallam (1993)[6], Upton and Haworth (1987), Weiss (1999), McErlean et al. (2004), Kostov et al. (2005)). But besides size aspects also managerial farm characteristics are important reasons why farm size changes over time. Sumner and Leiby (1987) argue that human capital affects farm size and growth. This is confirmed by several empirical applications testing the impact of managerial ability as well as life-cycle patterns on farm growth (e.g., Upton and Haworth (1987), Weiss (1999), McErlean et al. (2004)). Also Rizov and Mathijs (2003) found that farm growth decreases with farm age and that learning considerations are important. On the other hand, Bremmer et al. (2002) found no indications of the influence of the life cycle on firm growth.

Other important determinants of farm growth and survival are the financial structure of farms (Shepard and Collins, 1982, Bremmer et al., 2002), farm income (Shepard and Collins, 1982), management returns (Garcia et al., 1987), management intensity (Garcia et al., 1987), profitability (Shepard and Collins, 1982, McErlean et al., 2004), productivity (Shepard and Collins, 1982), the off-farm employment status (Upton and Haworth, 1987, Weiss, 1999), schooling and sex of the farm operator (Weiss, 1999), the degree of mechanization (Bremmer et al., 2002), family labor (Bremmer et al., 2002), market and industry characteristics (Rizov and Mathijs, 2003), participation in government programs (Garcia et al., 1987) and government intervention (Ahearn et al., 2002).

[6]Hallam (1993) provides an overview of empirical tests of Gibrat's law for the US and Canadian farm sector.

6.3 Empirical model

This section describes first the measurement of farm growth. The empirical model and the different econometric problems are explained. Next, the measurement of farm efficiency using the stochastic frontier analysis is described.

6.3.1 Measuring farm growth

In order to measure farm growth, farm size must be compared between two specific points in time. The definition of farm size is a fundamentally important issue (McErlean et al., 2004, Kostov et al., 2005). Several different measures of farm size are available: size unit[7], number of livestock units and acreages under cultivation, total capital value, net worth, gross sales and net income. Output value measures (e.g., gross farm sales) and input value measures (e.g., net worth) may be unsatisfactory due to the impact of inflation and changes in relative prices (Weiss, 1998, McErlean et al., 2004). A disadvantage of physical input measures (e.g., acreage farmed) is the fact that farms are characterized by a non-linear production technology and changes in farm size involve changes in factor proportions and changes in production technology (Weiss, 1998, 1999). Hence, since changes in farm size generally imply not only changes in factor proportions and production technology, but also changes in the output mix of multiproduct farms, none of these measures can fully characterize the process of farm growth (Weiss, 1999). Measurement of the size of farms is not straightforward when different production types are involved (Garcia et al., 1987). Using acreages under cultivation or the number of livestock will make a comparison between a crop-farm and a pig-farm impossible. Therefore, using standard size units based on the economic size can provide an acceptable solution (Lepoutre et al., 2004).

Gibrat's law states that firm growth is determined by random factors, independent of size. To test Gibrat's law the following equation can be used:

$$LnS_{i,t} - LnS_{i,t-1} = \alpha + \beta LnS_{i,t-1} + u_{i,t} \tag{6.1}$$

$S_{i,t}$ is the size of farm i at time t. The case where β equals zero is known as *Gibrat's law of Proportional Effects*. If $\beta < 0$, small farms tend to grow faster than larger farms, i.e. the effects of randomness are offset by negative correlation between growth and size. If $\beta > 0$, larger farms tend to grow faster than smaller farms.

[7] The size unit is calculated for all farms in the FADN-dataset based on the standard gross margin (CAE, 2000, FADN, n.d.)

Equation 6.1 can be generalized by adding farmer associated characteristics and farm specific variables $X_{i,t}$:

$$LnS_{i,t} - LnS_{i,t-1} = \alpha + \beta LnS_{i,t-1} + \sum \gamma X_{i,t} + u_{i,t} \tag{6.2}$$

Equation 6.2 can be estimated using balanced panel data via an effect model (random or fixed) or via ordinary least squares (OLS) regression. To identify the most appropriate estimation the Breusch and Pagan's Lagrange multiplier statistic and the Hausman's chi-squared statistic can be used.

There are several problems with using a linear regression as in equation 6.2. The first is the assumed linear effect of the additional explanatory variables $X_{i,t}$ (Kostov et al., 2005). Weiss (1999) used an applied non-linear functional form. However, specifying an *ad hoc* non-linear functional form is not a viable strategy, since it may influence the final results in an unpredictable way (Kostov et al., 2005).

Secondly, even in the simple model (equation 6.1), there is an underlying assumption that Gibrat's law holds (or is violated) globally. If we want to test whether Gibrat's law holds for some farms and not for others, the linear regression is too restrictive (Kostov et al., 2005). Such a test can nevertheless be designed using quantile regression methods as in Kostov et al. (2005).

Thirdly, measures of farm growth are only meaningful for surviving farms. Analyzing the determinants of farm survival is not only interesting in its own right but is also important for obtaining unbiased estimates in growth models (Weiss, 1999). Growth rates estimated on survivors only will be biased toward finding relatively lower growth rates for the larger holdings. This problem of sample selection bias can result in incorrect rejection of Gibrat's law, giving the impression that smaller farms tend to grow faster than larger farms (Shapiro et al., 1987, Sutton, 1997, Weiss, 1999, Lotti et al., 2003, Kostov et al., 2005). A way of solving this problem is using a two-step procedure, developed by Heckman (1979). In the first step a survival model is estimated. This is a probit (or logit) equation on the probability of farm survival from the complete sample. This equation is used to obtain an additional variable, where the values represent the inverse Mill's ratio for each observation. This additional variable is used in step two as a correcting factor into the least squares regression based upon a sample with only the survival farms.

Fourthly, Gibrat's law requires that the growth rates must be serial uncorrelated (Shapiro et al., 1987). Serial correlation can be tested in the following way:

$$LnS_{i,t} - LnS_{i,t-1} = \varphi + \varepsilon[LnS_{i,t-1} - LnS_{i,t-2}] \tag{6.3}$$

The null hypothesis is that $\varepsilon = 0$, which indicates that growth rates are unrelated over time (Shapiro et al., 1987). If $\varepsilon < 0$ then good fortune reverses itself. This problem can be corrected by estimating (Chesher, 1979):

$$LnS_{i,t} - LnS_{i,t-1} = \alpha + \beta LnS_{i,t-1} + \delta LnS_{i,t-2} + u_{i,t} \tag{6.4}$$

Fifthly, the problem of endogeneity can occur. This means that an explanatory variable in a multiple regression model is correlated with the error term, either because of an omitted variable, measurement error or simultaneity.

Finally, the variance of growth rates must be the same for all firms (homoscedastic with respect to size). We can test for heteroscedasticity by using Glesjer's method (Glesjer, 1969):

$$|u_{i,t}| = \varrho + \psi LnS_{i,t-1} \tag{6.5}$$

Generalized heteroscedasticity exists if $\varrho, \psi \neq 0$. $\psi < 0$ indicates that the variability of the growth rates declines with size, which indicates that small farms are more unstable. If heteroscedasticity is detected, a possible respons is to use heteroscedasticity robust statistics after estimation by OLS. A feasible GLS procedure can be used to correct for heteroscedasticity. In this way, weighted least squares estimators are calculated to develop heteroscedasticity-robust statistics (Wooldridge, 2000).

To analyze the sensitivity of the results, the data sample can be divided into subsamples of different farm types. There may be a poor degree of homogeneity between farms of different types and therefore pooling of different farm types may be appropriate (McErlean et al., 2004).

6.3.2 Measuring technical efficiency

Efficiency can be defined as the actual productivity of a firm to his maximum productivity (Farrell, 1957). We will use the stochastic frontier analysis to estimate efficiency predictions for each observation. A brief description and overview of the methodology used can be found in chapter 5. The use of stochastic frontier analysis implies the choice of the functional form. Testing the translog functional form against the Cobb-Douglas functional form, the translog functional form is preferred[8].

[8]The hypothesis that the coefficients of the second order term in equation 6.6 were 0, was rejected

The basic specification for the stochastic frontier model can be written as:

$$\begin{aligned}
Ln(Output_{it}) = \alpha &+ \beta_1 ln(Lab_{it}) && +\beta_2 ln(Cap_{it}) \\
&+\beta_3 ln(Area_{it}) && +\beta_4 ln(Med_{it}) \\
&+\tfrac{1}{2}\beta_6[ln(Lab_{it})]^2 && +\tfrac{1}{2}\beta_7[ln(Cap_{it})]^2 \\
&+\tfrac{1}{2}\beta_8[ln(Area_{it})]^2 && +\tfrac{1}{2}\beta_9[ln(Med_{it})]^2 \\
&+\beta_{11} ln(Lab_{it}) * ln(Cap_{it}) && +\beta_{12} ln(Lab_{it}) * ln(Area_{it}) \\
&+\beta_{15} ln(Cap_{it}) * ln(Area_{it}) && +\beta_{16} ln(Cap_{it}) * ln(Med_{it}) \\
&+\beta_{13} ln(Lab_{it}) * ln(Med_{it}) && +\beta_{18} ln(Area_{it}) * ln(Med_{it}) \\
&+\textstyle\sum_{j=1}^{n=12} \delta_j * YearDummy_j && +\textstyle\sum_{j=1}^{n=4} \delta_j * SectorDummy_j \\
&+v_{it} && -u_i
\end{aligned} \tag{6.6}$$

Where the dependent variable $Output_{it}$ is the deflated total yield (gross output) in Euros of the farm i in year t. The inputs used in the production proces are (i) Lab_{it} (the total amount of labor of the farm i in year t); (ii) Cap_{it} (the total amount of deflated farm capital[9] in Euros of the farm i in year t), (iii) $Area_{it}$ (the total amount of utilized agricultural area in hectares of the farm i in year t), (iv) Med_{it} (the total amount of intermediate consumption in Euros of the farm i in year t), (v) $YearDummy$ refers to year dummies from 1989-2002 and (vi) $SectorDummy$ refers to the different agricultural subsectors[10]. The efficiency of the i^{th} firm equals the ratio of the observed output for the i^{th} firm, relative to the potential output. The potential output is defined by the production function. The technical efficiency (TE) is calculated as:

$$TE_i = \frac{y_i}{exp(x_i\beta)} = \frac{exp(x_i\beta - u_i)}{exp(x_i\beta)} = exp(-u_i) \tag{6.7}$$

Where y_i is the dependent variable (= the output), x_i are the independent variables (the inputs) and u_i is the non-negative error term. As in chapter 5 the random effects formulation is preferred, but the panel data formulation with time-invariant efficiency[11] cannot be used. This is because efficiency estimates for each observation in each year are needed. Hence, instead of using a time trend[12], we use year dummies in equation 6.6.

[9]The total capital equals the total amount of assets, farm capital is calculated as total capital minus land capital, in this way overlap is avoided.

[10]The different agricultural subsectors are (i) specialist field crops, (ii) specialist grazing livestock (=dairy), (iii) specialist pigs and (iv) mixed farms. Types of farms are defined in terms of the relative importance of the different activities on the farm using the standard gross margins. (FADN, n.d.)

[11]An advantage of using the random effects panel data formulation with time-invariant inefficiency is that the firm specific technical efficiencies can be estimated more consistently. But the benefits of panel data come at the expense of another strong assumption that the firm efficiency does not vary over time (Cornwell and Schmidt, 1996)

[12]indicating technology change

6.4 Empirical model: results

6.4.1 Data

This study uses farm accountancy panel data from a group of 304 Farms in Flanders. The Flemish FADN[13]-data are collected and managed by the Centre of Agricultural Economics (CAE)[14]. Information of 304 farms for a period of 14 years (1989-2002) is available. In total the sample contains 4256 observations of 304 different farms. Table 6.1 shows some descriptive statistics of the variables.

Table 6.1: Descriptive statistics

	Minimum	Maximum	Mean	Std. Dev.
Size unit[a]	3	125	20	10
Gross output (Euro)	22163	1499698	205947	150944
Agricultural area (ha)	0	265	32	21
Labor (in man − equivalent units)	0.350	6.350	1.587	0.450
Farm capital (Euro)	14806	1188975	281466	165725
Intermediate consumption (Euro)	6238	853650	114012	94246
Age of farm manager	20	72	43	10
Solvency[b]	0.000	1.000	0.444	0.299
Subsidies interest[c] (Euro)	0	24822	2678	3021
Subsidies revenues[d] (Euro)	0	29400	396	1279
Subsidies income[e] (Euro)	0	90511	4479	5822
Share land in property[f]	0.000	1.000	0.270	0.262
Technical efficiency[g]	0.622	0.976	0.900	0.035

[a] based on the standard gross margin (FADN, n.d.)
[b] measured as own capital divided by total capital
[c] subsidies on investments (interest support)
[d] subsidies on animal products (subsidies on sale and purchase of animals are not included)
[e] direct payments to producers: suckler cow premium, slaughter premium, set-aside premium, arable crops hectare aid,...
[f] the amount of land in property over the total amount of utilized farm land
[g] measured as explained in section 6.3.2.

6.4.2 Farm efficiency: empirical results

The results of the estimation of the translog stochastic production frontier using stochastic frontier analysis, as explained in section 6.3.2, can be found in table 6.2. Calculating the technical efficiency as in equation 6.7, we have for each observation in our data sample a measure for farm performance. Over the

[13] Farm Accountancy Data Network
[14] The Belgian FADN-data were collected by the former CAE. Since 2006, the CAE is assimilated into the Institute for Agricultural and Fisheries Research. The Flemish FADN-data are now collected by the agricultural monitoring and study service of the Flemish Ministry for Agriculture.

period 1989-2000, we have an average farm efficiency of 89%. The lowest farm efficiency is 59% and the highest is 98%.

Table 6.2: Estimation coefficients of the translog stochastic production function

Variable	Coefficient	st. error	Variable	Coefficient	st. error
Constant	1.8084 ***	0.1172	D-1990	0.0247 *	0.0789
$Ln(Lab_{it})$	0.2831 ***	0.1003	D-1991	0.0449 ***	0.0010
$Ln(Cap_{it})$	0.0547	0.0594	D-1992	-0.0016	0.9097
$Ln(Med_{it})$	0.3840 ***	0.0520	D-1993	0.1241 ***	0.0000
$Ln(Area_{it})$	0.0244 *	0.0143	D-1994	0.1216 ***	0.0000
$[ln(Lab_{it})]^2$	0.0452	0.0280	D-1995	0.1066 ***	0.0000
$[ln(Cap_{it})]^2$	0.0535 ***	0.0093	D-1996	0.1092 ***	0.0000
$[ln(Med_{it})]^2$	0.1093 ***	0.0101	D-1997	0.0492 ***	0.0006
$[ln(Area_{it})]^2$	0.0052 ***	0.0005	D-1998	0.1368 ***	0.0000
$ln(Lab_{it}) * ln(Cap_{it})$	-0.0020	0.0270	D-1999	0.2746 ***	0.0000
$ln(Lab_{it}) * ln(Area_{it})$	-0.0014	0.0092	D-2000	0.1300 ***	0.0000
$ln(Lab_{it}) * ln(Med_{it})$	-0.0422 *	0.0235	D-2001	0.0415 ***	0.0039
$ln(Cap_{it}) * ln(Area_{it})$	0.0046 *	0.0028	Field crops	0.0567 ***	0.0000
$ln(Cap_{it}) * ln(Med_{it})$	-0.1131 ***	0.0155	Dairy	0.0468 ***	0.0000
$ln(Area_{it}) * ln(Med_{it})$	-0.0027	0.0031	Pigs	0.0674 ***	0.0000
D-1989	0.0545 ***	0.0146			
Number of observations	4256		Variances: $\sigma^2(v)$	0.01994	
Iterations completed	241		Variances: $\sigma^2(u)$	0.02102	
Log likelihood function	1568		Variances: σ^2	0.04096	

* significant at 10%, **significant at 5%, ***significant at 1%

6.4.3 Farm growth: empirical results

Structural change can be studied in different ways. Although entry and exit is quite important, we will only study expansion and contraction. We cannot study entry and exit of farms because of data constraints[15]. Calculating the changes in size, we observe the growth and decline of farms. Table 6.3 gives some descriptive statistics of (i) all observations of our data sample, (ii) the 50 observations with the highest growth in size and (iii) the 50 observations with the highest decline in size.

Analyzing the characteristics of the *high growth farms* and the *low growth farms*, several differences are observed. On average, high growth farms are larger in size and have younger and better educated farmers, while low growth farms are smaller and have older, less educated farmers. The solvency on high growth farms is lower, indicating that these farms have less own capital. Also the dependency on support payments differs. Low growth farms are more dependent on subsidies. Further, the specialist pig farms are overrepresented in the group of high growth farms. The overrepresentation of the specialist grazing livestock in the group of low growth farms is also remarkable. Concerning the link between efficiency and growth, we found that the efficiency of the 50 observations with the highest growth in size was 89.0% while the average farm efficiency of the 50 observations with the highest decline in size was 87.7%. The link between the farm growth in year t and the farm efficiency in year

[15] Our data set contains a lot of information but farms that enter and exit the data set do not necessarily enter or exit the agricultural sector

Table 6.3: Descriptive statistics of all observations and observations with the highest growth and decline in size (average values)

Variable	All observations (mean value)	50 observations with highest growth in size (mean value)	50 observations with highest decline in size (mean value)
Growth[a]	0.0100	0.1914	-0.1360
Size	20	27	14
Farmer's Age	43	39	46
Solvency[b]	0.4326	0.2862	0.4501
Subsidies Dependency[c]	0.0478	0.0342	0.0646
Share land[d]	0.2709	0.3047	0.3360
$Efficiency^{e}_{t-2}$	0.8918	0.8907	0.8771
Higher education (in%)	55	58	50
Successor on farm (in%)	14	14	18
No successor on farm (in%)	43	40	48
Doubt about succession (in%)	43	46	34
Specialist field crops (in%)	8	12	26
Spec. grazing livestock (dairy) (in%)	47	10	38
Specialist pigs (in%)	15	44	16
Mixed farms (in%)	30	34	20
Number of observations	3648[f]	50	50

[a] is measured as $ln(size_t) - ln(size_{t-1})$
[b] is measured as own capital divided by total capital
[c] calculated as a percentage of total revenues, indicating the dependency on support payments
[d] measured as land in property (in ha) divided by total utilized agricultural land (in ha)
[e] calculated as technical efficiency using stochastic frontier analysis
[f] to study growth, the data sample is reduced to a sample of 12 years (1991-2002), so we can link the farm efficiency from two years back to the farm growth

t-2 is also shown in figure 6.2 for all 304 farms during the period 1991-2002. As shown in figure 6.2 there is a lot of variation. Some farms which decline in farm size have high efficiency scores, so efficiency is certainly not the only determinant of farm growth. Many aspects can influence this analysis, such as policy measures or external events (e.g., food crises). The *Pearson* correlation between farm efficiency and farm growth is low but significant ($\rho = 0.069$). Also the rank correlation (*Spearman's rho*) is significantly different from zero ($\rho = 0.079$).

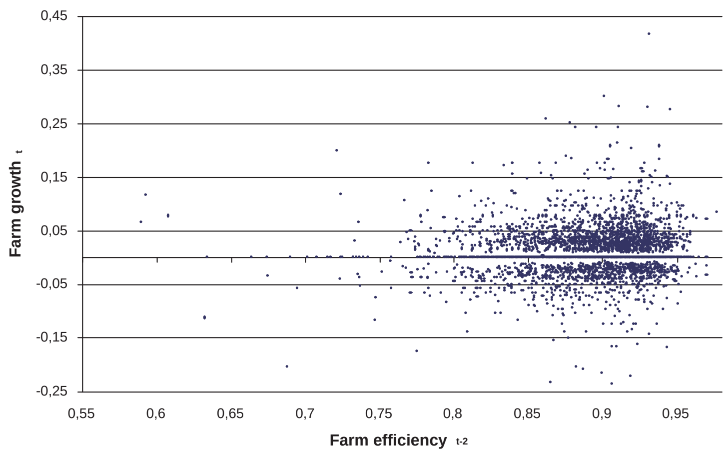

Figure 6.2: The link between farm efficiency and farm growth

6.4.4 Linking farm efficiency with farm growth: empirical results

To study the impact of farm characteristics on farm growth in more detail, an econometric model was estimated. Equation 6.2 can be rewritten as:

$$\begin{aligned} LnSize_{i,t} - LnSize_{i,t-1} = & \alpha + \beta_1 LnSize_{i,t-1} && +\beta_2 LnSize_{i,t-2} \\ & +\gamma_1 Farmer's\ Age_{i,t} && +\gamma_2 Farmer's\ Age^2_{i,t} \\ & +\gamma_3 Solvency_{i,t} && +\gamma_4 Sub\ Dependency_{i,t} \\ & +\gamma_5 Share\ Land_{i,t} && +\gamma_6 Efficiency_{i,t-\overline{1,2,3}} \\ & +u_{i,t} \end{aligned} \tag{6.8}$$

Using panel data, three different models were estimated: (i) a random effects model, (ii) a fixed effects model and (iii) an ordinary least squares without group dummy variables. A small *Pagan's Lagrange multiplier* statistic argues in favor of the classical OLS. Furthermore, serial correlation was found and therefore the variable $LnSize_{i,t-2}$ was added to correct this problem. Another possible problem is endogeneity. To measure farm performance, we opt for the farm efficiency of two years back. As measured in chapter 5 several managerial and structural farm characteristics have an impact on farm efficiency. Therefore, we use several proxy variables for unobserved explanatory variables such as age, size, solvency, and dependency on support payments. This plug-in solution to the omitted variables problem is not a perfect but a possible solution. An important note on the approach is the fact that the analysis is unconditional. Consider for example the solvency of a firm. We analyse the impact of the solvency on farm growth assuming that farm growth has no impact on solvency. Therefore, we use in our model the average farm efficiency of one, two and three years earlier ($t - \overline{1,2,3}$).

Next, the variance of the growth rates must be the same for all firms (homoscedastic with respect to size). Testing for heteroscedasticity by using Glesjer's method (Glesjer, 1969) we found heteroscedasticity. Therefore, we executed a feasible GLS procedure (FGLS) to correct for heteroscedasticity. An issue we could not take into account is the selection bias. The sample we used contained no information about farm survival, such that executing a two-step Heckman procedure was not possible. Therefore, the growth rate estimates will be biased. However the aim of this research is not to test if Gibrat's law hold, but to study the link between efficiency and growth.

Table 6.4 shows the results of the model estimation. To correct for heteroscedasticity, a feasible GLS procedure is executed (table 6.4). The results confirm the observations of the descriptive statistics of table 6.3. Age, solvency and dependency on support payments seems to decrease farm growth. Farms with a higher share of support payments have lower growth rates. Hence, dependency on support payments has a negative impact on farm growth[16]. Government payments have clearly impact on the structural change of the agricultural sector. The negative impact of support payments is not that surprising, Goetz and Bebertin (2001) found also that farmers quit at faster rates if they receive more government program payments.

Important in this analysis is the impact of the efficiency on growth. Farms with a high efficiency, have higher farm growth later on. The assumption of Jovanovic (1982) that firms learn about their efficiency as they operate seems to be correct. Generally, efficient farms grow, while the inefficient decline in size. Table 6.4 shows that an increase of 10% in efficiency will result in a 1.9% increase in farm growth.

Table 6.4: The expanded growth model

	OLS		FGLS	
Variable	coefficient	standard error	coefficient	standard error
Constant	-0.0120	0.0592	-0.1018	0.0687
$Log(Size)_{i,t-1}$	-0.0733 ***	0.0170	-0.0804 ***	0.0237
$Log(Size)_{i,t-2}$	0.0652 ***	0.0173	0.0762 ***	0.0243
$Farmer's\ Age_{i,t}$	-0.0014	0.0015	-0.0002	0.0014
$Farmer's\ Age^2_{i,t}$	0.0000	0.0000	0.0000	0.0000
$Solvency^a_{i,t}$	-0.0152 **	0.0078	-0.0194 **	0.0082
$Sub\ Dependency^b_{i,t}$	-0.2537 ***	0.0580	-0.2495 ***	0.0584
$Land\ Share^c_{i,t}$	0.0156 **	0.0080	0.0105	0.0084
$Efficiency^d_{i,t-\overline{1,2,3}}$	0.1332 ***	0.0550	0.1913 ***	0.0626

Dependant variable: $farm\ growth_{i,t}$ (measured as $ln(size_t) - ln(size_{t-1})$)
Number of observations: 3344
R-squared(FGLS): 0.033 and Adjusted R-squared(FGLS): 0.031
[a] measured as own capital divided by total capital
[b] calculated as a percentage of total revenues, indicating the dependency on support payments
[c] measured as land in property (in ha) divided by total utilized agricultural land (in ha)
[d] calculated as the average technical efficiency of one, two and three years earlier
* significant at 10%, **significant at 5%, ***significant at 1%

[16] Remind that this analysis is unconditional, meaning that we assume that farm growth has no impact on the dependency on support payments

6.4.5 Linking farm efficiency with farm growth: sectoral differences

Another important aspect in structural growth is the sectoral heterogeneity. Similar as in figure 6.2, the link between the farm growth in year t and the farm efficiency in year t-2 can be studied for different agricultural subsectors. Figure 6.3 shows the link for specialist dairy farms and pig farms. Observing figure 6.3 we see a relationship between efficiency and growth for specialist pig farms while this link is totally absent for dairy farms.

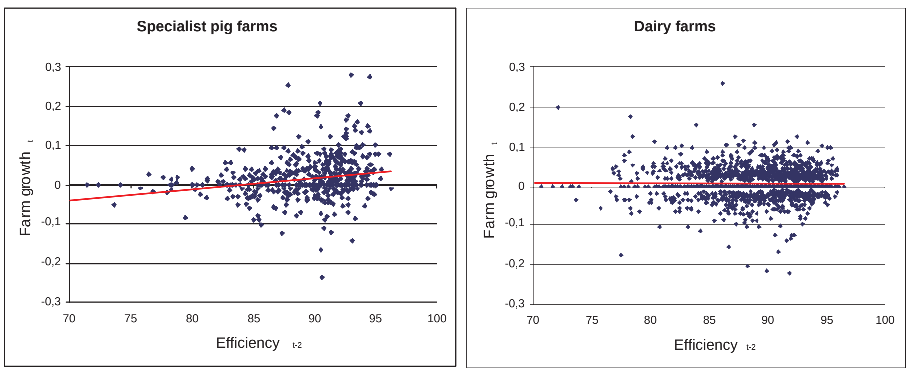

Figure 6.3: The link between farm efficiency and farm growth for specialist pig farms and dairy farms

To analyze the differences between the impact of efficiency on growth of different agricultural subsectors, the model could be estimated for each agricultural subsector separately. We observe three important subsectors: (i) dairy farms (grazing livestock), (ii) specialist pig farms and (iii) mixed farms. Table 6.5 shows the results of the estimations of equation 6.8 for the three different agricultural sub sectors[17].

Table 6.5 shows remarkable differences between the agricultural subsectors. The impact of efficiency on farm growth in the dairy sector is not significant. As seen in table 6.3 the dairy farms experience no high growth rates, especially in contrast to the pig farms. A possible explanation is the influence of policy measures (e.g., milk quota). The milk quota system in Flanders limited farm growth since its introduction in 1984. In the case of the dairy sector policy measures cloud the link between farm performance and farm growth. The impact of the dependency on support payments is significant on dairy farms. Higher dependency means lower farm growth. On the other hand, pig farms receive less European support payments and the impact of policy intervention is less influential. In general efficient pig farms will have higher growth rates

[17] Table 6.5 shows the results of the FGLS for the different subsectors

Table 6.5: The growth model for different agricultural sectors

	Dairy farms		Pig farms		Mixed farms	
Variable	coefficient	st. error	coefficient	st. error	coefficient	st. error
Constant	0.0945	0.0926	-0.5124 ***	0.1762	0.0302	0.1505
$Log(Size)_{i,t-1}$	-0.1430 ***	0.0322	-0.1109 *	0.0638	-0.0968 **	0.0455
$Log(Size)_{i,t-2}$	0.1374 ***	0.0327	0.0830	0.0654	0.0832 *	0.0457
$Farmer'sAge_{i,t}$	-0.0044 **	0.0023	-0.0029	0.0041	-0.0045	0.0027
$Farmer'sAge^2_{i,t}$	0.0000 *	0.0000	0.0000	0.0000	0.0000	0.0000
$Solvency^a_{i,t}$	-0.0123	0.0108	0.0113	0.0251	-0.0133	0.0133
$Sub\ Dependency^b_{i,t}$	-0.1465	0.1135	0.1772	0.2578	-0.1670 **	0.0924
$Land\ Share^c_{i,t}$	-0.0078	0.0142	0.0131	0.0192	-0.0084	0.0118
$Efficiency^d_{i,t-\overline{1,2,3}}$	0.0608	0.0793	0.7731 ***	0.1571	0.1938	0.1413
Number of observations	1558		513		993	
Adjusted R-square	0.028		0.033		0.032	

Dependant variable: $farm\ growth_{i,t}$ (measured as $ln(size_t) - ln(size_{t-1})$)
[a] measured as own capital divided by total capital
[b] calculated as a percentage of total revenues, indicating the dependency on support payments
[c] measured as land in property (in ha) divided by total utilized agricultural land (in ha)
[d] calculated as the average technical efficiency of one, two and three years earlier
* significant at 10%, **significant at 5%, ***significant at 1%

then inefficient pig farms. Hence, differences in efficiency exists between agricultural subsectors and these differences can (partly) explain structural change in agriculture. The link between efficiency and growth is weakened in strongly regulated agricultural subsectors (e.g., dairy farms).

6.5 Conclusion and discussion

The agricultural sector continues to be subject of structural change. This has important consequences for the productivity and efficiency of farming, equity within agriculture and the demand for government support, services and infrastructure. Policy makers can respond to structural changes by taking policy measures. For example providing zones and infrastructure for glasshouses. On the other hand, policy measures can be used to stimulate and favor certain changes for example support payments for organic agriculture. To support policy makers, it is important to investigate determinants of agricultural structures and structural change.

The objective of this chapter was to analyze the link between farm performance and structural change. In the proceeding chapter the impact of structural and managerial characteristics on farm efficiency was investigated. Knowing the performance of farms and understanding why farms differ in their relative efficiency can bring new insights in the process of structural change (see chapter 5). This chapter is a logical continuation of this research by analyzing the impact of farm performance on structural change. More specific, the link between farm growth and farm efficiency is studied.

In theory, good performing farms will grow in size and survive and bad performing farms will decline in size and fail. Jovanovic (1982) introduced a theoretical model of firm growth to explain deviations from the traditional proportional growth law. This law of Gibrat states that growth is a stochastic process determined by random factors and in particular that growth is independent of initial size (Gibrat, 1931). The passive learning model of Jovanovic (1982) states that firms learn about their efficiency as they operate. The efficient grow and survive while the inefficient decline and fail. Starting from Jovanovic (1982) several empirical studies add economic factors to the growth model by emphasizing the importance of experience, human capital and other managerial and structural characteristics. The contribution of this chapter is to investigate directly the impact of farm efficiency on farm growth.

Farm efficiency can be measured in a consistent way by using techniques such as the stochastic frontier analysis or non-parametric methods. These techniques are nowadays improved and have become more currently in use. Using the stochastic frontier method, efficiency estimates for each observation in our data sample are calculated. In this research data of 304 Flemish farms from 1989-2002 are used.

We found that differences in farm efficiency among farmers and agricultural subsectors can explain partly structural change in agriculture. Our empirical results show that farms with a high efficiency at the moment, will have higher growth in farm size two years later. These results confirm the hypothesis that in general efficient farms grow in size and inefficient ones decline in size. Furthermore, we found evidence of sectoral heterogeneity. Hence, differences in efficiency exists between agricultural subsectors and these differences can partly explain structural change in agriculture. Moreover, in strongly regulated agricultural subsectors (e.g., dairy farms) the impact of farm efficiency on farm growth is not significant. A possible explanation is the fact that policy measures (e.g., the milk quota) affect the link between farm efficiency and farm growth. Since 2000, nitrate policies started to have a more clear and pronounced impact on pig production in Flanders (Deuninck, 2006) than in the observed period (1989-2002), consequently we can expect that the impact of efficiency on growth has been decreased in this subsector. Furthermore, other aspects such as food crisis (e.g., dioxine crisis), disproportionate increase in the percentage of old farmers could trouble the relationship between growth and efficiency.

Our analysis does not show that farms have to grow to become more efficient but we showed that efficient farms have the tendency to grow. Policy makers aiming to increase the efficiency of farming, must take into account that their policy measures will result in structural changes.

Finally, this research investigates the impact of farm efficiency on farm growth and decline. An important extension of this research is the analysis of the impact of efficiency on farm survival. Hence, our research is a first indication, but to unravel the interlinking between farm performance and structural change more research is certainly necessary.

Chapter 7

Measuring farm sustainability and explaining differences in farm sustainability: Evidence from Flemish dairy farms

[1]Parts of this chapter have been published as (i) Van Passel, S., Nevens, F., Mathijs, E., Van Huylenbroeck, G., 2007, Measuring farm sustainability and explaining differences in sustainable efficiency, Ecological Economics 62: 149-161; and (ii) Van Passel, S., Mathijs, E., Van Huylenbroeck, G., 2006, Explaining differences in farm sustainability: Evidence from Flemish dairy farms, contributed paper prepared for presentation at the International Association of Agricultural Economists Conference, Gold Coast, Australia, August 12-18

In almost every case, environmental concerns have not been sufficiently integrated within economic sectors and decision-making, an essential postulate of sustainable development

— Sneddon et al. (2006)

Abstract

A major objective of European agricultural policy is to have a sustainable and efficient farming sector that is applying environmentally-friendly production methods. Policy makers aim to combine a strong economic performance and a sustainable use of natural resources. Therefore, it is important to measure and to assess farm sustainability. For a large dataset of Flemish dairy farms, a valuation method that is based on the concept of opportunity costs is used to calculate and analyze differences among the sample farms with respect to the creation of *sustainable value*. But more important than measuring the creation of sustainable value is to analyze differences in sustainability performance. Therefore, sustainability measures are calculated and differences in sustainability performance are explained. Using panel data, an effect model captures the determinants of sustainability performance of the studied farms. The empirical model shows that, in general, larger farms have a higher degree of sustainability. Also farmer's age and dependency on support payments proved to be determining characteristics for observed differences in sustainability performance. Furthermore, we observe a high sustainability performance on farms with higher levels of economic efficiency.

7.1 Introduction

One of the major objectives of European agricultural policy is to have a sustainable and efficient farming sector, which uses safe and environmental-friendly production methods and provides quality products that meet consumers' demands. Sustainability is seen as a key element towards a profitable long-term future for farming and rural areas. Policy makers aim to combine strong economic performance with the sustainable use of natural resources in the field of agriculture (European Commission, 2004, Fisher Boel, 2005). To achieve a competitive agriculture, farms have to apply conventional inputs as efficiently as possible. To create an environmental friendly agriculture farms have to deal efficiently with the natural resources (Reinhard, 1999).

An important conclusion of the United Nations Conference on Environment and Development in 1992 is that the major cause of the continued deterioration of the global environment is the unsustainable pattern of consumption and production (United Nations, 1992). While sustainable consumption targets consumers, sustainable production is related to companies or organisations that make products or offer services (Veleva and Ellenbecker, 2001). Despite the difficulty of defining sustainable production and the vagueness of several definitions, there is a clear consensus to move from definition attempts toward developing and using concrete tools for measuring and promoting actual sustainability achievements.

To meet with the challenges of sustainability, an approach for integrated assessment of companies is required to provide a good guidance for decision-making (Krajnc and Glavic, 2005*a*). The use and development of sustainability indicators can be an effective way to make the concept of agricultural sustainability operational (Rigby et al., 2001, van Calker et al., 2006). Public sector investment to increase farm performance requires accurate assessment of the efficiency of farmers and identification of the sources of inefficiencies in order to develop policy and institutional innovations to minimize inefficiencies (Sherlund et al., 2002). Therefore, it is important to measure and to assess farm sustainability. The aim of this chapter is to measure the sustainability of dairy farms in terms of *sustainable value* and *return-to-cost* and to find out why farms differ in sustainability performance. The Flemish dairy sector is used as test-case and example to identify farm sustainability. Using a large data set, the sustainable value creation of dairy farms and its evolution during 1995-2001 is analyzed. Further, the robustness of our results is tested by using different benchmarks for calculating the return-to-cost ratio. The existence of frontrunners and laggards among dairy farms is investigated by testing whether good/bad farm performance is repeated from year to year. Furthermore, possible causes of observed differences are studied using an empirical model. Finally, the link between partial productivity measures, eco-efficiency measures, traditional production economic efficiency measures (e.g., technical efficiency) and return-to-cost ratio is analyzed.

7.2 Theoretical framework

Pezzey and Toman (2002*a*) state that: *concern about sustainability is almost as old and enduring as the dismal science itself, though the word itself has come into fashion only in the past decades.* Since the publication of Our Common Future by the World Commission on Environment and Development (World Commission on Environment and Development, 1987), the idea of sustainable development came to the forefront of the public debate. Moreover, the concept of sustainable development has become a leading paradigm of policy makers and researchers. This World Commission added its voice to the appeal for new ways of measuring progress that would go beyond economic signals and capture a fuller sense of human and ecological well-being (Hardi and Zdan, 1997). However, sustainability proved to be a remarkably difficult concept to define and to apply in practice. Moreover, relevant *measurement* of sustainability is fraught with difficulties of principles and practice. Hence, there are, understandably, but nevertheless disappointingly, rather few published empirical studies on this topic (Pezzey and Toman, 2002*a*).

The need for procedures to measure sustainability is increasingly recognized (Tyteca, 1998). Although the concept has different meanings to different people, it is far from meaningless (Farrell and Hart, 1998).

Indicators can help to identify, define and communicate about sustainability issues and they can be used to forecast and monitor the results of policy choices (ESDI, n.d.). Good indicators provide key information about a physical, a social or an economic system and they allow analysis of trends and cause-and-effect relationships (Veleva and Ellenbecker, 2001). Moreover, indicators of sustainable development should provide solid bases for decision making at all levels (Becker, 1997, Capello and Nijkamp, 2002). Decision makers need indicators that show the links between social, environmental and economic goals to better understand how to achieve economic growth that is in harmony with the natural systems within which we live and work (Farrell and Hart, 1998). Indicators can be used (i) individually, (ii) as part of a set, or (iii) in the form of a composite index that combines individual indicator scores into a single number. Such a single aggregated number can be very useful in communicating information on general sustainability to the public and to decision makers (Farrell and Hart, 1998). Possible disadvantages are that the methods to achieve an aggregation are often subjective (Becker, 1997, Hueting and Reijnders, 2004) and that every index contains hidden assumptions and simplifications (Hanley et al., 1999). Therefore, such combined indicators need to be used judiciously. Farrell and Hart (1998) state that in many cases, indicators to measure sustainability are no more than combined lists of traditional economic, environmental and social indicators with the word *sustainable* added to the title. Nevertheless, such combination is a first significant step because it recognizes that all three areas (economic, ecological and social) matter: sustainable development is a holistic concept and ideally one should strive to consider all three pillars of sustainability simultaneously. Therefore, it is important that the development of indicators does not stop at this stage (Farrell and Hart, 1998). Economic and ecological analysis need to be combined (Kaufmann and Cleveland, 1995) and one should concentrate on the interaction rather than on just the environment itself (Jollands et al., 2003). The advantage of aggregate indicators is that the information is presented in a format tailored to decision makers (Costanza, 2000, Jollands et al., 2003, Azapagic, 2004). However, we need to be careful and informed about the way of aggregation, the uncertainties, the weights and the data source. Decision makers are too busy to deal with these details and the beauty of the aggregate indicator is the fact that it does the job for them (Costanza, 2000). But, no single indicator can possibly answer all questions and therefore multi-dimensional indicators can be needed (Opschoor, 2000, Veleva and Ellenbecker, 2001).

In recent years, different frameworks and indicator systems have emerged that claim to evaluate sustainability both at firm level and at higher level. Most of the focus in measuring and evaluating progress towards sustainable development has been at the national level (Veleva and Ellenbecker, 2000, Figge and Hahn, 2004*a*). Well-known examples are the ecological footprints (e.g., Wackernagel and Rees (1997)), genuine savings (e.g., Pearce and Atkinson (1993)), the index of sustainable economic welfare (e.g., Daly et al. (1989)), and the dashboard of sustainability (e.g., IISD (s.d.)).

Sustainability is a global concept and a firm is only a small subsystem that interacts in various ways with surrounding systems. Nevertheless, companies are essential actors in socio-economic life and as such they contribute to the realization of sustainable development (Tyteca, 1998). Corporations are the organisations with the resources, the technology, the global reach, and ultimately, the motivation to achieve sustainability (Hart, 1997). Defining and measuring corporate sustainability is more than just an academic concern. Corporate entities are increasingly under pressure to demonstrate how they contribute to the national sustainability goals outlined by governments (Atkinson, 2000). The concept behind sustainability indicators for business is simple. On the one hand, the aim of these indicators is to answer the question of how one might objectively know whether a company is moving towards or away from sustainability in all three dimensions: environmental, social and economic (Lawrence, 1997). On the other hand, defining the appropriate indicators is not easy (Veleva and Ellenbecker, 2000).

Assessing a company's contribution to sustainability can be measured by subtracting the costs from the benefits created by that company. For this purpose both internal and external costs and benefits need to be considered. External costs or benefits that arise from production (or consumption) falling on someone other than the producer (or consumer) are called externalities. An example of the assessment of the total external costs for UK agriculture is given by Pretty et al. (2000) and for the agricultural sector of the United States by Tegtmeier and Duffy (2004). If the benefits of a firm exceed the sum of internal and external costs, the firm contributes to sustainability. From a theoretical point of view this way of measuring sustainability provides a very powerful measure of corporate contributions to sustainability because they translate the requirements of the constant capital rule at the macro level into measures at the micro level (Figge and Hahn, 2004*a*), but the necessity to express environmental and social damage in monetary terms, severely limits the practicability.

A very popular measure to express corporate contributions to sustainability is the measure of eco-efficiency (e.g., Schmidheiny (1992), OECD (1998), WBCSD (2000), Jollands and Patterson (2004), Meul et al. (2007b)). Eco-efficiency can be defined as the ratio of created value per unit of environmental impact. Figge and Hahn (2004*a*) state that eco-efficiency has three major shortcomings to measure corporate contributions to sustainability. First, eco-efficiency is a relative measure giving no information on effectiveness. Second, advances in environmental performance due to improved eco-efficiency can be overcompensated because better eco-efficiency may lead to growth and thus increased use of environmental resources. This is called the rebound effect (Mayumi et al., 1998, Herring and Roy, 2002). However, environmental resources which are saved due to improved eco-efficiency might be employed by other companies which are less eco-efficient. Third, eco-efficiency does not take into account all social and environmental impacts simultaneously.

The concept of sustainable development at firm level has been more and more applied in recent years. Well-known examples are The Global Reporting Initiative (e.g., GRI (2002)), ISO 14031 (e.g., ISO (1999)), and the eco-efficiency framework of the World Business Council for Sustainable Development (e.g., WBCSD (2000)). Regardless of the large number of available sustainability indicators for companies, there is no framework to evaluate the sustainability of production systems (Veleva and Ellenbecker, 2000). Tyteca (1998) showed that the principles of productive efficiency can be used to elaborate sustainability indicators at the firm level. Callens and Tyteca (1999) worked out indicators based on both the concepts of cost-benefit analysis and the principles of productive efficiency. van Calker et al. (2006) used the multi-attribute utility theory to develop an overall sustainability function for Dutch dairy farming systems. Besides data at the attribute level, stakeholders and experts are used for the assessment of subjective and objective attributes respectively. The sustainability function showed to be a suitable method to rank and compare different dairy farming systems (van Calker et al., 2006). Krajnc and Glavic (2005*a*) designed a model for obtaining a composite sustainable development index in order to track integrated information on economic, environmental and social performance with time. In other words, they developed an aggregate measure which can be used to compare and rank companies regarding sustainable development. Using the concept of analytic hierarchy process, the impact of individual indicators on the overall sustainability of a company can be assessed. The analytic hierarchy process is a multi-attribute decision model used to derive weights of indicators by the prioritization of their impact on overall sustainability assessment of the company. In Krajnc and Glavic (2005*b*), the effectiveness of the proposed model is illustrated with a case study. Possible drawbacks of this model are the selection of indicators and the way in which the weights of indicators are determined (Krajnc and Glavic, 2005*a*,*b*).

Figge and Hahn (2004*a*) and Figge and Hahn (2005) introduced the concept of sustainable value, a new approach to measure corporate contributions to sustainability, based on the assessment of the value of capital beyond economic capital. They developed a valuation methodology to calculate the cost of sustainable capital and the sustainable value creation of companies. The capital approach and the concept of opportunity costs to determine a company's sustainable value are applied. Using the notion of capital, Costanza et al. (1991) define sustainability as the amount of consumption that can be continued indefinitely without degrading any capital stocks. For this, capital must be considered in the broad sense: inasmuch as the environment contributes to the productive process, it should be considered as a factor of production (El Serafy, 1991).

Using the capital concept can help in conceptualizing the measurement of sustainability and can facilitate the dialogue between economists and ecologists (Harte, 1995). In a broad sense capital consists of (i) natural capital, (ii) physical capital, (iii) human capital and (iv) intellectual capital. Except for natural capital, all other types are considered as human-made capital (Perman et al., 2003). The constant capital rule is the key to the capital approach to sustainability. The sustainable value approach relates the efficiency of capital use by companies (micro level) to the efficiency of a benchmark (macro level) (Figge and Hahn, 2005). A company contributes to more sustainable development whenever it uses every single form of capital more efficiently than other companies. With respect to the micro level the approach shows whether the different forms of capital have been allocated to the most value creating use. On the macro level, the sustainable value approach expresses the excess value created by a company while preserving a constant level of capital use on the macro level. Hence, the approach is based on the notion of strong sustainability (Figge and Hahn, 2004*a*, 2005).

Sustainable value is a monetary measure of sustainability. An important advantage of monetary measures is that it gives decision makers environmental information in a format they are familiar with and that readily enables comparison with other types of information (Farrell and Hart, 1998, Atkinson et al., 1999).

Most approaches to assess sustainability performance are burden orientated; they assess the costs or potential harm of resource use, while the sustainable value approach is value orientated. In fact, burden-orientated approaches focus on the level of environmental impacts caused by an economic activity compared to another set of environmental impacts (how resources should be substituted by each other), while value-oriented impact assessment analysis how much value have been created with this set of environmental impacts as compared with the use of these resources by other companies (where resources should be optimally allocated). Figge and Hahn (2004*b*) state that value- and burden-oriented impact assessments are necessarily complementary and both need to be considered to arrive at an optimal allocation of resources. On the other hand, the approach does not indicate whether the overall capital use is sustainable, but only how much a company contributes to a more sustainable use of capital. Another drawback is that the usability of the methodology is limited by the available data on corporate capital use and the opportunity cost of the different forms of capital (Figge and Hahn, 2005). Inevitably, indicators are often selected considering the availability of reliable data.

However, we decided to use the sustainable value approach in this chapter to measure farm sustainability, mainly because an important advantage of the approach is that it integrates different forms of economic, social and environmental resources. In fact, the sustainable value approach can, contrary to other methods, be seen as a fully integrated value oriented assessment tool which can give useful and good guidance for decision making.

7.3 Methodology

7.3.1 Methodology to measure sustainability

We will call all capital forms (or aspects derived from capital forms) in the remainder of this chapter *resources*, because we assume that they all contribute to the production of value added in a system (see also section 8.2.1).

Benchmarking is a necessary tool to evaluate corporate policies and performance (Krut and Munis, 1998). A firm contributes to more sustainable development whenever it uses every single form of resource more productively then other companies. In practice there is no such super-company, therefore it must be determined if the higher productivity of the use of one form of resource can compensate for the lower productivity of the use of another form of resource (Figge and Hahn, 2005). When assessing different resources, an aggregation problem is encountered. To solve this problem, a conversion of all environmental impacts into (burden) equivalents of one of the impacts may be considered. Figge and Hahn (2004*a*) and Figge and Hahn (2005) use the concept of opportunity cost to transfer impacts into value equivalents. The opportunity cost of a resource is the cost of an opportunity foregone (and the benefits that could be received from that opportunity), or the most valuable foregone alternative. Hence, one can consider the opportunity costs of all resources. The opportunity cost (or resource cost) can be calculated as:

$$opportunitycost = \frac{value\ added_{benchmark}}{resource_{benchmark}} \tag{7.1}$$

A firm creates value when it uses resources more productively than the (bench)mark. Hence, we can calculate the value spread by subtracting the opportunity cost (determined by the benchmark) from the productivity of resource use in company i. The value spread reflects the super-efficiency of the use of a resource (Figge and Hahn, 2005):

$$value\ spread_i = \frac{value\ added_i}{resource_i} - \frac{value\ added_{benchmark}}{resource_{benchmark}} \tag{7.2}$$

The value contribution is calculated by multiplying the value spread of a certain resource of a company with the amount of resource use by that company. The sustainable value created by company i can be calculated by adding up the value contributions for every form of resource s ($s \in [1; n]$). n forms of resources are considered.

$$sustainable\ value_i = \frac{1}{n}\sum_{s=1}^{n}(value\ spread_i^s * resource_i^s) \qquad (7.3)$$

To correct the overestimation of the value created, we divide by the number of resources n to calculate the sustainable value. Dividing by n does not serve to weigh the different forms of resources but only to avoid double counting of value creation (Figge and Hahn, 2005). This measure can be expressed by relating the value created to the cost of resource. Figge and Hahn (2005) calculate a sustainability measure as:

$$return - to - cost_i = \frac{value\ added_i}{value\ added_i - sustainable\ value_i} \qquad (7.4)$$

The more sustainable value is created, the more the value added of a firm exceeds the opportunity cost of its resource base (Figge and Hahn, 2005). The cost of the sustainability resource of company i is given by the difference between the value added and the sustainable value of company i. The return-to-cost ratio[2] of a firm equals unity if the value added corresponds to the cost of all forms of resources. A return-to-cost higher than one means that the company is overall more productive than its benchmark. In contrast to single resource productivities, the return-to-cost ratio considers all forms of resources simultaneously and relates them to the value created (Figge and Hahn, 2005).

The choice of the benchmark reflects a judgement as it determines the cost of all resources. This means that the benchmark level determines the explanatory power of the results of the sustainable value analysis. It should therefore be chosen with great deliberation (Figge and Hahn, 2005). Different figures can be considered as benchmark for national economies, regions, performance targets, a sector or a company. Consider the following four possibilities.

First, the weighted average return on resource of a sample of companies can be used. This benchmark is calculated as the total sum of the value added of all companies in the sample divided by the total resource use of that resource of all firms. Note that that the weighted average return on resource represents the average over all the years considered.

[2]Note that the name of this measure changed from sustainable efficiency (as in Figge and Hahn (2005) and in Van Passel et al. (2007)) to *return-to-cost ratio* (as in ADVANCE (2006) and as in our next chapter), this because this naming is more in line with common used definitions within economics

Second, an alternative benchmark is the best performance on each resource form using the highest return on resource realized by a company. Using for each resource the best scoring farm as benchmark, we create a virtual 'super-farm' as benchmark. The fact that this farm does not exist in the real world is not a problem. A possible disadvantage of this benchmark is the higher risk of using only one observation: the benchmark observation could be an outlier or a wrong observation.

Third, another possibility to benchmark is applying for every resource a performance target. Such an objective performance target for sustainable dairy farming in Flanders is e.g., $150kgNha^{-1}$ for the farm-gate N surplus (Nevens et al., 2006). In this way the method can easily be related to attempts to define sustainable farming systems in a region.

Fourth, in a last benchmark system, the unweighted average return on resource can be used. In this case, every farm will be taken into account regardless of size. This benchmark is calculated as the total sum of return on resource of all companies in the sample divided by the number of companies.

In this chapter, the weighted average of the FADN set of dairy farms is used as benchmark to calculate the sustainable value of each dairy farm in the FADN set. But the results will be compared with the results obtained using other types of benchmarks. We opted for this benchmark because an important aim of this study is to understand why farms differ in their creation of sustainable value and to analyze the evolution of the return-to-cost ratio. Calculating the weighted average as benchmark is much closer to how resources are really used. Using the unweighted average would imply that every farm (regardless of size) gets the same share if resources are put on the market.

The sector benchmark is calculated by the total sum of the value added of all observations in the sample divided by the total use of the resource of all observations covering all years. It can seem unusual to take the average of several periods as benchmark (opportunity cost), because strictly speaking farmers do not have to choose between using a resource in year 1 or 2 but can use the resource in year 1 and 2. Thus farmers do not have to make an exclusionary choice and using a resource in year 2 is therefore strictly speaking not an opportunity cost for year 1. However, in this chapter we use the average of all periods as benchmark because we try to understand the differences in return-to-cost and to identify trends in the variation of sustainability performance of the observations during the observed period. Note that data of the different resources is used in physical amounts or if necessary in deflated monetary amounts, so price effects, which can trouble comparisons between years, are eliminated. A possible drawback of using the benchmark covering all years is that we do not take technical change into account. Finally, using the weighted average covering all years as benchmark results in a better representation of the sector (645 observations) because in some years we have limited observations (e.g., 69 observations in 2001).

7.3.2 Explaining differences

To analyse which factors influence sustainable value creation, an econometric model can be formulated. The essential structure for an effect model using panel data is:

$$y_{it} = \alpha_i + \gamma_t + \beta x_{it} + \epsilon_{it} \tag{7.5}$$

The dependent variable y_{it} is the return-to-cost ratio of each farm in each year, the independent variables x_{it} are the potential determinants. Equation 7.5 can be estimated using unbalanced panel data via an effect model (random or fixed) or via ordinary least squares (OLS) regression. In effect models, variation across farms or time is captured in simple shifts of the regression function (changes in intercepts). Traditionally, there are fixed effects models and random effects models. In fixed effects models α_i is a separate constant term for each unit ($y_{it} = \alpha_i + \beta x_{it} + \epsilon_{it}$). In random effects models we have an individual specific disturbance ($y_{it} = \alpha + \beta x_{it} + \epsilon_{it} + u_i$). To identify the most appropriate estimation the Breusch and Pagan's Lagrange multiplier statistic and the Hausman's chi-squared statistic can be used (Wooldridge, 2000). The Breusch Pagan's Lagrange multiplier statistic can be used to test the use of the effect model against the classical regression with no group specific effects. High values of the Lagrange multiplier statistic argue in favour of one of the effects models (fixed or random). The Hausman test can be used to test the use of the fixed effects model against the random effects model. A low Hausman chi-squared statistic argues in favour of the random effects model.

7.4 An empirical application on Flemish dairy farms

As explained in the previous section, several frameworks to calculate the sustainable value creation of firms exist. These frameworks are mostly tested by calculating the sustainability of only a limited amount of companies. In our analysis, we use an extensive sample of data of a large amount of Flemish dairy farms. In this way, we apply and test the methodology developed by Figge and Hahn (2005). The *value contribution*, the *sustainable value* and the *return-to-cost ratio* are used as indicators to analyse the farm sustainability. Furthermore, an econometric model is constructed to analyse the impact of several managerial and structural farm characteristics on farm sustainability.

7.4.1 Data and variables

There are three questions that need to be addressed for the application of the sustainable value methodology (Figge and Hahn, 2005): (i) the choice of the economic activity or entity to be analyzed; (ii) the choice of the resources to be taken into account; (iii) the choice of the benchmark. In this chapter, we focus on Flemish dairy farms (specialist grazing livestock). Farm accountancy data from a group of dairy farms in Flanders is used during the period 1995-2001. This gives 645 observations (unbalanced panel data) and a balanced panel data set of 41 dairy farms during 7 years (287 observations). These data were collected by the European FADN database (Farm Accountancy Data Network). The Belgian FADN-data are collected and managed by the former Centre for Agricultural Economics (presently taken over by the monitoring division of the Agricultural Monitoring and Study service of the Flemish Ministry for Agriculture). Descriptive statistics of the sample data are given in table 7.1.

The different forms of resources that we take into account are: (i) labor, (ii) farm capital, (iii) utilized land, (iv) energy use and (v) nitrogen surplus. Labor, farm capital and land are traditional economic resources. Farm capital (assets) is calculated as total capital minus land capital, in this way overlap is avoided. Energy use and nitrogen surplus are environmental resources. In Flanders, as in other European regions, N losses are a major concern in agricultural practice. High stocking rates in the Flemish region result in a very high N pressure on the utilized agricultural area. These five different resources are found to be critical for the sustainability performance of a dairy farm (Reinhard et al., 2000, Nevens et al., 2006, Meul et al., 2007b). Information on other important resources (e.g., social aspects and other environmental aspects) was not available in our data set and could not be taken into account.

Table 7.1: Descriptive statistics

Variable	Mean	Minimum	Maximum	Std. Dev.
$Total\ output\ (Euro)$	150293	20445	622791	68765
$Land\ use\ (hectares)$	31.73	6.72	83.08	11.28
$Labor\ (full-time\ equivalent)$	1.48	0.63	3.50	0.34
$Capital\ (Euro)$	284466	37338	789404	152140
$Intermediate\ consumption\ (Euro)$	66361	13600	295465	31535
$Energy\ consumption\ (MJ)$	1248410	268185	3803592	522292
$Nitrogen\ surplus\ (kg\ N)$	8884	1934	25570	3879
$Farmer's\ age$	43	23	68	10
$Solvency^a$	0.42	0.00	1.00	0.30
$Sizeunit^b$	164.6	5.00	59.00	6.47
$Subsidies\ interest^c\ (Euro)$	2953	0	15111	3064
$Subsidies\ revenues^d\ (Euro)$	144	0	5512	531
$Subsidies\ income^e\ (Euro)$	4516	0	70147	3723
$Value\ added\ (Euro)$	83931	6845	327326	42885

[a] measured as own capital divided by total capital
[b] calculated for all farms in the dataset based on standard gross margin (FADN, n.d.)
[c] subsidies on investments (interest support)
[d] ubsidies on animal products (subsidies on sale and purchase of animals are not included)
[e] direct payments to producers: suckler cow premium, slaughter premium, set-aside premium, arable crops hectare aid,...

7.4.2 The evolution of the value contribution, sustainable value and return-to-cost of Flemish dairy farms

Using the balanced data set, the value contribution for the five different resources and the sustainable value (equations 7.2 and 7.3) can be calculated for each dairy farm in each year. The average value contributions and the average sustainable value of the 41 dairy farms can be calculated for each year during the period 1995-2001. As explained in section 7.3.1 the weighted average return on resource covering all years is used as benchmark. The sustainable value approach assesses how productive companies use their resources. Farms use resources to create a return. This return is compared to the opportunity cost of the resources, in this case the weighted average return of all farms in the dataset. This means that a farm only creates sustainable value if it creates more value added with its resources than the weighted average.

Figure 7.1 shows the evolution of the average sustainable value as well as the value contributions of each resource, calculated for the balanced set of 41 dairy farms during the period 1995-2001. The average sustainable value of each year is calculated as the sum of the sustainable value of all observations in one year divided by the number of observations in that year. We observe a negative sustainable value in 1995, increasing to a maximum in 1999. Afterwards, the sustainable value decreases. The value contributions show how much more or less value added the farm has created with a resource compared to the

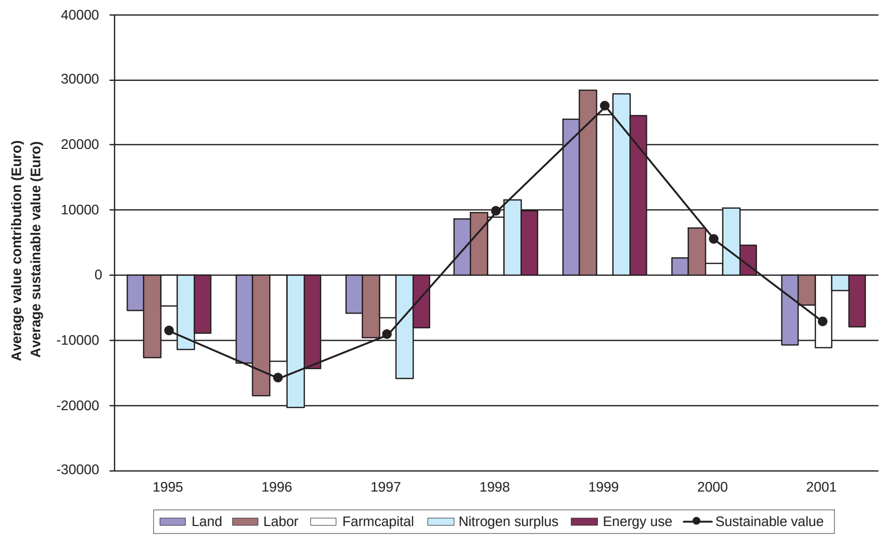

Figure 7.1: The average value contribution of each considered resource and the sustainable value of dairy farms in Flanders (1995-2001)

benchmark. Our analysis identifies which resources highly contribute to value creation and which resources are not used in a value-creating way. For example the contributions of land and farm capital are low in 1995 and in 2000. We can also see in figure 7.1 that nitrogen surplus can be identified as a critical resource and value driver in both 1997 and 2000. Furthermore, the evolution of the value contributions of resources can be analyzed. Nitrogen surplus has had an important bad impact during the first three years (1995-1997). Later on, the improvements of nitrogen use result in a positive contribution on the sustainable value. In this way the reduction of nitrogen surpluses in specialized dairy farms (Nevens et al., 2006, Meul et al., 2007b) has been taken into account calculating the sustainable value of the dairy farms. As mentioned earlier, larger farms can be expected to create higher (positive or negative) sustainable value than small farms. To correct for this size effect, the return-to-cost ratio can be calculated. In this way we can compare the sustainability performance of farms irrespective of their size. Calculating the average return-to-cost (equation 7.4) in each year, a low average return-to-cost of 0.8 is found in 1996. From 1996 on, the average return-to-cost increases, to almost 1.3 in 1999. This top was followed by a decrease in 2000 and 2001. A possible explanation of this decline is the decrease in profitability that has been observed during this period in the specialized dairy sector in Flanders (CAE, 2003). The average return-to-cost ratio of each year is calculated as the sum of the return-to-cost of all observations in one year divided by the number of observations in that year. Note that we used our benchmark covering all seven years (1995-2001), in this way the trends in the variation of the return-to-cost can be identified during this period.

7.4.3 Benchmark choice and robustness of the indicators

As explained in section 7.3.1, the choice of the benchmark is important in calculating the *sustainable value* and the *return-to-cost ratio* indicators. We choose as benchmark the weighted average return on resource of our data sample (benchmark 1 (base)). To analyse the robustness of the return-to-cost calculations, we will compare our results with the results obtained using other types of benchmarks described in section 7.3.1. A first alternative benchmark is the best performance on each resource (benchmark 2). Another possibility to benchmark is applying a performance target (benchmark 3). A nitrogen surplus target of 150 kg N per ha is used and the weighted average return on resource is applied for the other resources. Finally, we used the unweighted average return on resource (benchmark 4).

Table 7.2: Descriptive statistics of the return-to-cost ratio using different benchmarks

Return-to-cost ratio	Mean	Minimum	Maximum	Std. Deviation
Benchmark 1 (base)	0.98	0.28	1.91	0.28
Benchmark 2	0.35	0.10	0.70	0.10
Benchmark 3	0.82	0.32	1.61	0.19
Benchmark 4	0.94	0.27	1.83	0.27

Number of observations: 287 (41 dairy farms during 1995-2001)

The descriptive statistics of the return-to-cost ratio using different benchmarks can be found in table 7.2. Using the nitrogen surplus as benchmark (benchmark 3) results in a lower maximum and mean return-to-cost compared to the base benchmark. The reason is that only eight observations reach the target level of 150 kg N per ha. The return-to-cost ratios using the best performance of each resource as benchmark (benchmark 2) are much lower. Using this benchmark, the maximum possible return-to-cost ratio is 1. A return-to-cost of 1 would mean that the *super-farm* exists, one of the farms scores best on all resources. From our data, we obtain a maximum return-to-cost of 70% (table 2). The descriptive statistics of the return-to-cost ratio using the unweighted average return on resource as benchmark (benchmark 4) are very similar to the statistics of the return-to-cost using the weighted average of eco-efficiency as benchmark (benchmark 1 (base)). This seems to lead to the conclusion that farm size does not matter, although the significance of the impact of farm size on return-to-cost should be analyzed in more detail (see section 7.4.5). Figure 7.2 shows the evolution of the average return-to-cost (during the period 1995-2001). Depending on the applied benchmark, the values of the the return-to-cost ratio differ but the year to year variation is very similar.

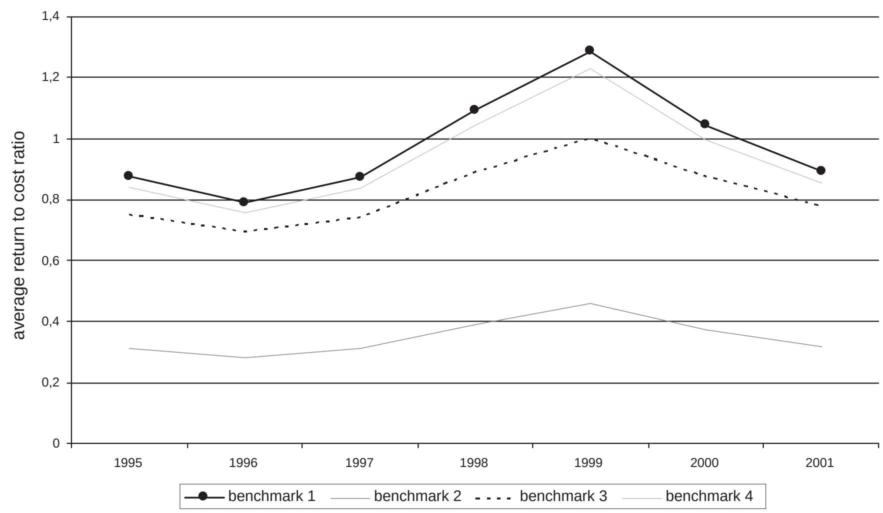

Figure 7.2: The average return-to-cost ratio using different benchmarks (1995-2001)

To test the robustness of the return-to-cost measure, the rank correlation (Spearman's rho) of return-to-cost using different benchmarks is calculated (table 3). The correlations are very high and significant. This indicates that the ranking of the dairy farms does not differ much using the other benchmarks to calculate the return-to-cost ratio. Similar results are obtained analyzing the robustness of the sustainable value measure (not reported). Note that the best scoring farm (maximum) is the same farm using the four different benchmarks. Also the worst scoring farm (minimum) is the same farm.

Table 7.3: Correlation between the rankings of return-to-cost ratio using different benchmarks

Return-to-cost	Benchmark 1[a]	Benchmark 2	Benchmark 3	Benchmark 4
Benchmark 1[a]	1.000	0.9967***	0.9336***	0.9994***
Benchmark 2		1.000	0.9415***	0.9986***
Benchmark 3			1.000	0.9365***
Benchmark 4				1.000

[a] benchmark base using the weighted average return on resource as benchmark
*significant at 10%; ** significant at 5%; *** significant at 1%

Note that within the scope of policy analysis the choice of an accurate benchmark is more important. In that case, using benchmark 2 (the best performance on each resource) or benchmark 3 (performance targets) can be very useful to analyse the efforts of farms in their aim to reach the targets or the best performance. The benchmark choice will be discussed in more detail in chapter 8.

7.4.4 The existence and tenacity of frontrunners and laggards

For a balanced set of 41 Flemish dairy farms, the return-to-cost ratio for seven successive years (1995-2001) is calculated. Again, as benchmark the weighted average return on resoruce is used but this time for each year. In contrast to the other sections, we do not use our benchmark covering all years but we use for each year the weighted average return on resource from that year as benchmark (It follows that the average sustainable value would be equal to zero for each year). We changed our benchmark choice because in this section we analyse the existence of frontrunners and laggards. In other words, we want to know if farms with a high (low) return-to-cost ratio in one year also have a high (low) return-to-cost ratio in the other years. To test the hypothesis that the same farms are obtaining high values of return-to-cost each year, the rank correlation coefficients (Spearman's rho) between the different years are calculated (table 7.4).

Table 7.4: Correlation between the rankings of return-to-cost ratio (1995-2001)

Return-to-cost	1995	1996	1997	1998	1999	2000	2001
1995	1.00	0.58***	0.65***	0.59***	0.41***	0.51***	0.56***
1996		1.00	0.75***	0.74***	0.66***	0.67***	0.60***
1997			1.00	0.79***	0.77***	0.80***	0.70***
1998				1.00	0.70***	0.68***	0.61***
1999					1.00	0.84***	0.67***
2000						1.00	0.67***
2001							1.00

*significant at 10%; ** significant at 5%; *** significant at 1%

All correlations are significant and positive (table 7.4), indicating that on average the same farms have high return-to-cost ratios during the observed period. Hence, we can say that there are in general frontrunners and laggards. In table 7.5 the descriptive statistics of the frontrunners (farms with a return-to-cost higher than 1 in each year) and the laggards (farms with a return-to-cost lower than 1 in each year) can be found. Nearly 22% (9/41) of all farms have a high return-to-cost ratio each year; also almost 30% (12/41) of all farms have a low return-to-cost ratio each year.

Observing the descriptive statistics (table 7.5), we found that the *laggard* farms have older farm managers with a lower education level. These farms are smaller in size and more dependent on support payments, on the other hand the solvency rates are higher and they have a larger share of land in property. The *frontrunner* farms are larger in size and less dependent on support payments.

Table 7.5: Descriptive statistics of all farms, frontrunners and laggards

Variable (average values)	All farms	Farms with a RTC > 1 (for all 7 years) (frontrunners)	Farms with a RTC < 1 (for all 7 years) (laggards)
Sustainable value (Euro)	0	19298	-16677
Age of manager	42.77	43.16	48.40
Solvency[a]	0.424	0.417	0.604
Size unit[b]	16.41	17.70	13.90
Subsidies dependency[c] (in %)	4.9	4.4	5.1
Share own land[d] *(in%)*	28.8	22.5	34.1
Higher education (in%)	66	78	25
Number of observations	41	9	12

RTC stands for the return-to-cost ratio
[a] measured as own capital divided by total capital
[b] calculated for all farms in the dataset based on standard gross margin (FADN, n.d.)
[c] calculated as the total amount of received subsidies divided by the total revenues, indicating the dependency on support payments
[d] measured as land in property (in ha) divided by total land (in ha)

A more profound analysis of the differences in return-to-cost can be found in the following section.

7.4.5 Explaining differences in farm sustainability

In section 7.4.4 we found that there are frontrunners and laggards. In this section the reasons behind the differences in return-to-cost between farms are analyzed.

Among the 645 observations during 7 years (unbalanced panel data), a large difference in the level of return-to-cost is found (figure 7.3). The performance of dairy farming differs clearly a lot. Possible determinants in our dataset which may partly explain the differences in performance are: (i) managerial characteristics (e.g., education of the farm manager), (ii) structural characteristics (e.g., farm size), (iii) milk composition (e.g., protein level) and (iv) farm strategy (e.g., farm growth). To analyse these differences, we calculate the average value of some determinants of all dairy farms and of the 10%-best-scoring-farms and of the 10%-worst-scoring-farms. Remind that we measure the performance using the return-to-cost ratio (equation 7.4). The descriptive statistics are provided in table 7.6.

Observing the descriptive statistics of the managerial characteristics in table 7.6, we see that the best scoring farms have a younger and better educated farm manager. Furthermore, the farms with a high return-to-cost ratio have also more children on the farm and the farm manager and/or partner receive

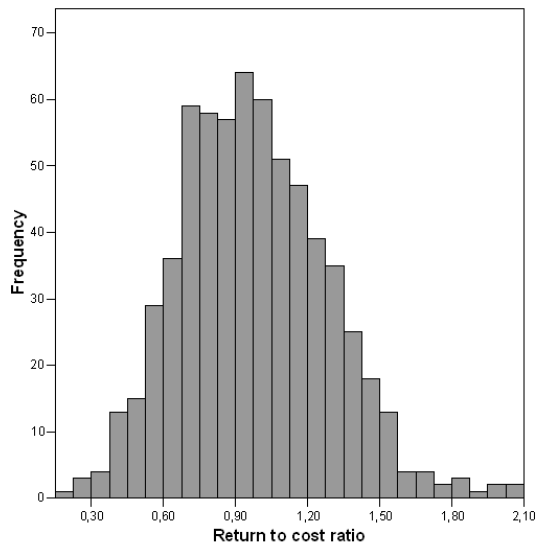

Figure 7.3: The frequency distribution of the return-to-cost ratio of all specialized Flemish dairy observations in the FADN-dataset (all farms during 1995-2001)

more off-farm earnings. Observing the structural characteristics we observe that the best scoring farms are larger (higher size unit, higher milk quota and more cows) and have a lower solvency rate. Farms with a high return-to-cost ratio pay also less additional milk levies but pay more environmental taxes (e.g., manure tax) and are less dependent on support payments. Furthermore, among the best-scoring farms there are fewer farms with only 1 type of milk cow. On the other hand, the best scoring farms have more cows per hectare, in other words these farms have higher cattle intensity. Analyzing the milk composition, we observe a higher milk quality on farms with a high level of sustainability performance. Based on structural and technical data, we can distinguish different farm categories or strategies (Vandermersch, 2006). As in Ondersteijn et al. (2003) and Vandermersch (2006), we observe fine-tuners (high milk quality), growers (growth in milk quota) and diversifiers (share on farm selling). Fine-tuning (high fat and protein level) results in a higher return-to-cost. Growing (increasing milk quota) has a diverse effect, farms with a low return-to-cost ratio barely increase their milk quota level. On the other hand, farms with a high return-to-cost ratio have also a smaller increase of their milk quota level than the overall average.

Analyzing the data in table 7.6, some important differences arise. To determine the significance of the impact of the managerial and structural factors on the farm sustainability, we estimate an econometric panel data model. We can estimate an effect model or an ordinary least squares regression. We found

Table 7.6: Descriptive statistics of all observations, best scoring observations and worst scoring observations (mean values)

Variable	All farms	Best 10% observations	Worst 10% observations
Return − to − cost ratio	0.976	1.553	0.481
Sustainable value	0^a	52432	-36216
Managerial characteristics			
Age of manager	43.10	40.66	47.25
Higher education (in%)	56	58	35
Successor on farm (in%)	11	9	8
No successor on farm (in%)	40	40	34
Doubt about succession (in%)	49	51	58
Number of dependent children	1.61	1.78	1.09
Off-farm earnings[b] *(in%)*	19	31	22
Structural characteristics			
Size unit[c]	16,46	20,45	13,05
Milk quotum(l)	312584	412052	187404
Number of cows	52.4	64.5	39.1
Additional milk levy (Euro)	349	89	425
Solvency[d]	0,42	0,39	0,56
Subsidies interest[e] *(in%)*	1.9	1.5	2.0
Subsidies revenues[e] *(in%)*	0.1	0.0	0.4
Subsidies income[e] *(in%)*	3.1	3.1	4.3
Environmental tax (Euro)	266	379	169
Shareland[f] *(in%)*	28.5	29.9	33.1
Only 1 type milk cow on farm (in%)	38	29	40
Cattle density (number of cows/ha)	1.70	1.83	1.44
Milk composition (quality)			
Fat level (l)	23526	31974	12621
Protein level (in%)	3.464	3.485	3.411
Milk quota growth (l)[g]	4456	2249	525
Share on-farm selling (in%)	1.29	1.18	1.19
Number of observations	645	65	65

[a] the mean sustainable value equals zero which is the consequence of choosing the weighted average as benchmark
[b] off farm income is measured as a dummy, (1 if farmer or partner receive significant off farm earnings)
[c] calculated for all farms in the dataset based on standard gross margin (FADN, n.d.)
[d] measured as own capital divided by total capital
[e] the subsidies are calculated as a percentage of total revenues, indicating the dependency on support payments
[f] measured as land in property (in ha) divided by total land (in ha)
[g] calculated as the difference in milk quota of the actual year with the previous year

a large Breusch and Pagan's Lagrange multiplier statistic; hence, the use of an effect model is preferred. Furthermore, we found a large Hausman's chi-squared statistic, which argues in favour of the fixed effects model. Therefore, equation 7.5 can be rewritten as:

$$
\begin{aligned}
Return-to-cost_{it} = {} & \alpha_i & & +\beta_1(Age)_{it} \\
& +\beta_2(Age^2)_{it} & & +\beta_3(Diploma)_{it} \\
& +\beta_4(Successor\ on\ farm)_{it} & & +\beta_5(Doubt\ about\ succession)_{it} \\
& +\beta_6(Size\ unit)_{it} & & +\beta_7(Solvency)_{it} \\
& +\beta_8(Subsidies\ interest)_{it} & & +\beta_9(Subsidies\ revenues)_{it} \\
& +\beta_{10}(Subsidies\ income)_{it} & & +\beta_{11}(Share\ own\ land)_{it} \\
& +\sum_{j=1}^{n=6}(Yeardummy)_j & & +\varepsilon_{it}
\end{aligned}
\tag{7.6}
$$

Table 7.7: Panel data estimation of determinants of the return-to-cost ratio

Variable	Coefficient	St. Error	Variable	Coefficient	St. Error
Age	-0.0292**	0.0132	$D-1995$	-0.0289	0.0283
Age^2	0.0003**	0.0001	$D-1996$	-0.0939***	0.0276
Diploma	-0.0189	0.0496	$D-1997$	-0.0514	0.0296
Successor on farm	-0.0081	0.0403	$D-1998$	0.1237***	0.0303
Doubt about succession	0.0186	0.0279	$D-1999$	0.4058***	0.0252
Size unit[a]	0.0080*	0.0050	$D-2000$	0.1035***	0.0280
Solvency[b]	-0.0837	0.0788			
Subsidies interest[c]	-1.7017***	0.6041			
Subsidies revenues[c]	0.8368	3.3525			
Subsidies income[c]	-2.0310***	0.6087			
Share own land[d]	0.0339	0.0989			
Number of observations	645		Lagrange multiplier		403,25
Adjusted R-squared	0.80		Hausmann statistic		38,47

Dependant variable: return-to-cost ratio
*significant at 10%; ** significant at 5%; *** significant at 1%
[a] calculated for all farms in the dataset based on standard gross margin (FADN, n.d.)
[b] measured as own capital divided by total capital
[c] the subsidies are calculated as a percentage of total revenues, indicating the dependency on support payments
[d] measured as land in property (in ha) divided by total land (in ha)

The results of the calculation of the one way fixed effects model can be found in table 7.7. Both structural and managerial farm characteristics have an impact on the return-to-cost of the farms (table 7.7). A significant structural farm characteristic is size. Larger farms have a higher return-to-cost. Further, life cycle aspects determine the sustainability performance of farming. Age, a managerial characteristic, has a significant negative effect on the return-to-cost of dairy farms. Generally, younger farmers have a higher return-to-cost ratio. However, from a certain age the negative impact of age is decreasing (indicated by the squared age term in the model). Finally, higher dependency on support payments results in a lower return-to-cost ratio. It is important to note that the used approach is unconditional. We assume that subsidies have an impact on the return-to-cost ratio but the return-to-cost ratio has no impact on subsidies. To solve a part of this problem the same model was tested but instead of using the subsidies dependency in year t the subsidies dependency in year t-1 was used. The results of this lagged model were similar and confirm our results. Finally, observing the year dummy variables in our model, we can demonstrate that the observed differences between the years (figure 7.1) in the return-to-cost were significant.

Furthermore, we can expand equation 7.6 with variables describing different farm categories. Therefore we add variables about the milk quality (fat level and protein level), about the growth in milk quota and about the share on farm selling of milk products (milk, ice cream, cheese,...). Once again we assume that our approach is unconditional, meaning that these variables have an impact on the sustainability performance (return-to-cost ratio) but the performance has no impact on these variables. The results of the estimation of the expanded one way fixed effects model can be found in table 7.8. The results of table 7.8 are similar with the results of table 7.7: indicating that both structural (size, solvency and dependency on support payments) and managerial (age) characteristics explain differences in return-to-cost. Furthermore, we found that higher amounts of additional levies on milk production result in a very small but significant decrease in return-to-cost. Finally, farms with a larger share of on-farm selling of milk or milk products have a significant higher return-to-cost.

Table 7.8: Panel data estimation of the enlarged determinants of the return-to-cost ratio

Variable	Coefficient	St. Error	Variable	Coefficient	St. Error
Age	-0.0292**	0.0132	$D-1995$	-0.0327	0.0284
Age^2	0.0003**	0.0001	$D-1996$	-0.0982***	0.0277
Diploma	-0.0195	0.0490	$D-1997$	-0.0443	0.0298
Successor on farm	0.0081	0.0403	$D-1998$	0.1279***	0.0305
Doubt about succession	0.0214	0.0277	$D-1999$	0.4067***	0.0250
Size unit[a]	0.0104**	0.0050	$D-2000$	0.1104***	0.0278
Solvency[b]	-0.1124	0.0785	*Additional levy*	-0.0000**	0.0000
Subsidies interest[c]	-1.3376**	0.6103	*Fat level*	0.0000	0.0000
Subsidies revenues[c]	1.0243	3.3640	*Protein level*	0.0042	0.0095
Subsidies income[c]	-1.9241***	0.6009	*Growth in milkquota*[e]	0.0000	0.0000
Share own land[d]	0.0694	0.0982	*Share on-farm selling*	1.0435***	0.2812
Number of observations	645		Lagrange multiplier	394.08	
Adjusted R-squared	0.80		Hausmann statistic	40.87	

Dependant variable: return-to-cost ratio
*significant at 10%; ** significant at 5%; *** significant at 1%
[a] calculated for all farms in the dataset based on standard gross margin (FADN, n.d.)
[b] measured as own capital divided by total capital
[c] the subsidies are calculated as a percentage of total revenues, indicating the dependency on support payments
[d] measured as land in property (in ha) divided by total land (in ha)
[e] calculated as the difference in milk quota of the actual year with the previous year

7.4.6 Sustainability performance versus economic and ecological performance

As indicated above, the return-to-cost ratio used in this chapter is an aggregated indicator taken into account different aspects of sustainability. Therefore, it is interesting to compare our results using the return-to-cost ratio with other measures, such as partial productivity measures, eco-efficiency measures and traditional (economic) efficiency measures.

Partial productivity measures take only one factor into account and are calculated as value added/input. Examples are labor, capital and land productivity. Eco-efficiency is calculated as value added/environmental impact. To test the interplay between the return-to-cost measure, the partial productivity measures and the eco-efficiency measures, the rank correlation (Spearman's rho) between these measures is calculated (table 7.9). The partial labor productivity, capital productivity and land productivity are used to measure economic performance. The eco-efficiencies of nitrogen surplus and energy use are used to measure environmental performance. We found that farms with a high return-to-cost ratio also have a high productivity (the rank correlation with capital productivity is low but still positively significant) and a high eco-efficiency (table 7.9). This indicates that the return-to-cost indicator can be a useful indicator to incorporate economic and environmental aspects. The low correlation between capital productivity and the other productivity and efficiency measures may indicate that to increase the productivity or performance of other resources, financial capital is needed and thus increasing land or labor productivity or environmental performance decreases the capital productivity.

Table 7.9: Correlation between the rankings of the return-to-cost ratio, partial productivities and eco-efficiencies

	Return-to-cost ratio	Partial productivities			Eco-efficiencies	
		Labor	Land	Capital	Energy	Nitrogen
Return-to-cost ratio	1.00	0.59***	0.58***	0.25***	0.65***	0.69***
Labor productivity		1.00	0.51***	-0.08	0.32***	0.36
Land productivity			1.00	-0.11***	0.51***	0.47***
Capital productivity				1.00	0.02	-0.01
Eco-efficiency energy					1.00	0.67***
Eco-efficiency nitrogen						1.00

*significant at 10%; ** significant at 5%; *** significant at 1%

In general, our results lead to the conclusion that sustainable farms (high return-to-cost ratio) have both good economic and environmental results. It means that economic performance may go hand in hand with environmental performance, indicated by the positive correlation between the partial productivity measures and the eco-efficiency measures and need not to be opposite to each other as is often thought. Finally, there is also a positive significant correlation between the eco-efficiency of energy and the eco-efficiency of nitrogen surplus. This confirms the conclusion of Meul et al. (2007b) that there is a positive relationship between the nitrogen use efficiency and the energy use efficiency on Flemish dairy farms.

Furthermore, we could also analyze the link between the return-to-cost ratio (indicating sustainable performance) with traditional (economic) efficiency measures such as technical, allocative and economic efficiency. These efficiencies are calculated by estimating the production frontier using stochastic frontier analysis (random effect model). Selected inputs were labor, total capital, nitrogen and concentrates. Output is measured in litres of milk. The Kopp and Diewert (1982) cost decomposition procedure is used to estimate technical, allocative and economic efficiencies (as in Bravo-Ureta and Rieger (1991) and Singh et al. (2001)). Table 7.10 reports results of economic performance analysis (technical, allocative and economic efficiencies). Farms with a high return-to-cost ratio are farms with a high technical and allocative efficiency (and thus also a high economic efficiency).

Table 7.10: Efficiencies of all observations, best scoring and worst scoring observations

Variable	All farms (mean value)	Best 10% *observations*[a]	Worst 10% *observations*[a]	F-value
Technical efficiency (in %)	85.7	89.2	80.3	31.8[b]
Allocative efficiency (in %)	49.0	51.1	40.4	22.2[b]
Economic efficiency (in %)	42.0	45.7	32.2	37.8[b]

[a] measured as the return-to-cost ratio indicating sustainability
[b] F-values are significant (higher than the test value 3.00 (F(0.95;2;774)))

Executing a one way anova, we can test if there are significant differences between the efficiencies (technical, allocative and economic) in the different groups (all observations, 10 % worst and best observations). High F-values are found, indicating significant differences in respectively technical, allocative and economic efficiency between all observations and the observations with very high and low return-to-cost scores. Furthermore, we found that farms with a high (low) return-to-cost have a significant higher (lower) economic efficiency than the average farm. This means that in general economic performance goes hand in hand with sustainability performance.

7.5 Conclusion and discussion

Kaufmann and Cleveland (1995) pose the following appeal: *Enough with theoretical arguments about sustainability indices! To study sustainability, quantify the economic and ecological determinants of the temporal and spatial patterns of the supply and demand, and model these dynamics. Just do it!* With the presented research, we tried to answer this call by focusing very particularly on the assessment of sustainable value creation in one activity of the Flemish agricultural sector.

Although sustainability is a global concept and a farm is only a small subsystem that interacts in various ways with surrounding systems, companies are essential actors to contribute to the realization of sustainable development. Indicators are needed to answer the question of how one might know whether a company is moving towards or away from sustainability. More specifically, using sustainability indicators is important for several reasons. Indicators can be used to educate farmers and other stakeholders about sustainable production. Furthermore, indicators provide farmers with a tool to measure their achievements toward sustainability. Further, indicators allows for comparisons between farms' performance in the economic, social and environmental aspects of their production. Indicators also inform policy makers about the current state and trends in farm performance or sector performance. Sustainability performance measures can be used as input for policy tools and stimulate better integration of decision-making. Finally, sustainability indices can encourage public participation in sustainability discussions.

An interesting approach to assess firm sustainability is the use of a capital approach (or resource approach) and the concept of opportunity costs to determine the company's value. The calculation of the sustainable value and return-to-cost ratio as in Figge and Hahn (2005) is a simplified and quantified expression for the complex farm system. While no measure of sustainability can be perfect, the sustainable value is a useful measure and describes the current sustainability performance. On the other hand, the return-to-cost indicator can be used to compare and rank farms. Benchmarking can help farmers and policy makers to highlight opportunities for improvement and where best practices might be found. Our analysis shows that this methodology is an interesting approach. This because it is possible to assess sustainable development of agriculture in an integrated way that provides good guidance for decision making. An important advantage of the methodology is the fact that it gives decision makers environmental information in a form they are familiar with and which can readily be compared with other types of information. The *sustainable value* and *return-to-cost* measures link many sustainability issues and thus, the number of decision-making criteria that need to be considered is reduced. A possible disadvantage is that information about some important social and environmental aspects is not available in current data samples and is often not quantifiable.

This methodology has already been tested using large multinational firms (e.g., Figge and Hahn (2005)). Our analysis shows that the methodology is also useable and workable for smaller enterprises. European agriculture is still dominated by family farms and agricultural products are produced by small to medium sized enterprises. Moreover we applied this methodology using a large amount of data. Calculating the sustainable value and return-to-cost ratio of 41 dairy farms during seven years provides us with several insights. Analysing the evolution of the sustainable value is more accurate than analysing the evolution of the value added, precisely because environmental aspects are incorporated in the calculation of the *sustainable value* measure. For example in our application of the Flemish dairy farms, the improvement of nitrogen use is incorporated in the calculation of the sustainable value.

Not only the performance measurement itself, but also the analysis of reasons why farms differ in return-to-cost ratio may yield new insights and can give feedback to the concerned policies and government interventions. The large amount of data available in the FADN database creates the opportunity to analyse differences in return-to-cost using an empirical model. The effect model captures potential determinants of the return-to-cost ratio of farming. We found that both managerial and structural farm characteristics explain differences in the sustainability performance of dairy farms. Our empirical model shows that farm size, farmer's age and the dependency on support payments are important characteristics in explaining differences in return-to-cost. We also found that farms with diversification strategies have higher sustainability levels. For example farms who decide to sell milk (products) directly to costumers on their farm obtain a higher level of return-to-cost ratio, because they can improve the value added of the produced milk.

The average weighted benchmark is preferred in our analysis because the main objective of this chapter was to understand why farms differ in their return-to-cost. This benchmark is found to be robust. However, within the scope of policy analysis the choice of an accurate benchmark may be more important. In that case using performance targets as benchmark may be very useful to analyse the efforts of farms to reach these targets. Therefore we suggest using either the ideal farm as benchmark or using given policy targets (such as the nitrogen target).

Our analysis further revealed that over the observed period (1995-2001), the same farms were found to contribute most towards sustainability. Hence, this proofs the existence of frontrunners and laggards among the farms in our data sample. We also found that sustainable farms (high return-to-cost ratio) appear to have both good economic and good environmental results or in other words that economic and environmental performance is not contradictory. Moreover, we found a low correlation between the capital productivity and the return-to-cost measure. This may indicate that increasing the return-to-cost of dairy farms requires financial capital or in other words that there may be a trade-off or substitution between financial capital and the other resources. Furthermore, our results reveal that farms with a high economic efficiency show to have a high return-to-cost, indicating that they create higher economic value using their resources. Hence, our results show that the aim of the European policy makers to combine strong economic performance with the sustainable use of resources is attainable and achievable and not far-fetched.

Finally, we can draw some lessons for public authorities. Policy measures that improve the passing of farms from elder less efficient farmers to younger farms will contribute to an improvement in sustainability performance. Based on the results of our study, policies to improve the exit of farms with elder farmers as well as policies to improve the transition of their farms and/or production assets to younger and more sustainable farms may contribute to an overall improvement in sustainability performance. Hereby subsidies should however been applied with care because our outcomes also suggest a negative correlation of the return-to-cost ratio with subsidies on investments. Apparently farms depending on subsidies are not stimulated to search for higher value added solutions while a high value added proves to be very important both for the economic performance as for the sustainability performance of farms. Therefore, farm policies should give incentives to develop value added strategies rather than keeping less economic efficient farms in production. For example, first installation subsidies could be subject to a more careful examination of the sustainability performance of a farm and made conditional on certain sustainability investments. On the other hand, it may be that subsidies result in certain positive amenities or have social benefits (e.g., survival) which were not included in our analysis. Also the problem of conditionality may be a problem as it is possible that it is an objective of policies to support farmers with a lower sustainability performance. Finally, our results indicate that stimulating on-farm selling of farm products or other diversified activities can contribute to a more sustainable dairy sector in Flanders.

Our results do not incorporate other contributions of farming to society such as contributions to biodiversity or landscape creation which may contribute to the creation of wealth in other rural sectors such as the real estate, tourism or drinking water provision sector. Further research taking into account these positive externalities should be stimulated. Another challenging topic for the future, besides the consideration of positive externalities, is the analysis of sustainability performance up or down the value chain.

In this chapter the sustainable value of a farm is compared within the Flemish dairy sector. In other words an intra-sector comparison (or best in class approach) was made. Note that benchmarking within a sector only shows the potential for improvements within a given activity. This implies that the agricultural structure remains constant and that dynamics are not taken into account. Comparing the sustainability performance of farms of different agricultural sectors would be very interesting but very difficult because of data constraints. Even though further development is desirable, the methodology employed in this chapter to measure farm sustainability proofs to be promising.

Chapter 8

Sustainability assessment through combining the sustainable value approach with efficiency analysis

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Measure what is measurable, and make measurable what is not
—Galileo Galilei

ABSTRACT

Appropriate assessment of firm sustainability facilitates actor-driven processes towards sustainable development. The methodology in this chapter builds further on two proven methodologies for the assessment of sustainability performance: it combines the sustainable value approach with frontier methods to analyze efficiency. The sustainable value methodology tries to relate firm performance to the use of different resources. This approach assesses contributions to sustainability of a system by comparing its resource productivity with the resource productivity of a benchmark, and this for all resources. The efficiency of a system is calculated by estimating the production frontier indicating the maximum feasible production possibilities. In this research, the sustainable value approach is combined with efficiency analysis methods to benchmark sustainability assessment. In this way, the production theoretical underpinnings of efficiency analysis enrich the sustainable value approach. The methodology is presented using two different functional forms: the Cobb-Douglas and translog functional form. The simplicity of the Cobb-Douglas functional form as benchmark is very attractive but it lacks flexibility. The translog functional form is more flexible but has the disadvantage that it requires a lot of data to avoid estimation problems. Using frontier methods for deriving firm specific benchmarks has the advantage that the particular situation of each company is taken into account when assessing sustainability.

8.1 Introduction

Sustainable development is now an important priority for many countries. Two economic paradigms of sustainable development can be distinguished: weak sustainability and strong sustainability. Weak sustainability is based on the idea that natural resources can to a certain extent be substituted as a direct provider of utility for the production of consumption goods (Neumayer, 2003). However, proponents of the strong sustainability view refuse this paradigm because they regard natural resources as non-substitutable. While weak sustainability could be seen as an extension to neoclassical economics, strong sustainability calls for a paradigmatic shift away from neoclassical environmental and resource economics towards an *ecological economics* (Neumayer, 2003). Ecological economics sees the human economy as part of a larger web of interactions between economic and ecological sectors (Costanza et al., 1991). Adherents of the weak sustainability paradigm favor marginal forms of analysis and tend to pay less attention to the concepts of the scale of an economy in relation to its resource base (Norton and Toman, 1997). Daly (1990) was an important architect of the strong sustainability view that advocates that resource substitutability is very limited and the sustenance of specific resource sectors is important (Pezzey and Toman, 2002*a*). Daly (1991*a*) states that: *Just as firms or households of the economy operate as a part of the aggregate economy, so the aggregate economy is likewise a part of a larger system, the natural ecosystem. Therefore, optimal allocation of a given scale of resource flow within the economy is one thing; optimal scale of the whole economy relative to the eco-system is an entirely different problem.* A profound description of the different notions of sustainability can be found in chapter 3.

The sustainable value approach developed by Figge and Hahn (2004*a*) and Figge and Hahn (2005) on which this chapter builds, leaves the total amount of each resource unchanged on the macro level and it can therefore be seen as an approach to measure strong sustainability. The focus is on the scale of an economy or part of an economy in relation to its resource base. In addition, the *sustainable value* approach can be seen as a value-orientated impact assessment of economic activities. Value-orientated approaches integrate economic, environmental and social aspects with respect to their opportunity costs, and analyze how much value is foregone when a bundle of resources is used. In other words, the value-orientated approach proposes where resources should be allocated; it addresses the question how much value would have been created with this set of resources if they had been used by more sustainable efficient firms (real companies or not). Remark that most approaches use a burden-oriented logic by concentrating on different environmental (and social) impacts in order to measure the overall damage (*the burden*) caused by economic activity (e.g., Pretty et al. (2000); Tegtmeier and Duffy (2004)). Burden-orientated approaches focus on the relative harmfulness of environmental and social impacts. In other words, burden-value orientated approaches analyze how resources should be substituted by each other by assessing the combination of environmental impacts compared to another set of environmental impacts.

It is obvious to use the most *sustainable* combination of resources within systems. In fact, less sustainable resource use should be (partly) substituted by more sustainable resource use. However, it is important to analyze and to compare sustainability between systems. Improvements in sustainability may also be searched by substituting companies that use their resources in an unsustainable way by companies that use their resources in a more sustainable way. The value-orientated sustainable value approach therefore assesses sustainability between systems by comparing the resource productivity of a system with the resource productivity of a benchmark (= the opportunity cost) and this for all resources. Policy makers and company managers can use the sustainable value approach to measure, monitor and communicate their sustainability performance. Furthermore the sustainable value approach can be used to identify characteristics of out- and underperformers (as in chapter 7). Moreover, future performance scenarios can be constructed to compare possible firm or policy actions. Policy makers can use the simulation results to take well founded decisions within a sustainability framework.

The choice of the most appropriate benchmark is important, especially within the scope of policy analysis but also for choosing the appropriate actions to realize the firm's objectives. Hence, using best performance or performance targets of each resource as benchmark can be very useful to analyze the efforts of firms in their aim to reach sustainability. To determine the system benchmark, frontier methods can be applied. Such methods can be used to assess sustainability within systems (as in Reinhard et al. (2000)). This chapter will use frontier methods to determine the sustainable value, and thus to assess sustainability between systems (or companies). Frontier methods (and efficiency analysis) can reveal linkages between the output and the resources used by firms, and in that way enrich the sustainable value approach. The approach compares the resource productivity of a company with the maximum feasible resource productivity of that company.

First, the theoretical background is formulated and the research objectives are explained (section 8.2). In section 8.3, the theoretical integration of frontier methods with the sustainable value approach is explained using two functional forms. In section 8.4, the proposed methodology is applied using two empirical applications (one for each functional form). Furthermore, the approach is tested on a data set of Flemish dairy farms. Finally, in section 8.5, conclusions and suggestions for further research are made.

8.2 Theoretical background

Economic, social and environmental efficiency can be seen as a necessary - but not sufficient - step towards sustainability (Callens and Tyteca, 1999, Templet, 2001). Sustainability can be enhanced by strategies which promote resource use efficiency in economic systems (Templet, 1999). Efficient use of resources forms the keystone of policy, planning and business approaches to sustainable development but there exists a wide range of potential interpretations of the efficiency concept (Jollands, 2006*a*,*b*). Because in our methodology several concepts of efficiency are used (e.g., technical efficiency, productivity, eco-efficiency), we start by explaining these concepts to avoid misunderstanding. Jollands and Patterson (2004) showed that efficiency is a core focus within economics, thermodynamics and ecology with as consequence that the term represents a multiplicity of meanings (Jollands, 2006*a*). Remind that all efficiency concepts are relative and context-dependent (Stein, 2001).

The idea of using production economics (frontier methods) in sustainability assessment is not new. Tyteca (1996) used production economics to define standardized, aggregate environmental performance indicators. These indicators do not require the specification of any a priori weight on the environmental impacts that are being aggregated (Tyteca, 1996). Callens and Tyteca (1999) and Tyteca (1998) worked out indicators of sustainable development using the principles of productive efficiency. In fact, they developed a model using an approach that is similar to the one normally to quantify output, input or overall productive efficiency. In our approach we start from a sustainability assessment method (the sustainable value approach) and use frontier methods to benchmark the value of firm resources.

After defining the efficiency key concepts (section 8.2.1), the sustainable value approach (section 8.2.2) and the objectives of the research (section 8.2.3) are explained.

8.2.1 Defining key concepts

There exist several definitions of productivity, efficiency and eco-efficiency. In this dissertation commonly accepted definitions within production economics are used. Productivity is calculated by dividing output by input. Farrell (1957) defines efficiency as the actual productivity of a company compared to maximum attainable productivity, measured by dividing the output by the input.

Besides productivity and efficiency, one can measure performance also in terms of eco-efficiency. A broadly accepted criterion for corporate sustainability is the eco-efficiency measure (e.g., Schmidheiny (1992); OECD (1998); WBCSD (2000)). Eco-efficiency, standing for a better management of the economy with less environmental pressure, presents a promising sustainability approach (Bleischwitz and Hennicke, 2004). There is a wide and diverse variety of terminology referring to eco-efficiency. A well-known definition of eco-efficiency is the ratio of created value per unit of environmental impact. In fact, this variant of eco-efficiency can be seen as environmental productivity (Huppes and Ishikawa, 2005), and is similar to the definition of productivity in economics.

So far, we used the terms *input* and *output*. Output can be expressed as total production (total yield) or as value added (total output minus intermediate consumption). To obtain value added as output, economics traditionally distinguishes land, labor and capital goods as inputs. These inputs are also called factors of production, which are resources used in the production of goods and services in economics. In a more or less similar way, the concept of capital can be used to identify resources used to produce output. Land, capital goods and labor can be seen as capital forms. To assess corporate sustainability a much broader interpretation of the concept of capital than traditionally used by economists, is needed (Dyllick and Hockerts, 2002). Pfeffer and Salancik (1978) define a resource as those means that an organization needs in order to survive. In fact the core argument of their resource dependency theory states that (i) organizations will respond to demands made by external actors or organizations upon whose resources they are heavily dependent and (ii) organizations will try to minimize that dependency when possible (Pfeffer and Salancik, 1978, Pfeffer, 1982). Frooman (1999) even states that the resource dependency theory defines a resource as essentially anything an actor perceives as valuable. In the language of traditional strategic analysis, firm resources are strengths that firms can use to conceive of and implement their strategies that improve its efficiency and effectiveness; firm resources include all assets, capabilities, organizational processes, information, knowledge,... (Barney, 1991). Therefore, we do not make any distinction between conventional economic resources (inputs or production factors) and environmental and social assets. Physically speaking certain environmental assets are (undesired) outputs rather than inputs. However, because companies need to emit pollutants to be able to produce value added, these environmental aspects can be seen as inputs from an economic point of view (Figge and Hahn, 2005). We will call all capital forms (or aspects derived from capital forms) in the remainder of this chapter *resources*, because we assume that they all contribute to the production of value added in a system. We prefer the term resources over the terms inputs or production factors or capital forms (economic, social and environmental) to indicate the assets used to create value. A detailed discussion about the treatment of environmental and social resources as inputs or as undesired outputs falls beyond the scope of this chapter. We refer for this to Färe and Grosskopf (2003) and Hailu (2003).

8.2.2 The sustainable value approach

The sustainable value approach is developed by Figge and Hahn (2004*a*) and Figge and Hahn (2005) and applies the logic of opportunity costs to the valuation of resources. Using the capital approach (e.g., Atkinson (2000)), all resources (economic, environmental and social) are needed to create value. Adhering to a paradigm of strong sustainability, we consider that a firm contributes to more sustainable development whenever it uses its resources more productive than other companies.

Following steps are required in the approach to calculate the sustainable value of a company. First, the scope of the analysis needs to be determined. In other words, which economic activity or activities or entity or entities will be chosen? Second, the relevant resources to take into account (e.g., labor and land) need to be determined. Theoretically, the choice should include those resources that are critical for the sustainability performance of the company within the chosen scope. Third, the benchmark level needs to be determined. The choice of the benchmark determines the cost of the resource use of a company, in other words the productivity that a company has to exceed. The benchmark choice reflects a normative judgement and determines the explanatory power of the results of the sustainability assessment.

Table 8.1 shows the calculation of the sustainable value for a dairy farm with a value added of 80 000 Euro. This company represents a dairy farm with 55 milk cows, 30 ha of land and a milk quota of 300 000 litres.

Table 8.1: Example of the calculation of the sustainable value

Resources[a]	Amount used by the company A	Productivity (80 000/A) Company B	Benchmark C	Value contribution (Euro)
Land	30 ha	2667	2600	2010
Labor	1.00 fte[b]	80 000	50 000	30 000
Farmcapital	300 000 Euro	0.27	0.27	0
Energy use	1 000 000 MJ	0.08	0.07	10 000
N-surplus	6000 kg N	13.33	17.78	-26 700
			Sustainable value =	3062

[a] Remind that we define resources as capital forms (economic, environmental and social) or aspects derived from capital forms
[b] Fte: full time equivalent

The amount used of every resource can be found in column A of table 8.1. The productivity (or return on resource) of each resource can be calculated (column B). For example, the return on land is 2667 Euro per hectare of land (80 000 Euro / 30 ha). In the same way the productivity of the benchmark (column

C) can be determined, these are the opportunity costs. In this example, we choose as benchmark the average return on resource of a large sample of dairy farms (as in chapter 7). For the farm gate N-surplus, we choose a performance target (150 kg N/ha) as benchmark. This is an objective performance target for sustainable dairy farming in Flanders (Nevens et al., 2006).

In a next step, the value contributions of each resource can be calculated ((B-C)*A in table 8.1). A positive value contribution indicates that the resource is used in a value-creating way by that company. To determine how much value is created by the entire bundle of resources, the sustainable value can be calculated by summing up all value contributions and by dividing this value by the number of resources. The sustainable value approach indicates how much more or less return has been created with the resources available in comparison with the benchmark. To take the company size into account, ADVANCE (2006) suggests calculating *the return-to-cost ratio*. This ratio was called *sustainable efficiency* in Figge and Hahn (2005) and in Van Passel et al. (2007), but the term return-to-cost terminology is more consistent with the efficiency and productivity concepts. The return-to-cost ratio is calculated by dividing the value added of a company by the cost of the sustainable resource. The cost of sustainable resource use is given by the difference between the value added and the sustainable value. The return-to-cost ratio equals unity if the value added corresponds to the cost of all resources. A return-to-cost ratio higher than one means that the company is overall more productive than its benchmark. In our example the return-to-cost of the farm is 1.04 (= 80 000 Euro / (80 000 Euro - 3062 Euro)). The return-to-cost ratio shows by which factor the farm exceeds or falls short of covering its cost of economic, environmental and social resources or in other words by which factor it exceeds or falls short compared with the benchmark productivity.

Remark that the sustainable value approach does not claim that the benchmark is sustainable. In other words, the approach does not indicate whether the overall resource use is sustainable, but only how much a company contributes to a more sustainable use of its resources than the benchmark. Another drawback is that the usability of the methodology is limited by the available data on corporate resource use and the opportunity cost of the different resources (Figge and Hahn, 2005). The sustainable value approach does not take qualitative aspects of sustainability into account. All relevant aspects should be quantified in a meaningful way. However, the sustainable value approach allows integrating economic, environmental and social performance. Rather than looking at how burdensome the use of resources is, it compares the value that can be created with the resource by different economic actors. The sustainable value approach is the first value-based methodology that allows an integration of different resources of companies and thus can be used to compare sustainability between systems.

8.2.3 Objectives

As already explained, the choice of the most appropriate benchmark is essential when using the sustainable value approach, because the benchmark determines the opportunity costs of each relevant resource. Moreover, the choice of the benchmark depends on the particular research objective. For example to assess the sustainability performance of BP, Figge and Hahn (2005) used the UK economy as benchmark. Within the ADVANCE project the sustainable value of 65 European manufacturing companies was calculated, although only environmental resources were considered. The EU-15 benchmark was used to calculate the sustainable value of each company. Assuming that environmental resources are not yet used in a sustainable way in the EU-15, a second benchmark was applied using performance targets. In this way the future performance scenario shows which companies will continue to create sustainable value under the more stringent future performance targets (ADVANCE, 2006). In chapter 7 the weighted average return of resource of a large sample of dairy farms is used to explain differences in farm sustainability. The results of the analysis were also compared using other types of benchmarks, showing that the benchmark choice had an important impact on the absolute numbers of the sustainable value but not on the ranking of the sustainability performance of the farms (see section 7.4.3).

Because benchmarks can give valuable indications to all decision makers, a well defined benchmark is essential. Otherwise decisions support systems can give wrong signals, resulting in wrong decisions. Furthermore, it is important that a benchmark is realistic and feasible for each company, but it is also preferable that a benchmark is ambitious.

Benchmarks using best performance or performance targets can be very useful to analyze the efforts of farms in their aim to reach the targets or the best performance. In our example in table 8.1 we choose the weighted average return on resource of a large sample as benchmark. In chapter 7 we opted for this benchmark because we tried to understand why farms differ in their creation of sustainable value. Using for example the best performance of each resource as benchmark will result in other value contributions. In fact, all value contributions would be less than zero; a value contribution of zero would indicate that the observation is the best performance. In this case, the aim of all companies would be to get value contributions of zero. If all value contributions are zero, then the sustainable value of that company would be zero (or the return-to-cost ratio would be equal to one), which is the maximum achievable score. A sustainable value of zero would mean that the *super-company* exists or in other words that such a company has the highest productivity for all resources. Using a basic best performance benchmark, we found a maximum return-to-cost ratio of 0.7, showing a large scope for improvement (see section 7.4.3).

However, the basic best performance benchmark using the best performance of each resource has important shortcomings. As indicated by figure 8.1, such a basic benchmark is not necessarily the best option to assess the performance of companies. Using a basic benchmark for all companies (independent of the actual resource use and combination) can result in a misleading measurement of the resource performance of a company. The unit isoquant K in figure 8.1 shows all the ways of combining two resources X_1 and X_2 to produce a given level of output Y. Points on the unit isoquant are efficient because their actual productivity equals the maximum feasible productivity. Observation a can improve the productivity of resource X_1 while observation r has the maximum productivity level. In fact, it seems very clear that in this case observation r is a perfect benchmark for observation a (even for both resources X_1 and X_2), the peer of observation a is observation r. The productivity level of observation a for the resource use of X_1 equals $\frac{0X_1^r}{0X_1^a}$. However, when looking to observation c, the peer for observation c, using the basic best performance benchmark, would be observation r but with the actual combination of resources X_1 and X_2, this is not always a feasible target. Therefore, a better peer for observation c would be observation s (figure 8.1).

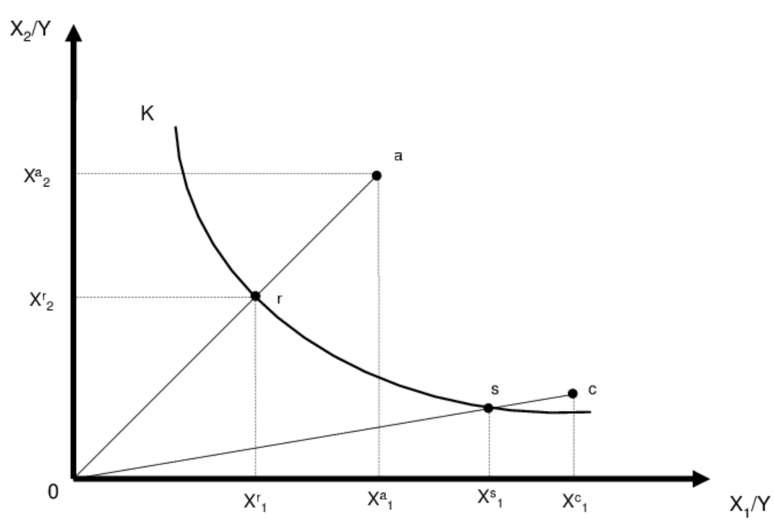

Figure 8.1: Unit isoquant K for resources X_1 and X_2 for a given level of output Y

To analyze the efforts of companies towards more sustainable practices, the use of a best performance benchmark within the calculation of the sustainable value of firms, is very promising. However, the basic best performance benchmark has major shortcomings and therefore using a benchmark as in figure 8.1 would be an important improvement to benchmark firm sustainability because in this case the value contribution of each resource is dependent of the use of the other resources and the sustainability of each company would be assessed compared to the relevant peers of that company. In applications, benchmark units (peers) can play an important role by facilitating diffusion of best practices from efficient units to inefficient ones (Kuosmanen and Kortelainen, 2005). Note that the focus in figure 8.1 (and in this research) is only on technical efficiency and not on allocative efficiency.

An important advantage of using *frontier method* benchmarks is that in this way the sustainable value approach takes production linkages into account. This is because production functions (estimated by frontier methods) show the link between the output produced and the resources used (including environmental and social resources). Therefore, in this chapter we will develop and test a methodology to improve the sustainable value method with frontier methods to construct a sound benchmark. Note however that different contexts require different benchmarks. In this application data driven benchmarks using frontier methods will be worked out. In certain applications other type of benchmarks (e.g. policy targets as benchmarks) could be useful to answer particular research questions.

8.3 Methodology

As indicated in figure 8.1, we suggest to use the sustainable value methodology and opt for a benchmark which (i) compares the combination of resources with other resource combinations and (ii) selects the most appropriate peer as benchmark for each company. This benchmark can be constructed using frontier methods. In this way, production theory is integrated with a value-orientated assessment method.

8.3.1 Formulation of the benchmark

In the frontier literature two broad classes of approaches are considered, namely the parametric and the non-parametric approaches. Parametric approaches (e.g., stochastic frontier estimations) take possible measurement errors and other noise upon the frontier into account. The disadvantage is that the researcher has to select a functional form for the production frontier. Non-parametric approaches are robust to the kind of specification error that may arise in the choice of functional form, but the properties of the inefficiency estimates cannot be determined. In this chapter we prefer to work with a parametric approach for estimating the production frontier, because in our empirical application farm data are used and we expect that data noise could play an important role in the estimation of an agricultural production function (Coelli et al., 1998). Note, however, that our approach is also compatible and operational with non-parametric approaches.

Consider the following production function:

$$\ln(y_{it}) = \alpha + f(x_{it}, \beta) + v_{it} - u_i \tag{8.1}$$

where y_{it} is the output of the i^{th} firm in year t; x_{it} are the input quantities in the production process used by the i^{th} firm in year t; v_{it} accounts for measurement error and random errors while the second error term u_i measures the technical inefficiency. The efficient amount of x_{it} can be expressed as:

$$x_i^{efficient} = g(y_i, x_1, ..., x_n, u_i) \tag{8.2}$$

In traditional production economics the inputs are for example labor and capital. The strategy of most parametric studies has been to include environmental effects in the output vector (e.g., Pittman (1983), Färe et al. (1989), Ball et al.

(1994), Hetemäki (1996)). As in Cropper and Oates (1992), Reinhard (1999) and Reinhard et al. (2000) we model the environmental assets as conventional inputs rather than as undesirable outputs, because this fits completely in the sustainable value approach. A second reason (also rather pragmatic) is the fact that environmentally detrimental input use is easy to measure (e.g., excess nitrogen use) which is not the case with environmental impacts (Reinhard, 1999). Nevertheless, we are aware that the question of whether environmental factors are inputs or outputs can be relevant e.g., with respect to returns to scale. This question has been recently debated by Färe and Grosskopf (2003) and Hailu (2003), but this discussion falls beyond the scope of this chapter.

We therefore specify the stochastic production frontier as:

$$\ln(VA_i) = \alpha + f(x_i, z_i, \beta) + v_i - u_i \tag{8.3}$$

for all companies indexed with a subscript i; VA_i denotes the value added; x_i is a vector of conventional economic inputs. Intermediate consumption is not considered as an economic input, because we choose the value added as output and not the total value of returns. z_i is a vector of environmental and social assets; v_i is a random error term intended to capture events beyond the control of the managers; u_i is a non-negative random error intended to capture technical inefficiency. The efficient amount of x_i and z_i can be expressed as:

$$\begin{aligned} x_i^{efficient} &= g(VA_i, x_1, ..., x_n, z_1, ..., z_n, u_i) \\ z_i^{efficient} &= g(VA_i, x_1, ..., x_n, z_1, ..., z_n, u_i) \end{aligned} \tag{8.4}$$

As mentioned in section 8.2.1, no distinction is made between conventional economic inputs (x) and environmental and social assets (z). We assume that they all contribute to the production of value added in a sustainable system. Therefore, we introduce the term resource r which includes economic, environmental and social capital forms (and aspects derived from capital forms):

$$r_i^{efficient} = g(VA_i, r_1, ..., r_n, u_i) \tag{8.5}$$

Remind that the sustainable value of a company with n different resources can be calculated as:

$$sustainable\ value_i = \frac{1}{n}\sum_{s=1}^{n} r_i \cdot [(\frac{VA}{r})_i - (\frac{VA}{r})_{benchmark}] \qquad (8.6)$$

where r_i stands for resource (economic, environmental and social) and VA_i for value added of company i.

Using efficiency analysis, we propose the following benchmark:

$$(\frac{VA}{r})_{benchmark} = (\frac{VA}{r_i^{efficient}}) = \frac{VA_i}{g(VA_i, r_1, ..., r_n, u_i)} \qquad (8.7)$$

Bringing equation 8.7 into equation 8.6 gives us the calculation of the sustainable value of a company i with an appropriate company specific benchmark:

$$sustainable\ value_i = \frac{1}{n}\sum_{s=1}^{n} r_i \cdot [(\frac{VA}{r})_i - (\frac{VA_i}{r_i^{efficient}})] \qquad (8.8)$$

Remark that the benchmark is different for each company, because the benchmark depends on the amount and combination of resources of that company (as in figure 8.1). To summarize, the benchmark calculation using frontier methods takes inefficiency of the resource use and substitution between the resources into account.

8.3.2 Formulation of the framework using functional forms

Before estimating the production frontier the researcher has to choose a functional form. An important step in any parametric empirical application is the selection of the appropriate functional form for the production function. A commonly used functional form is the Cobb-Douglas functional form. The simplicity of this functional form is very attractive, but a drawback is that the Cobb-Douglas production function assumes constant input elasticities, constant returns to scale for all firms and an elasticity of substitution to be equal to one. A number of alternative functional forms exist, such as the translog functional form (Christensen et al., 1973). An advantage of the translog form is that it imposes no restrictions upon returns of scale or substitution possibilities (Coelli et al., 1998). In the following sections, we use both forms.

8.3.2.1 Methodology using the Cobb-Douglas functional form

Assume a Cobb-Douglas technology with two resources r_1 and r_2 to produce VA (value added). Company i does not use his resources 100% efficiently, in other words u_i differs from zero.

We formulate the Cobb-Douglas stochastic production frontier model as:

$$\ln VA_i = \beta_0 + \beta_1 \ln r_1 + \beta_2 \ln r_2 + v_i - u_i \tag{8.9}$$

To perform the calculation, we first have to purge the output measure (VA) of its noise component (v_i) so that we can work in a deterministic framework:

$$\begin{aligned} \ln \widetilde{VA_i} &= \beta_0 + \beta_1 lnr_1 + \beta_2 \ln r_2 - u_i \\ with\ \ln \widetilde{VA_i} &= \ln VA_i - v_i \end{aligned} \tag{8.10}$$

We are looking for the input-orientated technically efficient resource $r^{efficient}$ for a given level of value added ($\widetilde{VA}$). This can be derived by simultaneously solving equation 8.10 and the resource ratio $\frac{r_1}{r_2} = k$. Remark that the solution of the simultaneous system of equation is made after the parameters of the production frontier have been estimated using maximum likelihood methods. After estimation we get:

$$\begin{aligned} \ln \widehat{VA_i} &= b_0 + b_1 \ln r_1 + b_2 \ln r_2 \\ \ln \widetilde{VA_i} &= \ln VA_i - v_i = \ln \widehat{VA_i} - u_i \end{aligned} \tag{8.11}$$

Note that $\widehat{VA_i}$ is the predicted frontier output and VA is the observed output. The $r^{efficient}$ are:

$$\begin{aligned} r_1^{efficient} &= [\widetilde{VA_i} \cdot \exp(-b_0) \cdot k^{b_2}]^{\frac{1}{b_1+b_2}} \\ r_2^{efficient} &= [\widetilde{VA_i} \cdot \exp(-b_0) \cdot k^{-b_1}]^{\frac{1}{b_1+b_2}} \end{aligned} \tag{8.12}$$

Bringing equations 8.12 into equation 8.8, we can calculate the sustainable value of company i using only 2 resources and assuming Cobb-Douglas technology as:

$$\begin{aligned} Sustainable\ value_i = & \frac{1}{2}\langle r_{i1} \cdot [(\frac{\widetilde{VA}_i}{r_{i1}}) - (\frac{\widetilde{VA}_i}{[\widetilde{VA}_i \cdot \exp(-b_0) \cdot k^{b_2}]^{\frac{1}{b_1+b_2}}})] \\ & + r_{i2} \cdot [(\frac{\widetilde{VA}_i}{r_{i2}}) - (\frac{\widetilde{VA}_i}{[\widetilde{VA}_i \cdot \exp(-b_0) \cdot k^{-b_1}]^{\frac{1}{b_1+b_2}}})]\rangle \end{aligned} \tag{8.13}$$

Because the Cobb-Douglas functional form has a constant elasticity of substitution (and equal to one), we can simplify the calculation of the sustainable value for company i as:

$$\begin{aligned} Sustainable\ value_i = & r_{i1} \cdot [(\frac{\widetilde{VA}_i}{r_{i1}}) - (\frac{\widetilde{VA}_i}{[\widetilde{VA}_i \cdot \exp(-b_0) \cdot k^{b_2}]^{\frac{1}{b_1+b_2}}})] \\ = & r_{i2} \cdot [(\frac{\widetilde{VA}_i}{r_{i2}}) - (\frac{\widetilde{VA}_i}{[\widetilde{VA}_i \cdot \exp(-b_0) \cdot k^{-b_1}]^{\frac{1}{b_1+b_2}}})] \end{aligned} \tag{8.14}$$

Note that the suggested benchmark offers two improvements. First, the benchmark incorporates inefficiency. Second, the benchmark allows substitution possibilities between the considered resources. In fact, each company can benchmark its resource use with the most appropriate peer.

In this case, we only considered technical (input-orientated) inefficiency. Remind that economic efficiency is the combination of technical efficiency and allocative efficiency. Assuming Cobb-Douglas technology, the economic efficiency input vectors can be calculated, because the Cobb-Douglas function is self-dual. For this, price information of each resource is needed, which is not always possible (and relevant) for all resources, especially for environmental and social aspects. Kopp and Diewert (1982) developed a method for decomposing the frontier cost deviations into measures of technical and allocative efficiency. Bravo-Ureta and Rieger (1991) developed a stochastic efficiency decomposition model based on Kopp and Diewert's deterministic methodology. Because the Cobb-Douglas production function is self-dual, the corresponding dual cost function can be derived and written as:

$$c_i = h(w_i, \widetilde{VA}_i; \delta) \tag{8.15}$$

where c_i is the minimum cost of the i^{th} firm associated with the value added of $\widetilde{VA}_i$; w^i is a vector of resource prices of the i^{th} firm; δ is a vector of parameters which are functions of the parameters in the production function. By using Shephard's Lemma, the economically cost minimizing vector $r^{totalefficiency}$ is derived by substituting the firm's input prices and output quantity into the system of demand equations (Singh et al., 2001):

$$\frac{\partial c_i}{\partial w_i} = r_i^{totalefficiency}(w_i, \widetilde{VA}_i; \delta) \tag{8.16}$$

8.3.2.2 Methodology using the translog functional form

In this section we develop the method when using a translog functional form. Assume a translog functional form with two resources r_1 and r_2 to produce VA (value added). Company i does not use its resources 100% efficient, in other words u_i differs from zero.

We formulate the translog stochastic production frontier model as:

$$\ln VA_i = \beta_0 + \beta_1 \ln r_1 + \beta_2 \ln r_2 + \beta_3 (\ln r_1)^2 + \beta_4 (\ln r_2)^2 + \beta_5 (\ln r_1 \cdot \ln r_2) + v_i - u_i \tag{8.17}$$

To apply the calculation, again we first have to purge the output measure (VA) of its noise component (u_i) so that we can work in a deterministic framework:

$$\begin{aligned} \ln \widetilde{VA}_i &= \beta_0 + \beta_1 \ln r_1 + \beta_2 \ln r_2 + \beta_3 (\ln r_1)^2 + \beta_4 (\ln r_2)^2 + \beta_5 (\ln r_1 \cdot \ln r_2) - u_i \\ &with \ \ln \widetilde{VA}_i = \ln VA_i - v_i \end{aligned} \tag{8.18}$$

We are looking for the input-orientated technically efficient resource $r^{efficient}$ for a given level of value added $(\widetilde{VA}_i)$. This can be derived by simultaneously solving equation 8.18 and the resource ratio $\frac{r_1}{r_2} = k$. Remark that the solution of the simultaneous system of equation is made after the parameter of the production frontier have been estimated using maximum likelihood methods. After estimation we get:

$$\begin{aligned} \ln \widehat{VA}_i &= b_0 + b_1 \ln r_1 + b_2 \ln r_2 + b_3 (\ln r_1)^2 + b_4 (\ln r_2)^2 + b_5 (\ln r_1 \cdot \ln r_2) \\ \ln \widetilde{VA}_i &= \ln VA_i - v_i = \ln \widehat{VA}_i - u_i \end{aligned} \tag{8.19}$$

Note that $\widehat{VA}_i$ is the predicted frontier output and VA is the observed output. The $r^{efficient}$ are:

$$r_1^{efficient} = exp[\frac{-(b_1 + b_2) \pm \sqrt{(b_1 + b_2)^2 - 4(-\ln \widetilde{VA}_i + b_0 - (b_2 + b_5) \cdot \ln(k) + b_4 \cdot (\ln k)^2) \cdot (b_3 + b_4 + b_5)}}{2 \cdot (b_3 + b_4 + b_5)}]$$

$$r_2^{efficient} = exp[\frac{-(b_1 + b_2) \pm \sqrt{(b_1 + b_2)^2 - 4(-\ln \widehat{VA_i} + b_0 - (b_1 + b_5) \cdot \ln(k) + b_3 \cdot (\ln k)^2) \cdot (b_3 + b_4 + b_5)}}{2 \cdot (b_3 + b_4 + b_5)}] \tag{8.20}$$

Once this is obtained, the same approach as in the Cobb-Douglas case can be followed by bringing the $r^{efficient}$ (equations 8.20) for every resource into equation 8.8. In this way the sustainable value can be calculated.

8.4 Empirical applications

In this section, the considered methodology is applied using empirical data. First, the Cobb-Douglas functional form is used as benchmark to calculate the sustainable value. Second, the translog functional form is used as benchmark to calculate the sustainable value. Finally, the impact on the sustainable value will be estimated for different policy options to illustrate how the approach may be used as a decision support system.

8.4.1 Cobb-Douglas functional form as benchmark

The first application uses the data of a large sample of Flemish dairy farms. As in chapter 7, we consider five different resources: (i) farm labor, (ii) farm capital, (iii) farm land, (iv) nitrogen surplus and (v) energy consumption (direct and indirect). Capital, land and labor can be seen as traditional economic resources, while nitrogen surplus and energy consumption are important environmental aspects in dairy farming. The dataset contains information of 645 Flemish dairy farms during the period 1995-2001. Some descriptive statistics of the data sample can be found in table 8.2.

Table 8.2: Descriptive statistics

Variable	Mean	Minimum	Maximum	Std. Dev.
Total output (Euro)	150 293	20 445	622 791	68 765
Land use (hectares)	31.73	6.72	83.08	11.28
Labor (full-time equivalent)	1.48	0.63	3.50	0.34
Farm capital (Euro)	284 466	37 338	789 404	152 140
Intermediate consumption (Euro)	66 361	13 600	295 465	31 535
Energy consumption (MJ)	1 248 410	268 185	3 803 592	522 292
Nitrogen surplus (kg N)	8884	1934	25 570	3879

As explained in section 8.2.1 we use conventional economic and environmentally detrimental resources to estimate a production function. As dependant variable the value added of the farms is used. Furthermore, time dummies are added to indicate the different years. This leads to the following Cobb-Douglas functional form:

$$
\begin{aligned}
VA_i = exp(\beta_0) \cdot Labor_i^{\beta_1} \cdot Capital_i^{\beta_2} \cdot Land_i^{\beta_3} \cdot N - surplus_i^{\beta_4} \\
\cdot Energyconsumption_i^{\beta_5} \cdot exp(\sum_{j=1}^{n} \gamma_j \cdot Yeardummy_i^j) \cdot exp(v_i - u_i) \qquad (8.21)
\end{aligned}
$$

We can rewrite equation 8.21 in logarithmic form as:

$$\ln VA_i = \beta_0 + \beta_1 \ln(Labor)_i + \beta_2 \ln(Capital)_i + \beta_3 \ln(Land)_i + \beta_4 \ln(N - surplus)_i + \beta_5 \ln(Energyconsumption)_i + \sum_{j=1}^{n} \gamma_j \cdot Yeardummy_i^j + v_i - u_i \quad (8.22)$$

The estimation results of equation 8.22 using maximum likelihood methods can be found in table 8.3.

Table 8.3: Estimation coefficients of the Cobb-Douglas production frontier

Variables	Coefficient	St.error	Variables	Coefficient	St.error
Constant	0.5297	0.4344	D-1995	0.0057	0.0358
Labor	0.2886***	0.0510	D-1996	-0.0519	0.0340
Farm capital	0.2496***	0.0220	D-1997	0.0602*	0.0355
Farm land	0.2184***	0.0479	D-1998	0.1757***	0.0398
N-surplus	-0.1828***	0.0462	D-1999	0.3842***	0.0414
Energy-consumption	0.6147***	0.0545	D-2000	0.1545***	0.0356
Number of observations		645	Iterations completed		20
Sigma		0.3975			

* significant at 10%; **significant at 5%; ***significant at 1%

We apply the methodology as explained in section 8.3.2.1. First, the output measure is separated from its noise component to be able to work in a deterministic framework (as in equation 8.10). Then, we calculate the input-orientated technically efficient resource for each resource considered using the estimated coefficients of equation 8.22 and the resource ratios. After we obtained the input technically efficient amount of each resource for each company, the sustainable value can be calculated using those values as benchmarks. Note that in this application the technical input-orientated efficiency is used. Farms can improve their efficiency by reducing their amount of resources and producing the same amount of output (value added). Flemish dairy farms have milk quota and have to pay super levies in the case of exceeding their milk quota. Farms have to obtain extra milk quota if they want to increase their production level. This makes the choice for an input-orientated efficiency approach the most appropriate.

We illustrate this using one of the observations in our dataset. This farm uses five resources to produce a value added of 146 448 Euro. Correcting this for random errors (in other words subtracting v_i) the value added becomes 149 283 Euro. The actual use as well as the technical efficiently use of the resources is calculated in table 8.4.

In our example the farm uses for example 50 ha of land, while the same amount of value added could be produced using only 41 ha agricultural land (table 8.4). Note that the ratio of the actual use to the technical efficient use of the resources

Table 8.4: Actual and technical efficiently resource use of a sample farm for achieving a value added of 149 283 Euro

Resource	Actual use (r)	Technical efficient use ($r^{efficient}$)
Labor (fte)	1.50	1.23
Farm capital (Euro)	244 039	200 024
Farm land (ha)	50.09	41.09
N-surplus (kg N)	13 308	10 908
Energyconsumption(MJ)	1 950 770	1 598 926

Fte= full time equivalent

is the same for all resources. This is due the choice of the Cobb-Douglas formulation as functional form. As already mentioned the Cobb-Douglas functional form has an elasticity of substitution equal to one.

The sustainable value of all observations of the dataset can be calculated using the input efficient resource use as benchmark. In figure 8.2 the sustainable value of all our observations from low sustainable value to high sustainable value is represented.

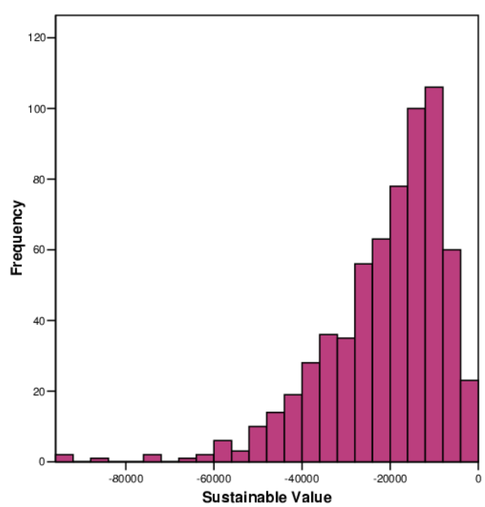

Figure 8.2: The frequency of the sustainable value of all observations

It is quite obvious that for all farms the sustainable value is negative. In fact, a sustainable value of 0 would indicate that the farm uses all its resources in the most productive way. Such a super farm does not exist in our sample. Nevertheless, large differences are observed ranging from dairy farms with a sustainable value of -2000 Euro till -94 000 Euro. Farms can improve their sustainable value by applying their resources in a more productive way, in other words, by moving towards the production frontier. However, farms can improve their sustainable value by replacing more sustainable value creating resources by resources with low value contributions. The value contributions of all resources are equal using the Cobb-Douglas functional form as benchmark, because of the constant elasticity of substitution. Hence, using the Cobb-Douglas functional as benchmark cannot identify substitution possibilities because the assumption of constant elasticity of substitution. That is the reason why all value contributions of all resources are equal.

Figure 8.3 shows the evolution of the sustainable value and the return-to-cost of the dairy farms in the data sample between 1995 and 2001. Note that we used in this case a balanced panel data sample, in other words only the farms with data for all seven consecutive years (1995-2001) are used in figure 8.3. The average sustainable value of the farms fluctuates between -18 000 Euro and -23 000 Euro, except in 1999. In 1999 the average sustainable value of our dairy farms was over -26 000 Euro. As already explained, the sustainable value calculations do not take the farm size into account. Therefore we use a size independent ratio: the return-to-cost ratio. The return-to-cost ratio relates the value added created by a farm to the opportunity costs it causes. The average return-to-cost is calculated as the sum of the return-to-cost ratios of all observations in one year divided by the number of observations in that year. Using the Cobb-Douglas production frontier as benchmark, a maximum return-to-cost of 0.96 has been found. The minimum return-to-cost of an observation in our datasample is 0.51. That farm uses his resources only half as productive as the benchmark (the maximum attainable production). We do not observe large yearly average return-to-cost shifts (figure 8.3). Remark that in this case a low average sustainable value certainly does not mean a low average return-to-cost ratio, moreover the reverse is true. For example, in 1999, we observe a low average sustainable value and a high average return-to-cost ratio in comparison with the other years. This is not very surprising knowing that the average value added in 1999 was high (resulting in a high return-to-cost). Note that although the productivities of the different resources in 1999 were in general higher in comparison with other years, the benchmark productivities were also higher, because the farms could achieve higher productivities due to beneficial circumstances (e.g., weather conditions). This can result in lower sustainable values for the farms in 1999.

As indicated by figure 8.1, we suggest using a frontier benchmark instead of using a simple best performance benchmark. In chapter 7 different benchmark types were used to analyze the robustness of the result. The rank correlation between the return-to-cost ratio using the weighted average return on resource as benchmark and the return-to-cost ratio using the basic best performance on each resource form as benchmark was very high (spearman's rho = 0.9967). The use of a feasible benchmark for each company (applying frontier meth-

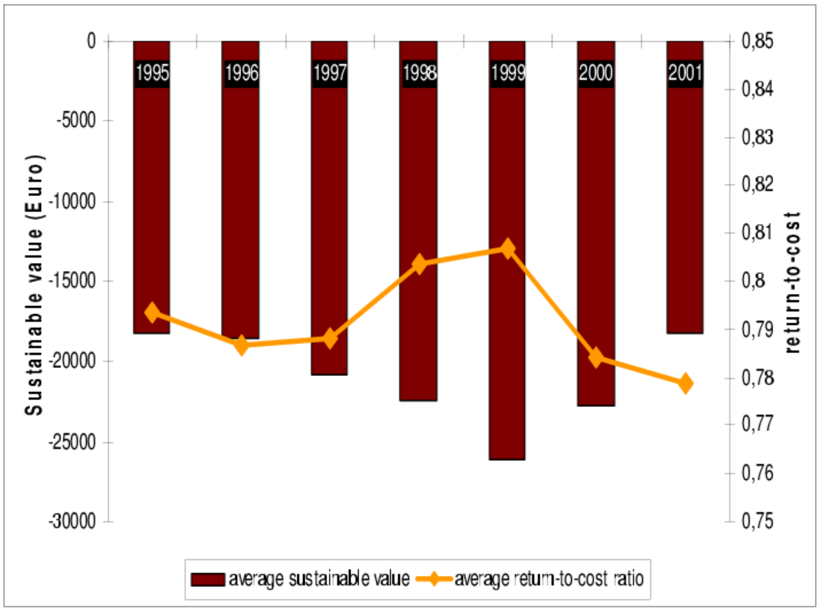

Figure 8.3: The evolution of the average sustainable value and return-to-cost ratio of Flemish dairy farms

ods) results in a different ranking. We found a much lower rank correlation (spearman's rho = 0.2327) between the return-to-cost ratio using the simple performance on each resource as benchmark and the return-to-cost ratio using a Cobb-Douglas production frontier as benchmark. This confirms our point that the sustainable value approach can differ by using frontier methods to benchmark the resource use of companies. The benchmark using frontier aspects takes underlying production aspects (e.g., substitution possibilities between resources) into account. Hence, each farm is compared with a realistic but ambitious peer. That is why we call this *frontier* benchmark approach more complete than the *basic* benchmark approach.

8.4.2 Translog functional form as benchmark

Important drawbacks of the Cobb-Douglas functional form are the restrictive properties. The Cobb-Douglas functional form is easy to estimate but it has constant input elasticities, a constant return to scale and a substitution elasticity equal to unity. The translog functional form does not impose these restrictions upon the production structure, it is a more flexible functional form. But this is at the expense of having a form which is more difficult to estimate and which can suffer from degrees of freedom and multicollinearity problems (Coelli et al., 1998). Using for example five different resources as in section 8.4.1 would result in a production function with 21 variables. There are a lot of observations needed to estimate such an equation. Estimations with only 645 observations (as in section 8.4.1) were inadequate. Therefore, we will use an extended data sample (2651 observations) with only two resources (farm labor and farm capital). We will use only economic resources because in our extended data sample information about environmental resources was not yet available. Our data sample contains 2651 observations of Flemish dairy farms during the 1989-2002. Note that in this example farm capital includes land capital.

In this case the translog functional form can be written as equation 8.17:

$$\ln VA_i = \beta_0 + \beta_1 \ln(Labor)_i + \beta_2 \ln(Capital)_i + \beta_3 (\ln(Labor)_i)^2 + \beta_4 (\ln(Capital)_i)^2 \\ + \beta_5 (\ln Labor_i \cdot \ln Capital_i) + \sum_{j=1}^{n} Yeardummy_i^j + v_i - u_i \tag{8.23}$$

Table 8.5: Estimation coefficients of the translog production frontier

Variables	Coefficient	St.error	Variables	Coefficient	St.error
Constant	9,2995***	0,1262	$D-1994$	0,1641***	0,0350
Labor	1,0696***	0,1802	$D-1995$	0,1156***	0,0346
Capital	0,3573***	0,0849	$D-1996$	0,0559	0,0358
$Labor^2$	0,2141**	0,1046	$D-1997$	0,1582***	0,0359
$Capital^2$	0,0448***	0,0155	$D-1998$	0,3453***	0,0395
$Labor \cdot Capital$	-0,2125***	0,0634	$D-1999$	0,5452***	0,0411
$D-1990$	-0,1018***	0,0298	$D-2000$	0,2681***	0,0373
$D-1991$	-0,0654**	0,0303	$D-2001$	0,1693***	0,0433
$D-1992$	0,0155	0,0323	$D-2002$	0,0998**	0,0390
$D-1993$	0,2577***	0,0347			
Number of observations		2651	Iterations completed		28
Sigma		0.5179			

* significant at 10%; **significant at 5%; ***significant at 1%

The estimation results of equation 8.23 using maximum likelihood methods can be found in table 8.5. We apply the methodology as explained using a two resource example in section 8.3.2.2. First we separate the output measure with its noise component to work in a deterministic framework (as in equation 8.18). Then we calculate the input-orientated technically efficient resource for each resource considered using the estimated coefficients of equation 8.23 and the resource ratio. After we obtained the efficient resource amount to produce the value added for each resource and for each company, the sustainable value can be calculated using those values as benchmarks.

This can be illustrated using one of the observations in our dataset. This farm use two resources to produce a value added of 76 949 Euro. Correcting this for random errors (in other words subtracting v_i) the value added becomes 67 602 Euro. The actual use and the technical efficient use of the resources can be found in table 8.6.

Table 8.6: The actual and technical efficient resource use of the a sample farm for achieving a value added of 67 602 Euro

Resource	Actual use (r)	Technical efficient use ($r^{efficient}$)
Labor (fte)	1.55	0.88
Total farm capital (Euro)	298 571	225 942

Fte= full time equivalent

In our example the farm uses for example 1.55 full time equivalent (fte) units of labor, while the farm could create the same amount of value added using only 0.88 fte of labor (table 8.6). Note that the relation between the actual use to the technical efficient use of the resources is not the same for all resources (in contrast with the Cobb-Douglas functional form). Our results indicate that farms use labor less efficiently than capital. Mark that our analysis does not take allocative efficiency into account. In other words the prices of the inputs are not considered.

The sustainable value of all observations of the dataset can be calculated using the input efficient resource use as benchmark. In this case the (negative) impact of labor capital will be higher than the (negative) impact of total farm capital in the calculation of the sustainable value (see figure 8.4). Farm capital is used in a more value-creating way (in fact a less value-wasting way) than labor capital. Figure 8.4 shows the average value contributions of farm capital and labor and the average sustainable value of a balanced panel set of Flemish dairy farms (55 dairy farms during 1989-2002). We observe a decrease in sustainable value till 1999. Starting from 1999 we see a rather limited increase in sustainable value creation.

Farms can improve their sustainable value by applying their resources in a more productive way. They can increase their technical efficiency by moving towards the production frontier. On the one hand, farms can decrease the amount of

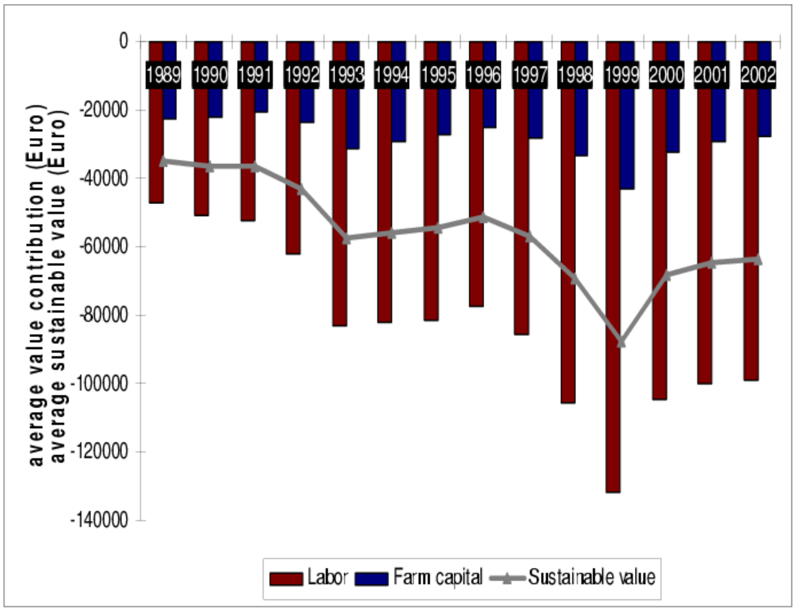

Figure 8.4: the evolution of the average value contributions and sustainable value of Flemish dairy farms

resources used while producing the same amount of output. On the other hand, farms can change the composition of resources, value-wasting resources can be partly substituted by value-creating resources (or less value-wasting resources). The sustainable value methodology using the translog production frontier as benchmark considers both possibilities. In other words substitution effects between resources are clearly taken into account to determine the opportunity cost (or benchmark) of each resource. A major drawback are the data requirements to estimate the translog production frontier (a lot of observations are needed). The more resources that are considered as critical to assess firm sustainability the more data is needed.

8.4.3 Benchmarking sustainability assessment for policy evaluation

The sustainable value approach shows how a scarce amount of resources should be used in order to generate high returns. It is interesting to know which firms are creating value considering economic, environmental and social resources, but it is even more crucial to know the impact of (future) decisions on sustainable value. If a company or policy maker has to choose between several options, it is important that in terms of sustainable development that option is selected that increases the sustainable value of the company, sector or region. In this section, we present how the suggested approach may be used to support policy making. We illustrate the approach for the Flemish dairy sector using a large accountancy data sample (see section 8.4.1). Assume that policy makers consider to improve the sustainability performance of the Flemish dairy sector based on the two following policy options. The first option is to provide subsidies to improve the energy use (direct and indirect) of dairy farming (e.g., decrease in concentrate use or electricity use). Assume that these measures will result in an average decrease of 10% energy use while the value added remains the same. The second option is to provide subsidies to invest in labor saving techniques (e.g., time management tools, removing administrative burden). Again we assume that these measures will result in an average decrease of 10% labor use while the value added remains the same. Because policy makers have a limited budget, they have to choose between option A (energy use decrease) or option B (labor use decrease).

To support policy makers, the sustainable value (and return-to-cost ratio) of both options can be simulated. To do so, we use the balanced panel data of dairy farms as in figure 8.3, and we simulate the sustainable value of every farm in the sample for a future year for three options: option A, option B and the base scenario. We use the estimated Cobb-Douglas functional form as benchmark. The base scenario is a simulation of the sustainable value without a policy intervention (business as usual). As in section 8.4.1 five different resources are selected: labor, farm capital, farm land, energy consumption and N-surplus. The resource use is calculated as the average of the seven preceding years. Furthermore, the value added and the yearly variation (indicated by the coefficients of the year dummies in table 8.3) are fixed on the average values of the preceding years. To calculate the impact of the options, the energy use and labor use are decreased with 10% compared to the calculated average (or base scenario) for option A and option B respectively.

The simulation results can be found in figure 8.5. As expected (given the assumptions) the average sustainable value and the return-to-cost ratio increase for the two options. More interesting is the fact that subsidizing a decrease in energy use results in a higher increase of sustainable value than subsidizing a decrease in labor use. In other words, these results suggest policy makers to support energy use reduction instead of labor use reduction.

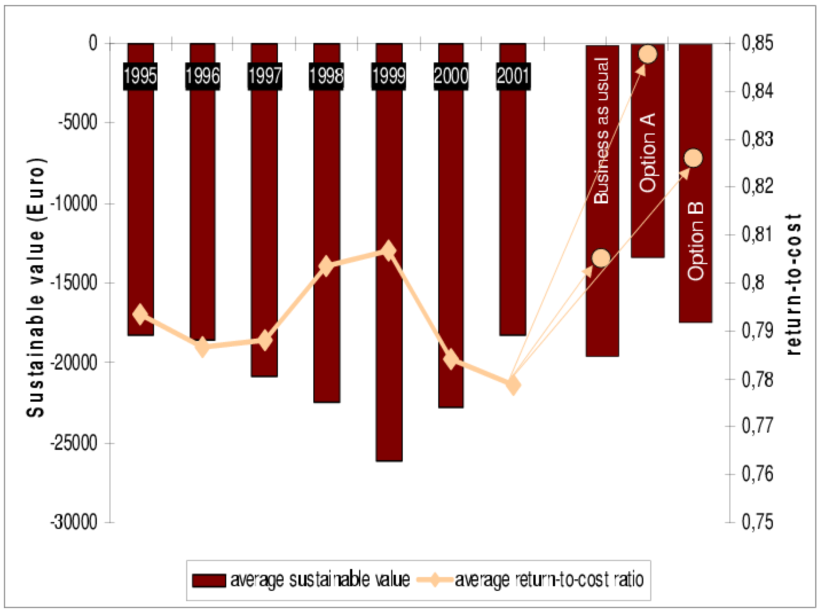

Figure 8.5: The evolution of the average sustainable value and return-to-cost ratio of Flemish dairy farms including the simulation results of the policy options (business as usual, option A: energy use decrease; option B: labor use decrease)

Furthermore, we can analyze the simulation results considering characteristics of the farm manager. Table 8.7 shows that the return-to-cost ratio is higher for young, educated farmers with certainty about their succession. Furthermore, we found in each case a similar trend as in figure 8.5: option A is preferred over option B which is better than business as usual.

We are aware of the simplicity of the suggested policy options. To support policy makers, the suggested options have to be refined in more detail (e.g., differentiating among farmers receiving a subsidy). Further, the impact of the suggested policy measures on all different resources and on the value added must be studied and estimated before incorporating these results within the sustainable value approach. Our assumption of equal value added while decreasing the energy or labor use is for example not very realistic. Nevertheless, these results show that the suggested approach can be very useful to support decisions of policy makers and company managers and that the impact of potential decisions can be evaluated within an integrated sustainability framework.

Table 8.7: Average return-to-cost considering managerial farm characteristics for the different policy options

	Return-to-cost ratio Bussiness as usual	Return-to-cost ratio Option A: energy use decrease	Return-to-cost ratio Option B: labor use decrease
Education of farmer			
no education (34%)	0.766	0.809	0.786
education (66%)	0.826	0.872	0.847
Age of farmer			
young ($\leq$ 39 year) (34%)	0.814	0.860	0.835
middle (40-46 year) (37%)	0.803	0.848	0.824
old ($\geq$46 year) (29%)	0.798	0.842	0.818
Succession of farmer			
no successor (37%)	0.811	0.857	0.832
doubt about succession (59%)	0.797	0.842	0.818
successor (5%)	0.858	0.906	0.880

Number of dairy farms: 41

8.5 Conclusion and discussion

The performance of companies is usually defined in terms of return to capital and profit. Recently, the view on performance has been broadened. To create value, companies do not only need economic capital but also environmental and social resources. This means that all relevant firm resources should be considered when assessing firm performance. In this broad view, high performance is similar to improved sustainability.

Different assessment tools have been developed to assess firm sustainability. An interesting approach is the one developed by Figge and Hahn (2004*a*) and Figge and Hahn (2005), who apply a value-orientated methodology to calculate the cost of sustainable resources. Their approach is based on the notion of strong sustainability, because it assumes that the amount of each resource remains unchanged on the macro level (Figge and Hahn, 2005). This means that firm performance is analyzed as a scale issue rather than as the optimal efficient allocation of resources. The approach considers the total amount of resources rather than just the change in resource use. Thus, the sustainable value approach introduces scale-sensitivity into the performance analysis. Note that the sustainable value approach does not determine the optimal sustainable scale of (economic) activity. The sustainable value approach starts from the constant resource rule: it leaves the amount of resources unchanged (Figge and Hahn, 2005), but an unchanged amount of resources does not mean an optimal amount of resources.

In chapter 7 the sustainable value and the return-to-cost ratio of a large sample of Flemish dairy farms was calculated and differences in the return-to-cost ratio were detected and explained. For this, the weighted average return on resource was chosen as benchmark. However, within the scope of policy analysis the choice of a more accurate benchmark is important, because for policy makers a benchmark indicating the maximum attainable productivity level is more useful to analyze the efforts of firms in their aim towards best performance.

Correct benchmarking is important for the following reasons. First, improvement in eco-efficiency (as measured by the sustainable value approach) is often the most cost-effective way of reducing environmental pressures (Kuosmanen and Kortelainen, 2005). Efficiency improvements can be seen as the first important step towards sustainability. Therefore, it makes economic sense to exploit these options as much as possible. Second, policies targeting efficiency improvements tend to be more easily adopted than policies that restrict the level of economic activity (Kuosmanen and Kortelainen, 2005).

Our approach combines the sustainable value approach, which can be seen as an indicator for eco-efficiency, with the frontier approach to benchmark the possible improvement. Using maximum feasible production possibilities as benchmark offers several advantages. First, substitution possibilities between different resources (economic and environmental) are not ignored as in traditional eco-efficiency analysis. Second, the constructed benchmark takes inefficiency of the considered resources into account. Third, using frontier methods to construct benchmarks provides specific benchmarks for each company adjusted to the particular situation of the company (in other words to the actual resource use). The sustainability of each company is assessed in comparison with its relevant peers. Feasible targets can help to motivate decision makers (managers (e.g., farmers) and policy makers) to take realistic but ambitious measures towards sustainability. Fourth, our approach can be used to simulate and estimate the impact on firm sustainability of possible policy measures. In this way, this method can be used as an integrative sustainability assessment tool for policy measures. The main limitation of the suggested method is its extensive data requirement. As in Kuosmanen and Kortelainen (2005) our method is based on relative efficiency assessment of comparable units. Hence, data must be accurate and reliable, and the sample size must be sufficiently large.

The methodology using frontier methods to benchmark the sustainable value of firms has been illustrated with two functional forms: the Cobb-Douglas and the translog functional form. The Cobb-Douglas functional form is very attractive because it is easy to estimate and to interpret. However, a major drawback of using the Cobb-Douglas functional form is the lack of flexibility. The value contributions of the different resources are identical because of the fixed elasticity of substitution (equal to 1). A possible solution is using the translog functional form which is more flexible and allows to take substitution between resources into account. Our example for Flemish dairy farms shows that labor is used less productively than farm capital. Farm capital is used in a more value-creating way, or better in a less value-wasting way, than labor capital. A disadvantage of the use of the stochastic translog functional form is the data requirement, as many data are needed to avoid estimation problems such as multicollinearity problems.

The described methodology seems very promising to assess system sustainability and can also be used to support policy makers. The sustainable value approach has the advantage not to look from the negative externality point of view but from the value added point of view. Therefore, we think that our methodology may be a powerful tool, not only to assess firm sustainability, but also to guide companies towards sustainability. Starting from the available resources and looking at their contribution to the value added of a firm, the dependencies and possible substitutions in the resource base are analyzed without reducing the economic output. This makes the options for sustainability improvement more concrete, interesting and realistic for both firm managers, seeking a private economic optimum, and for policy makers, seeking a more social welfare optimum.

To make the sustainability assessment tool fully operational, more research is needed such as testing other techniques to estimate the frontier (e.g., data envelopment analysis, goal frontiers) or other functional forms. Furthermore, the possibilities for both back casting as forward casting, taking into account the impacts of different policy instruments, should be explored. Our rather simplistic example shows that this is possible but that this approach should be further expanded based on more detailed information about costs and benefits of the suggested policy options. Finally, besides the choice of the benchmark, we have to determine the scope of the analysis and we have to incorporate the relevant resources to calculate the sustainable value of a company. So far, the relevant resources were based on literature and the availability of data. But with the increased collection of data on several environmental and social aspects (e.g., CO_2 contribution, animal welfare) the scope for further research analysis will certainly become wider.

Conclusions

Chapter 9

General discussion, conclusions and perspectives

Unless we change direction, we are likely to end up where we are going

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Farming activities are inseparably connected with environmental and social concerns. Therefore, one of the guiding principles of agricultural policy is to aim for strong economic performance that goes hand in hand with the sustainable use of natural resources. Sustainability can be seen as a key element towards a profitable long-term future for farming and rural areas. Nevertheless, without a clear theoretical framework and without empirical applications policy objectives may be never put into practice. It is important to measure and evaluate progress towards sustainability to support decision takers (farmers, policy makers,...) in their aim for a sustainable and efficient farming sector. Without integrated information on sustainability problems, awareness of these issues will be limited and the formulation and monitoring of responses will be difficult. In other words, it is not only essential to recognize the importance of sustainability, but also the need to measure progress towards sustainability.

The first part of this dissertation gave an overview of the different notions, interpretations and measurement methods of sustainability performance to avoid common misunderstanding about the meaning of sustainability. Knowing the very different notions, underlying assumptions and methodologies to target sustainability is essential before analyzing contributions towards sustainability in practice.

In the second part of this dissertation, several empirical applications about measuring performance have been presented. The first applications measured performance in a more traditional economic way, while the following applications also integrated environmental aspects to measure the sustainability performance of agricultural firms. Sustainable agriculture can be reflected by various economic, social and environmental factors that are closely interconnected with each other and furthermore, farms could and should play an important part in the attainment of sustainability goals.

9.1 Conceptual issues: defining and assessing sustainability

The sustainability debate is obscured by a number of misunderstandings. Two major problems can be distinguished. First, several different notions of sustainability exist. Second, there is a clear discrepancy between theoretical and practical sustainability.

9.1.1 Different notions of sustainable development

There are many and very different definitions of sustainability. Hence, it is clear that there is no universally agreed definition that can be applied at all times and places. But, in a very diverse world the variety of definitions, meanings and interpretations can also be seen as an advantage. A single definition that attempts to capture the diversity of sustainable development would be impractical. In fact, the flexibility of the meaning of sustainability is a major advantage as sustainability draws much of its power and creativity from its ambiguity. Sustainability is certainly not meaningless, it is just inevitably vague. However, this means that a decent description and discussion is needed in each research or policy document. In this way, the underlying assumptions and notions become clear to the reader; otherwise, a sound discussion is made impossible.

The following aspects mostly return within the sustainability debate: (i) natural resources are finite and there are limits to the carrying capacity of the Earth's ecosystems, (ii) economic, environmental and social goals must be pursued within these limits, and (iii) there is a need for inter- and intragenerational equity. Sustainability is frequently defined with emphasis on the multidimensionality (economic, social and environmental dimension), situated on different levels (e.g., national and firm level), time scales and systems (e.g., agricultural systems). Another approach is the focus on the dynamic, evolutionary process of change towards sustainability. In this view, (sustainable) efficiency can be seen as a necessary but not sufficient step towards sustainability.

A well-known description of two different notions of sustainable development is the distinction between weak and strong sustainability. The weak sustainability view states that the substitutability of human-made resources for natural resources is more or less unlimited. Furthermore, proponents of weak sustainability accept that technology makes it possible to exceed the material limits of natural resources or to substitute resources if they are depleted or if productivity limits are reached. This technological optimism is rejected by proponents of the strong sustainability view. Advocates of strong sustainability find that the perfect substitution argument violates the law of the conservation of matter. In other words, proponents of both weak and strong sustainability have a different view (i) on the substitution possibilities between resources and (ii) on the possibilities of technological progress. In fact, both views are non-falsifiable under scientific standards because they rest on assumptions and claims about the future that are non-refutable. This certainly does not mean that the discussion about substitution between resources and technological change is meaningless. In contrast, the notions of weak and strong sustainability are very useful to place empirical applications and to discuss the underlying assumptions of empirical work. Note that besides the notions of weak and strong sustainability, other notions can also be of great interest as theoretical framework, for example inter- and intragenerational equity frameworks or the transition framework.

9.1.2 Theoretical versus practical sustainability

An important step is to move from trying to define sustainability towards developing concrete tools for measuring and promoting achievements in sustainability. In fact, this means that sustainability has to be defined in considerably narrower terms in order to establish operational rules. Hence, sustainability assessment is inevitably based on strong simplifications both of the theoretical paradigm and of the characteristics of systems of concern. Several approaches can be distinguished to integrate the sustainability dimensions: (i) a set of indicators (visual integration) and (ii) a single, composite index or a limited amount of aggregated indicators (numerical integration). Money can be used as a common unit to integrate sustainability indicators. These purely monetary assessment approaches are useful for documenting core aspects of weak sustainability, while (bio)physical approaches are more useful for tracking strong sustainability. Not all assessment methods can be easily classified because there exist several methods combining monetary and biophysical approaches.

Weak sustainability measures attempt to put a monetary value on environmental services to internalize the externalities into the budgets of nations, households or enterprizes. These measures can be divided into two major groups: flow-based measures and stock-based measures. Flow-based measures attempt to adjust net national product to transform it into an indicator of sustainability, a well-known example is the index of sustainable economic welfare (ISEW). Stock-based measures are grounded in the concept of natural and man-made capital stocks. A well-known example is the genuine savings (GS) approach. The GS approach is based on a theoretical framework (the Hartwick rule), but in fact it is only an indicator of unsustainability: negative GS rates indicate unsustainability while positive GS rates do not indicate sustainability. In contrast, there is no theoretical framework to underpin the ISEW. On the other hand, the ISEW is one of the exceptions that do measure the problem of equitable distribution. A general remark is that GS and ISEW studies give no support towards sustainability policies, because they give no answers to the question of how to reach sustainability. Both approaches share many of the methodological problems, but several improvements are possible.

Strong sustainability measures are more diverse than weak sustainability measures, as a consequence many rules have been suggested to operationalize strong sustainability. There are two different main schools of thought. One assumes no substitution between natural resources and other resources and unlimited substitution between forms of natural resources (e.g., the ecological footprint approach), while the other requires a subset of total natural resources to be preserved in physical terms so that its functions remain intact. This is described as the critical natural capital approach and examples are the material flows and the sustainability gaps.

Besides the measurement of sustainability at national level, sustainability can also be analyzed at firm level. Sustainability measurement at firm level provides useful information for firm management and to support the decision making process. There is a wide range of methods such as indicator approaches, life cycle assessment, and (eco)-efficiency analysis. Note that also aggregate firm indicators (e.g., eco-efficiency) are useful to complement existing measures and to highlight progress towards sustainability on sectoral, regional or national level.

From the overview of the different methods to measure sustainability, four main conclusions can be formulated. First, many different methods exist and each method has its own pros and cons. Second, the majority of the sustainability assessment methods have recently been developed and methodological improvements are possible and often necessary. Third, there is a growing amount of empirical applications using the developed methods to assess sustainability but more empirical applications are still necessary. The end use validation of methodologies reflects the fact that the methods are effectively used and that the methods can be useful as decision aid tool. Any meaningful analysis of sustainability needs to pay attention to indicate which concepts and assumptions are used in the measurement methodology. Fourth, it is recommended to use a combination of methods, for example using both biophysical and monetary assessment approaches. The choice of assessment method depends on the exact research question, the underlying notions and assumptions, and the level considered (e.g., nation, industry or company level).

9.2 Methodological issues

9.2.1 Performance

A key question of this dissertation is *how to measure performance*. Performance can be seen as the way in which someone or something functions, operates or behaves. In the empirical applications the performance of farms was considered. A common way to measure performance in economics is to use the notion of efficiency. Under this approach, a firm has a high efficiency if it realizes a maximum attainable output considering the (economic) resources used. Such an efficiency indicator measures economic performance. However, sustainability performance is defined by the integrated achievement of social, environmental and economic performance. In this way, a firm has a high sustainable efficiency if it realizes a maximum attainable output considering all the resources used (economic, environmental and social resources). An integrated measurement of performance is thus a *conditio sine qua non* to measure sustainability and to achieve more sustainability. This certainly does not mean that the measurement of economic performance is unnecessary. Measuring economic performance is still useful to support decision makers in their aim to improve economic results. However, it is important to realize that an integrated measurement is necessary to have a more complete picture of the performance of nations or firms. Note that while measuring economic performance is complex on its own, measuring sustainability performance is both complex and controversial. Indeed, the development of many sustainability assessment methods still are in their infancy.

Assuming that efficiency improvement is a first step towards higher performance, we applied several efficiency indicators to measure performance. In the first two applications, economic performance is measured using technical efficiency as indicator for performance. Measuring efficiency is thus seen as a consistent way of monitoring (economic) farm performance. Techniques such as the stochastic frontier analysis to measure efficiency are reliable and nowadays widely used. Integrating environmental and social considerations into the calculation of economic performance is essential to measure sustainability performance. There exist different ways to do so. Thereby, the question is not which is the best method to assess sustainability but which methodology is the most appropriate in a given situation.

Our approach to assess farm performance consists of two successive steps. First, the economic farm performance and the structural change were studied (see chapter 5 and 6), which gave us a detailed contextual analysis. The next step was the integration of environmental (and social) resource use in the economic analysis to assess farm sustainability performance (see chapter 7 and 8). It is our conviction that in this way an important step is made in the assessment of farm sustainability. Possible future steps are the use of other assessment methods or the incorporation of effectiveness considerations (see section 9.5).

9.2.2 The sustainable value approach

One interesting method to measure firm contributions towards sustainability is the sustainable value method. We showed by using this methodology that it is possible to assess the sustainable development of agriculture in an integrated way that provides good guidance for decision making. Analyzing the evolution of sustainable value is more accurate than analyzing the evolution of value added, because environmental resources are integrated in the calculation of sustainable value. Policy makers and company managers can use the sustainable value approach to measure, monitor and communicate their sustainability performance. Furthermore, the sustainable value approach can be used to identify characteristics of out- and underperformers. The sustainable value approach can be applied as input for policy tools and stimulate better integration of decision-making. The sustainable value results can encourage public participation in sustainability discussions. However, the approach does not indicate whether the overall resource use is sustainable, but only how much a company contributes to a more sustainable use of its resources. Another drawback is that the methodology depends on the availability of data on corporate resource use and on the benchmarks of the different forms of resources. Inevitably, only the resources are selected on which reliable data exists. However, this does not reduce the possibilities of the sustainable value approach to integrate economic, environmental and social resources in a coherent way.

As explained in part I, several notions exist to describe sustainability. The sustainable value approach calculates the sustainable value at micro level but the approach leaves the total amount of each resource unchanged at the macro level. It can thus be described as a strong sustainability approach. However, the sustainable value approach assumes perfect substitution between users (or systems). The approach is in fact not focused on substitution of resources but based upon reallocation of resources. This highlights the limitation of the weak versus strong sustainability framework, because this framework only considers substitution of resources. Nevertheless, the sustainable value framework can be useful to study the possibilities to reduce the resource use by for example introducing carrying capacity constraints instead of leaving the amount of resource use unchanged.

The novelty of this dissertation is that the sustainable value approach was applied on small and medium sized enterprizes (instead of using multinational firms). Because of the large amount of observations available in agriculture, the farm sector presents possibilities to explore the potential of the sustainable value approach. The use of a representative group of farmers made a statistical analysis possible, and thus the significance of our results could be analyzed in contrast with previous studies using the methodology.

The sustainable value approach can be seen as an improved measure of eco-efficiency (see section 4.3.7.2). It is an improvement because the sustainable value approach can take several social and environmental impacts simultaneously into account. Furthermore, the sustainable value approach accounts for rebound effects by restricting the total resource use at the macro level. We can say that the sustainable value approach is appropriate for analyzing sustainability, although the analysis is still simplified by disregarding non-linearities and dynamics. However, through combining the sustainable value approach with efficiency analysis, thus allowing non-linearities to be taken into account, we realized the next step in the further methodological improvement.

9.2.3 The importance of benchmarks

The value-orientated sustainable value approach compares sustainability between systems by comparing the resource productivity between peers and this for each resource. As mentioned before, benchmarking can help farmers and policy makers to highlight opportunities for improvement and indicate where best practices might be found. The choice of the benchmark reflects a judgement, as it determines the cost of all resources. The choice of the most appropriate benchmark is thus very important. There are several possible benchmarks, each with advantages and disadvantages. The choice of the benchmark depends on the particular objectives. A best performance benchmark can be very useful within the scope of policy analysis or for choosing the appropriate actions to realize the firm's objectives. Benchmarks can give valuable signs to all decision makers. A well defined benchmark is essential, as otherwise decision support systems can give wrong signals, resulting in wrong decisions. Furthermore, it is important that a benchmark is realistic and feasible for each company. Therefore, we defined benchmarks using efficiency and frontier analysis. In this way, the production theoretical underpinnings of efficiency analysis enrich the sustainable value approach.

Using the maximum attainable production possibilities as benchmark offers several advantages. First, substitution possibilities between different resources (economic and environmental) are not ignored as in traditional eco-efficiency analysis. Second, the constructed benchmark takes inefficiency of the considered resources into account. Third, using frontier methods to construct benchmarks provides specific benchmarks for each company adjusted to the particular situation of the company (in other words to the actual resource use). The sustainability of each company is assessed in comparison to its relevant peers. Feasible targets can help to motivate decision makers (farm managers and policy makers) to take realistic but ambitious measures towards sustainability. Fourth, our approach can be used to simulate and estimate the impact of policy measures on firm sustainability . In this way, this method can be used as an integrative sustainability assessment tool for policy measures. A benchmark, indicating the maximum attainable productivity level, is useful to analyze the efforts of firms in their aim towards best performance. Policies targeting efficiency improvements tend to be more easily adopted than policies that restrict the level of economic activity.

9.3 Summary of the empirical results

The conceptual and theoretical framework of this dissertation (part I) was described in a broad way, considering different notions of sustainability and several methods to assess (sustainability) performance. It is necessary to define sustainability in considerably narrower terms than the general and vague sustainability definitions in order to establish operational rules of thumb to assess sustainability. We restricted our applications to assess the sustainability of agricultural firms.

The main objectives of the empirical applications were to measure farm performance and to explain differences in performance. The first two applications measured economic performance, while the other applications measured performance in a more integrated way, described as sustainability performance. The following conclusions can be drawn from the results of the empirical analysis in this dissertation (part II).

In the first application, the farm performance was measured with efficiency techniques. The agricultural sector faces a continuous process of structural change with consequences for productivity and efficiency of farming. A sound way for monitoring this process is measuring technical efficiency with a stochastic frontier model. An empirical model was estimated to study the impact of managerial and structural characteristics on farm efficiency of Flemish farms. Not only the performance measurement itself, but also the insights why farms differ in their efficiency, can result in new knowledge of the process of structural change and can provide feedback to the concerned policies and government interventions. The empirical results lead to the following conclusions. Several structural characteristics explain differences in efficiency: location of the farm, type of farm sector, farm solvency, farm size, land property and the dependency on support payments. In general, larger farms are working more efficiently. Farms with a high share of land in own property seem to have a higher efficiency. Financial determinants have an impact on efficiency: solvency (own capital/total capital) has a negative impact on efficiency. A possible explanation is that farmers with a low solvency have higher repayment obligations and those farmers are stimulated to work more efficiently. Another possible explanation is that these farms have invested in more modern technologies and thus are able to increase their efficiency. Further, the more a farm depends on support payments, the lower its efficiency in using its resources. Besides structural characteristics, managerial characteristics also have an impact on farm efficiency. Farm managers with a higher level of education are more efficient than farmers with a lower level of education. Farms with a successor are more efficient than farms without a successor. Analyzing the non-linear relation between efficiency and the age of the farm managers, we found an increasing positive relation till a certain age, afterwards the relation becomes negative. Lower solvency rates and higher education increase the impact of the age level on efficiency. Note that the approach we used is unconditional, meaning that we assumed that the farm characteristics have a possible impact on efficiency but that efficiency has no impact on the farm characteristics.

The second empirical application investigated the determinants of structural change in agriculture. Understanding structural change is important for sustainability, because of the implications for the performance of farming, agricultural output and resource use. This application investigated the link between structural change and farm performance. More specifically, the research was restricted to analyze the link between farm growth and farm efficiency. Using a large data set of Flemish farms, their growth and efficiency was calculated and the impact of efficiency on growth was analyzed. Theoretically, efficient farms grow and survive, inefficient ones decline and fail. Starting from Gibrat's Llaw and the passive learning model of Jovanovic (1982), a growth model was constructed. Model estimations showed that in general, the efficient farms grow while the inefficient farms decline in size. Furthermore, we found differences in farm efficiency among farmers but also differences between agricultural subsectors (sectoral heterogeneity) that partly explain structural change in agriculture. In strongly regulated agricultural subsectors (e.g., dairy subsector), the impact of farm efficiency on farm growth showed to be not significant, indicating that policy measures (e.g., milk quota) affect the link between farm efficiency and farm growth.

Policy makers aim to combine strong economic performance and sustainable use of natural resources. Therefore, it is important to assess and measure farm sustainability. In a third empirical application, the sustainable value of farms was calculated to measure sustainability performance. This integrated valuation method is based on the concept of opportunity costs and can be seen as an improved eco-efficiency measure. This approach developed by Figge and Hahn (2004*a*) and Figge and Hahn (2005) is value orientated. It analyzes how much value has been created with a set of resources compared with the use of these resources by a benchmark. Their approach is focused on the scale of resource use and is based on the notion of strong sustainability because the sustainable value approach expresses the excess value created by a company, while preserving a constant level of each resource use on the macro level. It does not consider inter- or intragenerational aspects. In this application, the sustainable value approach was applied on agricultural farms for the first time. Our analysis showed that the sustainable value approach is suitable to assess farm sustainability. It may cover the use of economic, environmental and social resources in the farming sector and thus integrate economic, ecological and social challenges. Hence, sustainable value provides an integrated monetary assessment of the sustainability performance of farming enterprizes. But more important than measuring the creation of sustainable value is analyzing the difference in sustainability performance. Using panel data of Flemish dairy farms, an effect model was estimated to capture the determinants of sustainability performance. Once again, we found that both structural and managerial characteristics have an impact on farm performance. In general, young farm managers show to have a higher sustainability performance. Furthermore, larger farms are having a higher return-to-cost ratio (indicating sustainability performance), than smaller farms. Also, the dependency on support payments has a significant negative impact on performance. Note that similar impacts of managerial and structural aspects were found on both economic performance (measured as technical efficiency) and sustainability performance (measured as return-to-cost ratio). Furthermore, our analysis revealed that over the observed period, the same farms were found to contribute most towards sustainability, indicating the existence of frontrunners. Analyzing the link between economic performance and sustainability performance, we found that in general, economic performance goes hand in hand with sustainability performance. Hence, our results lead to the conclusion that sustainable farms realize both good economic and environmental results. We found a low correlation between financial capital productivity and sustainability performance. This may indicate that increasing the sustainability performance of dairy farms requires financial capital, or in other words that there may be a trade-off or substitution between financial capital and other resources.

In a final empirical application, the sustainable value methodology was improved. The methodology was combined with efficiency analysis to benchmark sustainability. In this way, the theoretical underpinnings of efficiency analysis enrich the sustainable value approach. The methodology was illustrated with two functional forms: the Cobb-Douglas and the translog functional form. The Cobb-Douglas functional form is very attractive because it is easy to estimate and to interpret. However, a major drawback of the Cobb-Douglas functional form is the lack of flexibility. The value contributions of the different resources are identical because of the fixed elasticity of substitution (equal to 1). A possible solution is to use the translog functional form which is more flexible and allows to take substitution between resources into account. Our example for Flemish dairy farms showed that labor is used less productive than farm capital. Compared to labor, farm capital was used in a more value-creating way, or better in a less value-wasting way. Combining the results of the third and last empirical application, we see that financial capital is needed to increase the sustainability performance of farms and Flemish dairy farms are using farm capital in a value-creating way. A disadvantage of the stochastic translog functional form is the data requirement, as many data are needed to avoid estimation problems such as multicollinearity problems. Using frontier methods for deriving firm specific benchmarks has the advantage that the particular situation of each company is taken into account when assessing its sustainability.

9.4 (Policy) recommendations

A first recommendation is that integrated performance indicators should be used more often. Not only economic performance indicators should be used to assess performance but also sustainability performance indicators. Some actions can result in economic improvements but in an overall decrease of performance if these actions have a strong negative impact on environmental and/or social criteria. We found that in general, economic and sustainability performance go hand in hand.

We applied the sustainable value approach to assess farm sustainability. As mentioned, this approach can be seen as a strong sustainability approach on the macro level because the total amount of each resource remains unchanged. An increase in total resource use is not allowed. However, more importantly, the approach focuses on the reallocation of resources, which can be of great interest for policy makers. It seems self-evident that policies should favor farmers who use their resources in a more sustainable way but so far a method to assess sustainability between resource users is not common in practice. The sustainable value approach is extremely suitable to support decision makers in their selection of *good resource users* and thus to target this group. Policy makers can then decide to reward good performers or decide to help bad performers to improve their sustainable resource use. Hereby providing subsidies should be avoided as we found that dependency on support payments is negatively correlated with farm sustainability. Note however that the analyzed subsidy schemes have not the objective to reward good performers. Hence, it is possible that other subsidy schemes with a focus on performance could have a positive correlation with farm sustainability. Besides, an interesting way is to use good performing farms as examples for the sector as a whole. Sustainable farms may be used as a mirror for future farms. Therefore, it is essential to develop and use methods to identify sustainable farms.

The proposed performance measures (e.g., return-to-cost ratio) analyze the farm performance in an integrated way and are therefore more interesting than current indicators such as productivity or eco-efficiency. Furthermore, such indicators do not compare observations with a benchmark. Comparing actual performance with the performance of a benchmark can be very useful to guide farmers towards sustainability. The suggested approach could help decision makers to identify farms that best suit policy objectives. It also provides information to what extent resource use can be improved conditional on the current technology. Moreover, we did not only measure farm performance, we also tried to explain differences in performance. Insights in the underlying determinants is essential for policy development.

Our empirical results lead to the following policy recommendations. Policies that stimulate the transfer of resources used by non-efficient farms to efficient farms will improve the performance of the overall agricultural sector. Our results indicate that it may be crucial that farms without successor stay not too long in business. Taking over their production factors by other more efficient farmers will result in a higher performance of the sector. Furthermore, improving the education level of farmers will result in a higher overall performance. Size has a positive impact on economic and on sustainability performance, indicating that policies that hamper the growth strategy of farms, should be avoided. Note however that in our measurement of sustainability performance, we did not take social aspects into account. It may be that larger (industrial) farms score less with respect to social criteria.

Investigating the link between farm growth and efficiency, we found that in general efficient farms increase in size while inefficient farms decrease in size. Nevertheless, we found evidence of sectoral heterogeneity indicating that policy measures and other aspects (e.g., food crisis) in certain agricultural subsectors could affect this link. Our analysis does not show that farms have to grow to become more efficient but learns that efficient farms have the tendency to grow. Policy makers aiming to increase the efficiency of farming, must take into account that their policy measures will result in structural changes. Subsidies should be applied with care because our results also suggest a negative relation between subsidies and performance. Apparently, farms depending on subsidies are not stimulated to search for higher value added solutions while a high value added proves to be very important for both the economic performance and the sustainability performance of farms. Therefore, policies should give incentives to develop value added strategies rather than keeping less economic and unsustainable farms in production. Our results indicate for example that stimulating on-farm selling of farm products (or other diversified activities) can contribute to a more sustainable dairy sector in Flanders. Of course, it may be that subsidies result in certain positive contributions such as landscape amenities or have social benefits (e.g., survival) which were not included in our analysis.

It is interesting to know which firms are creating value considering all relevant resources, but it is also crucial to know the impact of (future) decisions on the sustainable value. Evaluating two basic policy options, we showed that our approach is promising in this respect and may be able to evaluate the impact of potential decisions within an integrated sustainability framework.

9.5 Recommendations for future research

A first observation is that empirical research to assess the sustainability performance of farms is still scarce. It still seems there are more papers describing theoretical aspects and models of sustainability than papers using empirical data to describe, measure, analyze, explain and assess contributions to sustainability.

In this dissertation, we measured performance in terms of efficiency. Efficiency relates the used resources to the obtained results. Measuring performance as efficiency is however not sufficient to assess performance entirely. Performance should also be measured in terms of effectiveness. This concept compares the results to the desired outcomes or objectives, dealing with the degree of failure or success. Hence, to assess farm sustainability both the efficiency and effectiveness of the use of farm resources should be analyzed. Further research that incorporates efficiency and effectiveness analysis is certainly needed.

Considering the sustainable value approach to assess sustainability, several recommendations for future research can be formulated. More empirical applications using data from all relevant resources can be useful to describe the sustainability performance of companies. Other environmental and social resources such as air and soil quality, as well as the quality of life should be considered. Contributions of farming to society such as contributions to biodiversity or landscape creation should also be incorporated into the calculation of the sustainable value of farms. So far, the relevant resources were based on literature and the availability of data. But with the increased collection of data on several environmental and social aspects (e.g., CO_2 contribution, animal welfare) the scope for further research will certainly become wider.

In our research, we only made an intra-sector comparison, showing only the potential for improvements within a given activity. This implies that the agricultural sector remains constant and that dynamics are not taken into account. Comparing the sustainability performance of farms of different agricultural sectors would be a very interesting and challenging topic. Another interesting topic is the analysis of the sustainable performance up or down the value chain. Furthermore, as mentioned the sustainable value approach can be seen as a strong sustainability approach because the total amount of each resource use remains constant at the macro level. To strengthen the strong sustainability approach, the sustainable value approach can be redefined by introducing carrying capacity constraints.

Besides further improvements and further use of the sustainable value approach, empirical applications using other approaches are also needed. A diverse use of methodologies to assess sustainability fits with the definitional diversity of sustainability.

No matter which method is applied, the selection of resources is essential. More research is necessary to analyze which resources are critical in the assessment of sustainability. Furthermore, it is important how these resources should be treated and defined. Several questions still remain without a clear answer: - How can we link environmental resources to environmental impacts? - How should we treat and model environmental resources in economic models (as inputs or as bad outputs)? - Should we aggregate certain resources and how should we aggregate them? - Scarcity is essential in defining resources but how can we define resources in a comprehensive and clear way? - How do we avoid double counting of certain resources and impacts? - What about the multiple effects of certain resources? - How do we consider direct and indirect resource use? - How far will we go with taken into account indirect resource use and indirect impact? These questions are especially relevant in striving towards a more integrative way of assessing sustainability.

Several methods exist to assess sustainability but few offer possibilities for ex ante assessment of policy tools towards sustainability. In chapter 8 we showed that our approach has a certain potential in this respect. Further development of this potential is certainly a challenging topic.

Summary

Summary

Sustainability can be seen as a key element towards a profitable long-term future for farming and rural areas. Strong economic performance should go hand in hand with the sustainable use of natural resources. To assess sustainability, both a clear framework and empirical work measuring, explaining and evaluating contributions towards sustainability are needed.

In the first part of this dissertation several existing definitions of sustainability are explained. An overview is also given of the different measurement methods of sustainability performance. There is no single description that captures the broad diversity of sustainability. Definitions of sustainability are inevitably vague but sustainability draws much of its power and creativity form this ambiguity. The following aspects often return within the sustainability debate: (i) natural resources are finite and there are limits to the carrying capacity of the Earth's ecosystems, (ii) economic, environmental and social goals must be pursued within these limits, and (iii) there is a need for inter- and intragenerational equity.

It is not only essential to describe the meaning of sustainability and to recognize the importance of sustainability, but it is also crucial to measure progress towards sustainability. Measuring sustainability means that it has to be defined in more narrow terms in order to establish operational rules for sustainability. We can formulate four main conclusion form our overview of the different methods to measure sustainability. First, many different methods exist and each method has its own pros and cons. Second, the majority of the sustainability assessment methods have recently been developed and methodological improvements are possible, often even necessary. Third, there is a growing

amount of empirical applications using the developed methods to assess sustainability but empirical applications are still scarce. Any meaningful analysis of sustainability needs to pay attention to indicate which concepts and assumptions are used in the measurement methodology. Fourth, using a combination of methods is recommended, for example using both biophysical and monetary assessment approaches. The choice of assessment method depends on the exact research question, the sustainability notion (e.g., weak or strong) and the underlying assumptions and the level considered (e.g., nation, industry or company level).

The second part of this dissertation presents several empirical applications of measuring farm performance. The first applications measure performance in a traditional economic way, while in the following applications environmental aspects are integrated to measure the sustainability performance of agricultural firms. The main objectives of the empirical applications were to measure farm performance and to explain differences in farm performance.

Measuring economic performance as technical efficiency, we found that several structural and managerial farm characteristics explain differences in performance such as farm size, education of farmer (positive impact), farm solvency, dependency on support payments and the farmer's age (negative impact). Our results of the analysis of the link between structural change (e.g. farm growth) and farm performance (technical efficiency), illustrate that in general the efficient farms grow while the inefficient farms decline in size. Differences in farm efficiency among farmers but also differences between agricultural subsectors partly explain structural change in agriculture.

In a third empirical application the sustainable value approach, developed by Figge and Hahn (2005), is used to measure progress towards farm sustainability. The sustainable value approach seems very promising to assess contributions towards sustainability. The method has the advantage to look from the value added point of view and not from the negative externality point of view. Therefore, in our opinion this methodology may be a powerful tool, not only to assess firm sustainability, but also to guide companies towards sustainability. An integrated and balanced performance indicator can be calculated starting from the available resources and their contribution to the value added of a farm. The approach makes the options for sustainability improvement more concrete, interesting and realistic for both managers, seeking a private economic optimum, as for policy makers, seeking a more social welfare optimum. Analyzing the determinants of sustainability performance of Flemish dairy farms, we found once again that both managerial and structural farm characteristics such as farm size (positive impact), farmer's age and dependency on support payments (negative impact) have an impact on performance. Furthermore, our results reveal that sustainable farms realize both good economic and environmental performance. Our results also showed that a high sustainable value goes hand in hand with a more productive resource use of land, labor and environmental resources but not with a more productive use of financial capital. This may indicate that increasing the sustainability performance of dairy farms requires financial capital or in other words that there may be a trade-off or substitution between financial capital and the other resources.

In a final empirical application, the sustainable value methodology to assess integrated farm performance is combined with efficiency analysis. In this way, the theoretical underpinnings of efficiency analysis enrich the sustainable value approach and farm specific benchmarks could be derived. This has the advantage that the particular situation of each company can be taken into account in the assessment of its sustainability. Our results show that the Cobb-Douglas functional form is very attractive to use as benchmark to calculate the sustainable value of dairy farms. However, the Cobb-Douglas functional form lacks flexibility and a lot of assumptions have to be made. The translog functional form is more flexible but has the disadvantage that it requires a lot of data to avoid estimation problems. The advantage of using the translog functional form is that besides the assessment of sustainability between farms, also the resource use within farms can be analyzed. We found that Flemish dairy farms use farm capital in a more value-creating way than labor.

As a general conclusion, we can summarize that the developed and used methodologies are useful to measure contributions of farms towards a more sustainable use of the available resources. This approach does not indicate whether the overall resource use is sustainable, but how much a company contributes to a more sustainable use of its resources. Not only traditional economic resources such as land, labor or capital are considered but also environmental resources such as direct and indirect energy use and nitrogen use. In this way, the realization of economic value of a farm using a combination of available resources can be analyzed. In this way, we can compare the performance of farms but we can also analyze the underlying determinants that can help to explain the difference in sustainable resource use between farms. Furthermore, a simple simulation shows that the approach can be used to compare policy tools or can be used to analyze the impact of certain policy measures to reallocate the resource use from farms realizing less value added to farms realizing more value added with their resources. This will be a challenging but interesting future research topic. A limitation of the current approach is that the total amount of available resources is remained constant at sector level.

Samenvatting

Samenvatting

Duurzaamheid kan beschouwd worden als een belangrijke lange termijn doelstelling voor landbouw en het platteland. Het landbouwbeleid streeft dan ook naar een combinatie van duurzaam gebruik van de beschikbare natuurlijke hulpbronnen én sterke economische prestaties van de landbouwsector. Om duurzaamheid te kunnen beoordelen, is echter zowel een duidelijk kader als empirisch onderzoek vereist om de progressie naar een duurzame landbouw te meten, te verklaren en te evalueren.

In het eerste deel van dit doctoraat worden verschillende duurzaamheidsbegrippen toegelicht. Ook wordt een overzicht gegeven van de verschillende methoden om duurzaamheidsprestaties te meten. Uit de literatuur blijkt dat het begrip duurzaamheid erg divers wordt geïnterpreteerd zodat het quasi onmogelijk is een eenduidige definitie te formuleren. Daarom zijn bestaande beschrijvingen van duurzaamheid onvermijdelijk vaag maar aan de andere kant resulteert deze diversiteit aan definities juist in de nodige creativiteit. De volgende aspecten keren geregeld terug in duurzaamheidsbeschrijvingen: (i) natuurlijke hulpbronnen zijn eindig en de draagkracht van onze ecosystemen is beperkt, (ii) economische, ecologische en sociale doelstellingen dienen verwezenlijkt te worden binnen deze beperkingen en (iii) er is nood aan gelijkheid zowel tussen als binnen generaties.

Niet alleen de beschrijving van duurzaamheid en de erkenning van het belang van duurzame ontwikkeling zijn belangrijk, ook het meten van de vorderingen naar meer duurzaamheid is essentieel. Duurzaamheid meten, betekent dat we dit veelomvattende begrip duidelijk dienen af te lijnen en te vertalen naar operationele regels. Op basis van een overzicht van verschillende bestaande benaderingen om duurzaamheid te meten, kunnen volgende conclusies worden geformuleerd. Ten eerste blijken er verschillende methoden te bestaan met telkens voor- en nadelen. De meerderheid van deze methoden zijn pas recent ontwikkeld en verbeteringen zijn zeker mogelijk en soms zelfs wenselijk. Ten tweede stijgt stilaan het aantal empirische toepassingen die gebruik maken van één of meer methoden om duurzaamheid te beoordelen. Toch blijft er een sterke nood aan meer empirisch onderzoek. Toepassingen dienen ook duidelijker aan

te geven welke duurzaamheidsconcepten en aannames aan de basis liggen van de toepassing. Tot slot bevelen we een combinatie van methoden aan om een duidelijker beeld te krijgen van de duurzaamheid van landen, regio's, sectoren of bedrijven. Een combinatie van zowel biofysische als monetaire benaderingen is wenselijk. De methodologische keuze hangt immers sterk af van de onderzoeksvraag, het duurzaamheidsconcept (bv. sterke of zwakke duurzaamheid) en de bijhorende aannames en de schaal van de toepassing (bv. land, sector of bedrijf).

Het tweede deel van dit doctoraat bestaat uit empirisch onderzoek dat de bedrijfsprestaties in de Vlaamse landbouw tracht te meten en te verklaren. De eerste twee toepassingen meten de efficiëntie en structuurontwikkeling van de bedrijven op een traditioneel economische manier, terwijl we in de volgende toepassingen ecologische aspecten integreren om zo de duurzaamheid van landbouwbedrijven te kunnen beoordelen. De belangrijkste doelstellingen van de empirische toepassingen zijn het meten van bedrijfsprestaties en het verklaren van de prestatieverschillen tussen bedrijven.

Een belangrijke economische indicator om bedrijfsprestaties te meten is technische efficiëntie. Uit onze analyse van de technische efficiëntie komt naar voor dat verschillende structurele en persoonsgebonden eigenschappen de verschillen in bedrijfsprestaties verklaren. Zo vonden we dat bedrijfsgrootte en het opleidingsniveau van de bedrijfsleider een positieve impact hebben op efficiëntie terwijl solvabiliteit, de afhankelijkheid van subsidies en de leeftijd van de bedrijfsleider een negatief effect hebben op de economische prestaties van landbouwbedrijven. Daarnaast onderzochten we het verband tussen structurele veranderingen (zoals de groei van landbouwbedrijven) en de bedrijfsprestaties. We vonden dat efficiënte bedrijven groeien terwijl inefficiënte bedrijven afnemen in grootte. Niet alleen verschillen in efficiëntie tussen bedrijven, maar ook de sector waartoe het bedrijf behoort, is belangrijk voor het verklaren van structurele veranderingen.

In een derde toepassing wordt voor het eerst in de literatuur de door Figge en Hahn (2005) ontwikkelde duurzame-waarde methode voor het meten van de duurzaamheidsbijdrage van ondernemingen toegepast op landbouwbedrijven. De duurzame-waarde benadering is zeer interessant om vorderingen op het gebied van duurzaamheid te beoordelen. Het voordeel van deze methode is dat ze vertrekt vanuit de toegevoegde waarde die bedrijven produceren met de ingezette hulpbronnen en niet vanuit de negatieve externaliteiten die worden veroorzaakt. Door de toegevoegde waarde te relateren aan alle ingezette hulpbronnen kan berekend worden in welke mate een bedrijf de ingezette hulpbronnen op een meer productieve wijze gebruikt dan andere bedrijven. We zijn van oordeel dat deze methode niet alleen nuttig kan zijn om duurzaamheid te meten maar ook om bedrijven te begeleiden richting meer duurzaamheid. De methode meet de bijdrage van de ingezette hulpbronnen van een bedrijf om toegevoegde waarde te creëren en komt zo tot een geïntegreerde en gebalanceerde indicator om bedrijfsprestaties te vergelijken. Deze benadering maakt de keuze voor duurzaamheidsverbeteringen meer concreet en realistisch voor zowel bedrijfsleiders, op zoek naar een privaat economisch optimum, als voor beleidsmakers, op zoek naar een sociaal economische optimum. Gebruik makende van de duurzame-waarde methode worden in het onderzoek de mogelijke determinanten van de duurzaamheidsprestaties van Vlaamse melkveebedrijven bestudeerd. We vonden ook hier dat zowel structurele als persoonsgebonden eigenschappen een invloed hebben op bedrijfsprestaties. De grootte van landbouwbedrijven heeft een positieve impact op de duurzame bedrijfsprestaties, terwijl de leeftijd van de bedrijfsleider en de afhankelijkheid van subsidies een negatieve impact hebben op de duurzaamheidsprestaties van landbouwbedrijven. Verder toonden we aan dat landbouwbedrijven met een hoge duurzame waarde meestal goed scoren zowel op economisch als op ecologisch vlak. Verder blijkt dat een hogere duurzame waarde samenhangt met een productiever gebruik van land en arbeid en andere hulpbronnen, maar niet van kapitaal, wat wijst op een zekere substitutie. Verbetering van de duurzaamheidsprestaties van melkveebedrijven gaat m.a.w. gepaard met een toename van de kapitaalsinzet.

In een laatste empirische toepassing leveren we een bijdrage aan de verdere theoretische onderbouw en verfijning van de duurzame-waarde methode door ze te combineren met efficiëntieanalyse. De beoordeling van de duurzaamheidsprestaties van bedrijven met de duurzame-waarde methode hangt immers sterk af van de vergelijkingswaarde die wordt gehanteerd. Door via efficiëntieanalyse een bedrijfsspecifieke (en dus meer haalbare) vergelijkingswaarde te bepalen, kunnen we bedrijfsspecifieke richtwaarden formuleren. Dit heeft als voordeel dat de specifieke situatie van elk bedrijf afzonderlijk in rekening wordt gebracht om de duurzaamheid te beoordelen. Uit de toepassing blijkt dat de Cobb-Douglas functionele vorm gemakkelijk te gebruiken is als richtwaarde voor de duurzame-waarde beoordeling. De translog functionele vorm heeft daarentegen het voordeel meer flexibel te zijn (minder assumpties) maar er zijn veel gegevens nodig om deze functie te kunnen schatten en gebruiken als richtwaarde. Zo kunnen we met behulp van de translog functionele vorm de duurzame waarde niet alleen gebruiken om bedrijven te vergelijken maar ook om hulpbronnen te vergelijken binnen bedrijven. Zo bleek uit onze berekeningen dat Vlaamse melkveebedrijven bedrijfskapitaal op een meer duurzame wijze inzetten dan arbeid.

Als algemene conclusie kunnen we stellen dat de door ons toegepaste en ontwikkelde methoden toelaten om de bijdragen van landbouwbedrijven tot een duurzaam gebruik van de beschikbare hulpbronnen te meten. Hierbij wordt niet gemeten in welke mate individuele bedrijven al dan niet duurzaam zijn maar wel in welke mate een bedrijf er in slaagt om meer of minder toegevoegde waarde te halen uit de in een sector ingezette hulpmiddelen. Naast de klassieke in de economie gehanteerde hulpbronnen zoals land, arbeid en kapitaal beschouwen we ook ecologische hulpbronnen zoals de ingezette nutriënten of energie. Zo kan binnen een sector worden nagegaan welke bedrijven de hoogste economische waarde realiseren met deze hulpbronnen. Dit laat niet enkel toe bedrijven met elkaar te vergelijken maar ook te onderzoeken welke factoren een meer duurzaam gebruik van hulpbronnen kunnen verklaren. Via een eenvoudige simulatie werd ook aangetoond dat de methode perspectieven biedt om beleidsinstrumenten met elkaar te vergelijken of na te gaan in welke mate beleidsmaatregelen er in slagen hulpbronnen te verschuiven van bedrijven die er minder toegevoegde waarde mee creëren naar bedrijven die er meer toegevoegde waarde mee realiseren. Dit is zeker een veelbelovende piste voor toepassing en verder onderzoek. Een beperking is wel dat de huidige methode hierbij de totale hoeveelheden beschikbare hulpbronnen constant houdt.

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Scientific curriculum vitae

Steven Van Passel was born in Wilrijk (Antwerp) on May 12, 1979 and has followed secondary education at the Onze-Lieve-Vrouw-van-Lourdescollege in Edegem. He has received the degree of Master in Bioscience Enginering (agricultural science) at the Katholieke Universiteit Leuven in 2002 with great distinction. In 2005, he has also received the degree of Master in economics at the Katholieke Universiteit Leuven with great distinction.

In September 2002, he has started as scientific researcher at the Policy Research Centre of Sustainable Agriculture, where he worked on the development of tools and concepts to support model analysis of economic indicators and model development. Furthermore, he made economic studies of the Flemish agricultural sector on both farm and sector level. Besides, traditional economic analysis, he applied and developed integrative methods to analyze the sustainability of farming in Flanders. Since January 2007, he works at the Institute for Agricultural and Fisheries Research where he works on methods to assess the sustainability of agriculture. He also works on the European funded SVAPPAS-project (Sustainable Value Analysis of Policy and Performance in the Agricultural Sector).

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